# Assessment of compensatory measures for impacts of offshore windfarms on seabirds

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# Assessment of compensatory measures for impacts of offshore windfarms on seabirds

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## Keywords

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## Further information

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## Foreword

Natural England commission a range of reports from external contractors to provide evidence and advice to assist us in delivering our duties. The views in this report are those of the authors and do not necessarily represent those of Natural England.

Please note that there are parts of the report, which refer to the guillemot account, that have been redacted, as they relate to unpublished research findings. We hope that we will be able to publish the report in full shortly.

MacArthur Green

# Assessment of compensatory measures for impacts of offshore windfarms on seabirds


| | |
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**Note: this report contains material that was kindly provided prior to publication. The authors requested that this material be redacted until their findings are published.**

# CONTENTS

## LIST OF TABLES

MacArthur Green

MacArthur Green

## LIST OF FIGURES

## EXECUTIVE SUMMARY

Impacts on Special Protection Areas (SPAs) for seabirds are predicted to occur from the operation of offshore wind farms. In order to gain consent to construct and operate an offshore wind farm, an appropriate assessment of the potential impacts on SPAs needs to determine if there will be no adverse effect on site integrity. Where it is not possible to conclude this, a project can only be consented where there are no alternative solutions to the proposed project and there is an imperative reason of overriding public interest. When this occurs, it is necessary for compensation to the network of SPAs for the impacted feature(s) to be provided to maintain the coherence of the network.

This study reviewed the available literature for potential compensation measures for nine qualifying features of eight SPAs in England predicted to be impacted by offshore wind farm developments. Seven of these were breeding features:

- Breeding black-legged kittiwakes (*Rissa tridactyla*) at Flamborough and Filey Coast SPA;
- Breeding gannet (*Morus bassanus*) at Flamborough and Filey Coast SPA;
- Breeding guillemot (*Uria aalge*) at Flamborough and Filey Coast SPA;
- Breeding razorbill (*Alca torda*) at Flamborough and Filey Coast SPA;
- Breeding Atlantic puffin (*Fratercula arctica*) at Flamborough and Filey Coast SPA;
- Breeding Sandwich tern (*Thalasseus sandvicensis*) at North Norfolk Coast SPA; and
- Breeding lesser black-backed gull (*Larus fuscus*) at Alde-Ore Estuaries SPA.

Two were non-breeding features:

- Non-breeding red-throated diver (*Gavia stellata*) at Outer Thames Estuary SPA; and
- Non-breeding red-throated diver at Liverpool Bay SPA.

Following the review of potential compensation measures for each SPA feature, the potential for impacts to their populations to be compensated by recommended methods was assessed at three impact scenarios (low, medium and high) for three compensation scenarios (low, medium and high). This approach allowed different combinations of the three possible scales of impact and compensation measures to be explored.

Impact scenarios were largely based on existing cumulative impact levels predicted for each SPA qualifying feature (low impact scenario), a pro-rated estimate of the additional impacts that might arise from Round 4 offshore wind farms (medium impact) and a pro-rated estimate of the additional impacts from 100GW associated with net-zero electricity generation in 2050 (high impact). These impact levels were based on some realistic predictions combined with some additional larger scale predictions that were likely to be less realistic (pro-rata increases for the high impact scenario less are likely as wind farms move further offshore), but useful in understanding limits to compensation measures.

Compensation measure scenarios were based, wherever possible, on empirical evidence for population responses to these measures (high compensation). In most cases, medium compensation scenarios were based on halving the level from the empirical data source (medium compensation) and halving again (low compensation). As such high compensation scenarios were the most realistic and low compensation scenarios the most precautionary.

Confidence in the ability of compensation measures to overcome predicted impacts were based on a modified version of the Intergovernmental Panel on Climate Change (IPCC) guidance on communicating uncertainty. Confidence was assessed using various metrics on the empirical evidence of populations' responses to either impacts or management measures combined with the applicability of this evidence to the situation being assessed. This approach was then applied to the assessment method (e.g. Population Viability Analysis) used to study the efficacy of the compensation measure to the impact and the three scales described above. Narratives on these summaries of evidence and applicability were then used to describe and either justify the value of confidence reached, or to modify that confidence value up or down. These final confidence values (low, medium or high) were used to assess whether the assessment results at the three scales of impact and compensation had low, medium or high confidence of success.

Results showed that in most cases low and medium impact scenarios could be compensated for at high, medium or low compensation levels. Where the same compensation measure was recommended for more than one species, the confidence in these varied depending on the available evidence for the compensation measures and the confidence in those, or in species specific differences in ecology.

The closure of sandeel and sprat fisheries in UK waters was a recommended method for compensation for black-legged kittiwake, guillemot, razorbill, Atlantic puffin and Sandwich tern. Fisheries bycatch mitigation was a recommended compensation measure for northern gannets. Control of fox predation was a recommended compensation method for Sandwich tern and lesser black-backed gull. The creation of strict marine reserves within existing SPAs was a recommended compensation measure for red-throated diver.

It was apparent that closure of sandeel and sprat fisheries in UK waters was the single most important compensation measure that could be applied. This was predicted to positively affect multiple SPA features across a wide range of SPA in the UK and the assessment showed that this one measure could compensate for high levels of impact. As such this has the potential to be a strategic level compensation measure for the offshore wind industry, at least in the North Sea.

# 1 INTRODUCTION

Prior to exiting the European Union (EU), the United Kingdom (UK) Government and Devolved Administrations designated protected sites as required by the European Commission (EC) Birds [2009/147/EC] and Habitats Directives [94/43/EC]. This resulted in Special Protection Areas (SPA) being designated for breeding seabirds and non-breeding seabirds. These remain protected in the UK by the enactment of the European Directives into UK law (The Habitats Regulations). The Habitats Regulations require that relevant authorities responsible for issuing licences to or consents for plans or projects undertake an appropriate assessment of potential impacts of those plans or projects on the qualifying features of SPAs where no likely significant effect cannot be concluded. Where the appropriate assessment cannot determine that an adverse effect on site integrity can be ruled out beyond any reasonable scientific doubt, consent can only be granted when the competent authority can determine that there is an Imperative Reason of Overriding Public Interest (IROPI) and there is no alternative to the project. If these can be determined, there is a duty on the competent authority to ensure that compensation is provided to maintain the overall coherence of the SPA network. It is important to note that while the scope of this project was to examine the effects of compensation on specific impacted SPA qualifying features, the Habitats Regulations do not require compensation to be applied to the impacted SPA qualifying feature, only to "ensure that the overall coherence of [the UK site network] is protected". However, guidance from the European Commission (EC) and Defra states that there is an order of preference for the location of compensation measures (European Commission 2012). The EC guidance states that this preference is:

1) *Compensation within the Natura 2000 site provided the necessary elements to ensure ecological coherence and network functionality exist within the site.*
2) *Compensation outside the Natura 2000 site concerned, but within a common topographical or landscape unit, provided the same contribution to the ecological structure and/or network function is feasible. The new location can be another site designated as Natura 2000 or a non-designated location. In the latter case, the area must be designated as Natura 2000 site itself and be subject to all the requirements of the 'nature' directives.*
3) *Compensation outside the Natura 2000 site, in a different topographical or landscape unit. The new location can be another site designated as Natura 2000. If compensation takes place on a non-designated location, the area must be designated as Natura 2000 site itself and be subject to all the requirements of the 'nature' directives.*

It was therefore assumed for this study that measures that compensate the SPA affected would be the most favoured, followed by compensation at another SPA but within the same region and finally compensation at another SPA but outside the same region within the UK.

Offshore wind farms have the potential to impact on the qualifying features of SPAs for breeding and non-breeding seabirds through collision mortality, displacement effects, barrier effects, and possibly by habitat change and indirect effects on key prey species, although these last examples are less known. At present, the in-combination impacts predicted from offshore wind farms on the North Sea coast of the UK have proven to be sufficient for the competent authority to be unable to conclude no adverse effect on the integrity of more than one Special Protection Area. If these projects can be shown to have an IROPI and there is no alternative to the proposed development, then compensation will be required to maintain the overall coherence of the SPA network. This report builds on a previous assessment of possible compensation measures (Furness et al. 2013) for seabirds in the UK but is focused on specific SPA qualifying features and aims to provide a more in-depth assessment of compensation on the population of those features. It therefore has a particular focus on certain SPA sites in England, and considers through literature review and PVA scenario modelling, whether particular measures could fully compensate for predicted impacts of offshore wind on selected seabird features of these sites.

### 1.1 Aims and Objectives

The aim of the project was to provide Natural England with an evidence-based assessment of the likely effectiveness of compensatory measures that may be proposed to offset predicted impacts on populations of the SPA qualifying features of interest, either in the context of the consenting process for individual offshore windfarm proposals or strategic plan-level Habitats Regulations Assessments. This report also provides a qualitative estimate of the degree of confidence that can be attached to each assessment of likely effectiveness. The project's findings aimed to be sufficiently robust and specific to guide Natural England's decision making in giving advice to developers, regulators etc. regarding its views on compensation measures regarding the key SPA qualifying features. The findings also aimed to be sufficiently general to inform Natural England's decision making in giving advice in other cases too.

The objectives of the project were defined by Natural England as:

1. identify the key biological questions that must be answered in order to inform an evidence-based judgement regarding the likelihood of success of implementing the suggested compensation measures in various locations in offsetting impacts of varying magnitude on various key SPA qualifying features.
2. review the literature and/or conduct analyses to answer as many of these questions as possible.
3. draw up a set of plausible effect levels and resultant population level impacts to be explored for each SPA qualifying feature.
4. draw up for each compensation measure for each SPA qualifying feature a range of scenarios reflecting different ways in which the measure might be implemented:
    - the nature of the intervention (e.g. the source of the pressure which is the target of the measure);
    - the location of the intervention (at the impacted site or elsewhere);
    - the magnitude of the predicted impact being compensated for;
    - the size of the compensation ratio used to upscale the target of the measure relative to that estimated to be "needed" to offset the predicted impact (to reflect the degree of uncertainty of success).
5. present a review of the evidence and use the results of the literature review and any analyses to reach an evidence-based judgement of:

i. the sufficiency and likelihood of success of each scenario considered under each SPA qualifying feature compensation measure for each impact level; and

ii. the confidence that can be attached to each of those assessments.

6. set out the key constraints to the successful implementation of each suggested SPA qualifying feature compensation measure, review how these constraints vary (e.g. geographically) and identify the situations (e.g. areas) in which such constraints are least strict for each suggested SPA qualifying feature compensation measure.
7. present an evidence-based, high-level review of the scale of compensation ratios likely to be needed to yield a reasonable guarantee of success for each suggested SPA qualifying feature compensation measure.
8. summarise key findings in a series of colour-coded tables/matrices.
9. identify further research needed regarding the most poorly understood aspects of each species' ecology which, if improved, could significantly increase the degree of confidence that can be attached to assessments of the likelihood of success of various compensatory measures.

## 1.2 SPA qualifying features

The subjects of this study were qualifying features of SPAs where Natural England have previously advised that an adverse effect on site integrity could not be ruled out beyond reasonable scientific doubt or have ongoing concerns that this may be the case in the near future (Table 1).

**Table 1 SPA qualifying features which are the subject of this study.**

| SPA | Qualifying feature |
|---|---|
| Flamborough and Filey Coast SPA | Breeding black-legged kittiwake *Rissa tridactyla* |
| | Breeding northern gannet *Morus bassanus* |
| | Breeding common guillemot *Uria aalge* |
| | Breeding razorbill *Alca torda* |
| | Breeding Atlantic puffin *Fratercula arctica* * |
| North Norfolk Coast SPA | Breeding Sandwich tern *Thalasseus sandvicensis* |
| Alde-Ore Estuary SPA | Breeding lesser black-backed gull *Larus fuscus* |
| Outer Thames Estuary SPA | Non-breeding red-throated diver *Gavia stellata* |
| Liverpool Bay/Bae Lerpwl SPA | Non-breeding red-throated diver |

* Feature of the breeding seabird assemblage.

## 1.3 Location of compensation

Guidance on the location of compensation measures from Defra recommends two levels of hierarchy:

- "*Measures that replicate or benefit the same feature within the affected site*"; and
- "*Measures that replicate or benefit the same feature outside the affected site*".

European Commission guidance on compensation measures states that, "*Compensatory measures should be located to accomplish the highest effectiveness in maintaining the overall coherence of the Natura 2000 network*". Consequently, three levels of hierarchy of preferred locations are given as:

- "*Compensation within the Natura 2000 site*";
- "*Compensation outside the Natura 2000 site concerned, but within a common topographical or landscape unit*"; and
- "*Compensation outside the Natura 2000 site, in a different topographical or landscape unit*".

Thus, Cook et al. (2011) was used to determine the geographic scale for breeding seabirds that can be matched to "a common topographical or landscape unit" and Furness et al. (2015) for non-breeding seabirds.

### 1.4 Key biological questions

Natural England wishes to identify the key biological questions that need to be answered when considering potential compensation measures for the SPA qualifying features listed in Table 1. These questions will need to be answered in order to inform an evidence-based judgement regarding the likelihood of success of implementing various compensation measures in various locations to compensate for estimated impacts of varying magnitudes on various SPA qualifying features.

### 1.5 Review of evidence

In order to address the key biological questions a comprehensive review of the available evidence was needed. This was designed to build upon the existing evidence base in Furness et al. (2013), who reviewed the available evidence up to 2012 for potential compensation methods for key species in England in relation to offshore wind farm predicted impacts.

### 1.6 Population level impacts

In order to assess the likelihood of success of potential compensation measures, the likely population level impacts will need to be known. Using Population Viability Analyses (PVA) the likely level of compensation for a given level of impact can be explored. Here the Seabird PVA Tool[1] was used to provide projections of population size over time and counterfactual metrics of end population size and population growth rate. These were used to explore the efficacy of potential compensation scenarios and their likelihood of success.

Three levels of impact were considered and provided by Natural England. A low impact scenario was based on 1% change in baseline mortality for the species being assessed. A medium impact scenario considered the potential impact arising from an additional 7 GigaWatts (GW), this being the anticipated additional capacity of Offshore Wind Leasing Round 4 in English waters. Potential impacts at this level were estimated on a pro-rata basis from the current (at the time of writing) in-combination impacts on each SPA, which was equivalent to approximately 26 GigaWatts (GW) of capacity that was operational, consented and in-planning. Finally, the high impact scenario was

[1] http://ec2-54-229-75-12.eu-west-1.compute.amazonaws.com/shiny/seabirds/PVATool/R/

also based on the predicted impact associated with the current 26GW but increased pro-rata to reach 100 GW of installed capacity by 2050 i.e. an additional 74GW.

For each SPA population being assessed three compensation scenarios were derived (low, medium and high) based on species specific information for each compensation measure. Thus, for each SPA population there were nine possible combinations of impact level against compensation level.

The purpose of these assessments was to explore the potential limitations of compensation to address a range of impacts when the potential scale of the impact from future OWF expansion plans are considered. It is important to note that these assessments are indicative only and were not intended to pre-judge any future plan-level HRA impact assessments.

## 1.7 Assessment of confidence in findings

Assessment of confidence in the potential compensation measures assessed using PVA was undertaken using a modified form of the IPCC guidance on the consistent treatment of uncertainty using calibrated language (Mastrandrea et al. 2010). These authors recommended using the interaction between Evidence and Agreement to assess confidence in assessment (Table 2).

**Table 2 The interaction of evidence and agreement statements, and their relationship to confidence. Confidence increases to the top right corner as suggested by the increase strength of shading. From Mastrandrea et al. 2010.**

| Agreement ↑ | | | | Confidence ↑ |
|---|---|---|---|---|
| | High agreement<br>Limited evidence | High agreement<br>Medium evidence | High agreement<br>Robust evidence | |
| | Medium agreement<br>Limited evidence | Medium agreement<br>Medium evidence | Medium agreement<br>Robust evidence | |
| | Low agreement<br>Limited evidence | Low agreement<br>Medium evidence | Low agreement<br>Robust evidence | |
| | Evidence (type, amount, quality, consistency → | | | |

Using the advice from the IPCC, we created a structured approach to the qualitative assessment and recording of data and model confidence across a range of important data criteria: type, amount, quality, consistency and agreement (Table 3). We have recommended criteria for each data value within each data dimension that the IPCC recommends.

**Table 3 Criteria for assessing the value of different data dimensions (adapted from Mastrandrea et al. 2010).**

| Dimensions | Criteria | Value |
|---|---|---|
| Type of evidence | Qualitative data | Limited |
| | Semi-quantitative data / expert judgement | Medium |
| | Quantitative data | Robust |

| Dimensions | Criteria | Value |
|---|---|---|
| Amount of evidence | Small sample size | Limited |
| | Medium sample size | Medium |
| | Large sample size | Robust |
| Quality of evidence | Apply expert opinion and record reasoning | Limited |
| | | Medium |
| | | Robust |
| Consistency of evidence | Few studies agree | Limited |
| | Most studies agree | Medium |
| | All studies agree | Robust |
| Applicability of evidence | Evidence obtained from a site far from the site being studied, or from a site known to be ecologically different to the site being studied. | Low |
| | Evidence obtained from a site nearby or with high similarity to the site being studied | Medium |
| | Evidence obtained from the site being studied | High |

For each empirical metric used to recommend suitable compensation methods (e.g. evidence of the effects of fisheries on seabird survival and/or productivity) and the analytical method used to assess the efficacy of those recommended methods (e.g. PVA) an assessment was made of evidence type, amount, quality and consistency. Rather than using agreement as a term in the confidence matrix, we have applied the applicability of the data to the specific use as an alternative approach. The interaction of evidence and applicability were used to assess confidence in each metric using the matrix in Table 4.

**Table 4 Assessment of confidence using assessments of evidence and applicability.**

| | | Evidence | | |
|---|---|---|---|---|
| | | Limited | Medium | Robust |
| Applicability | High | MEDIUM | HIGH | VERY HIGH |
| | Medium | LOW | MEDIUM | HIGH |
| | Low | VERY LOW | LOW | MEDIUM |

For each metric a narrative explaining the reasoning behind the decisions made for the scoring of evidence and applicability was constructed. This narrative is then expanded to collate the confidence across the relevant metrics to provide an overall confidence assessment of the compensation method, or the assessment of its efficacy. It is then necessary to use this information to explain and justify the overall reasoning for a final confidence assessment in the whole process of assessment. The narrative can be used to explain a decision to change the overall confidence assessment, either to increase or decrease that final confidence assessment. This is necessary when there are many factors that need to be taken into account in making a final recommendation and the structured approach to assessment of confidence was used to provide a transparent approach to aid communication and decision making. Throughout this report changes in the scores as a result of the considerations in the narrative tables are highlighted using bold text.

Finally, the confidence in the evidence for the compensation approach and the assessment method need to be carefully considered. The confidence assessed so far does not address the combined scale of the predicted impact and the scale of the predicted compensation. Since this assessment is addressed using multiple scenarios, it may be that the confidence in the approach and the assessment method are high but the level of compensation being assessed is insufficient to overcome the level of impact being imposed. The confidence in the approach and assessment needs to be considered alongside the predicted population level effects. The confidence is then used to decide on the efficacy of the proposed compensation mostly when the decision is closer to the margins of acceptability. It can also be used to strengthen the decision, e.g. when the low compensation level is sufficient to overcome a high impact scenario and the confidence is High, then there can be a stronger conclusion drawn, or when the high compensation level is insufficient to overcome the low impact scenario and the confidence is low.

### 1.8 Future monitoring and adaptive management

For each SPA qualifying feature, recommended monitoring and adaptive management were considered. These were related specifically to the nature of the threat to the populations being assessed and the type of compensation being suggested. It will be important that monitoring of populations is targeted at the specific demographic parameter or parameters expected to be impacted and the related type of compensation. In all cases the overall aim is to maintain the coherence of the UK SPA network. The primary aim for compensation to all SPA qualifying features assessed here, except red-throated diver, was to recover, maintain, or increase their population sizes. For red-throated diver the predicted impacts negatively affect the distribution of the species within the site, so the aim of compensation was to improve the distribution of the species within the site.

# 2 FLAMBOROUGH AND FILEY COAST SPA – BREEDING BLACK-LEGGED KITTIWAKE

The Flamborough and Filey Coast SPA (FFC SPA) is on the east Yorkshire coast of the North Sea. The site is in two sections with a gap in the middle of Filey Bay. The Flamborough section is south of this gap and the Filey section is north. The habitats within the site include clifftop, sea cliff and intertidal rock habitats, and the coastal sea out to two km from the coast.

The site was designated due to its nationally and internationally important breeding seabird colony, currently the largest mainland seabird colony in England. The SPA includes the only mainland breeding gannet colony in England, the largest kittiwake colony in the UK and the largest guillemot and razorbill colonies in England. The whole site supports more than 200,000 seabirds during the breeding season.

The sea adjacent to the SPA is used by the breeding seabirds for a range of activities, including bathing, preening, displaying, loafing and local foraging. Offshore of the SPA the oceanographic frontal system known as the 'Flamborough Front' results in nutrient-rich waters and contributes to sustaining many of the qualifying features of the site.

## 2.1 Conservation status of kittiwake

The biogeographic population (North Atlantic, i.e. subspecies *R. t. tridactyla*) was estimated at 2,750,000 pairs, of which 370,000 pairs breed in Great Britain and 49,000 pairs in all-Ireland (Mitchell et al. 2004). The IUCN lists the global population of kittiwake as "vulnerable" as the population has declined rapidly over more than three generations and is predicted to continue declining.

The UK population was red listed in Birds of Conservation Concern (BOCC) 4 (Eaton et al. 2015) having been previously listed as 'amber' in BOCC3. National surveys found an increase in breeding numbers in the UK of 25% between 1969 and 1986 which followed many decades of population growth through the early 20$^{th}$ Century, but a decrease of 25% between 1986 and 2000 (JNCC 2020). JNCC SMP data show a long-term decline in the population index for Scotland from 100 in 1986 to 31 in 2018 (JNCC 2020). In England, the population index has also declined, but much less than in Scotland, reaching 68 in 2018 (JNCC 2020). Trends in Wales are similar to those in England.

It is protected under the Birds Directive as a migratory species. The SPAs in Great Britain were estimated to hold 56.5% of the Great Britain breeding population of kittiwakes present in 2000 (Stroud et al. 2016), a decrease from the 78% in SPAs estimated in the previous SPA review (Stroud et al. 2001). One site in Northern Ireland also qualifies (Rathlin Island).

Data collated by Natural England (Natural England 2020) indicate that the kittiwake population in the Flamborough Head and Bempton Cliffs SPA increased between the Operation Seafarer national census in 1969 and the Seabird Colony Register (SCR) national census in 1987 (Figure 1). The population declined between 1987 and the Seabird 2000 count in 2000 and has not changed much since then.

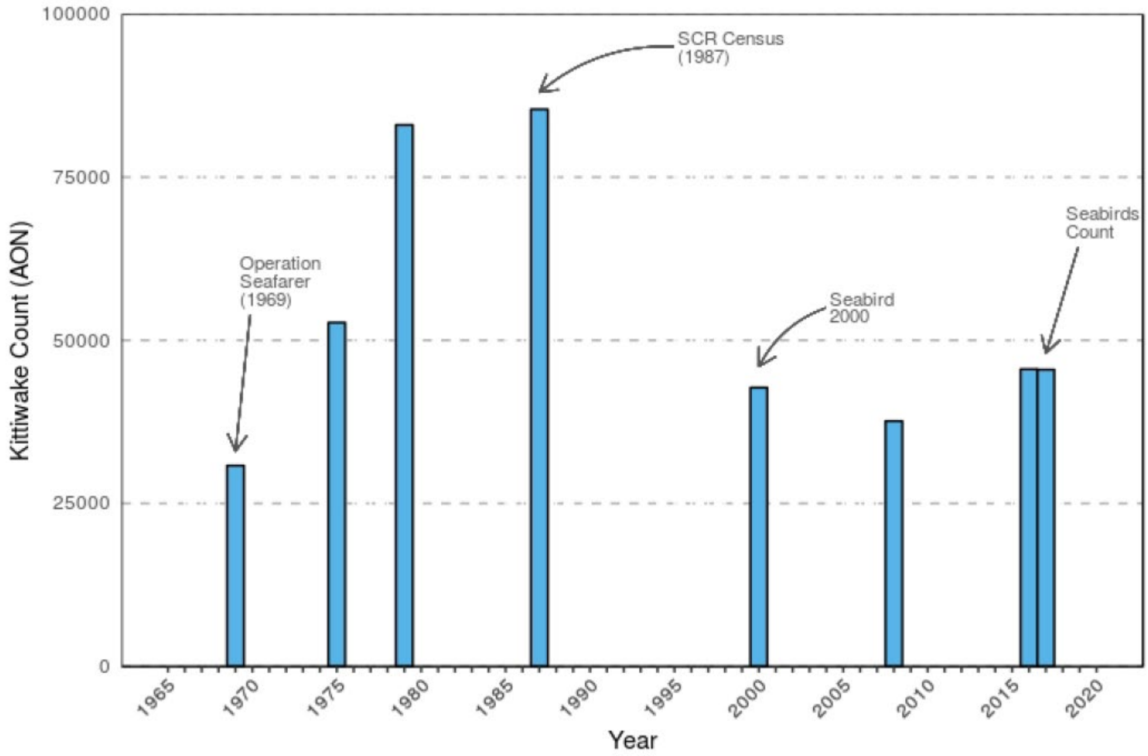


**Figure 1 Whole colony counts of kittiwake (Apparently Occupied Nests - AON) recorded at Flamborough Head and Bempton Cliffs 1969-2017 with national census counts indicated (Natural England 2020).**

There are 29 sites in Scotland with breeding kittiwake listed in the citation as a SPA feature, either as a main component of the breeding seabird assemblage, or as a breeding feature itself. There are two in England (Farne Islands and FFC), one in Wales, and one in Northern Ireland. In the most recent assessment of site condition, the conservation status of the breeding kittiwake feature at four sites in Scotland was classified as Favourable Maintained but was classified as Unfavourable at 25 sites. Declines are especially large in Shetland and Orkney, where SPA populations have fallen by 90% since designation. Overall, the Natura suite for breeding kittiwake should be considered at present to be in Unfavourable conservation status. Breeding numbers at FFC SPA was 44,520 pairs at the recent re-classification and 51,535 pairs in 2017 (Aitken et al. 2017), an increase of 16%. This is still lower than the 85,395 breeding pairs from 1987 (Natural England 2020). This makes FFC SPA by far the largest kittiwake colony in the British Isles, and an increasingly important proportion of the entire kittiwake breeding population in the British Isles. In 1969, FFC SPA held about 30,800 pairs of kittiwakes from a population of 470,388 pairs in Britain and Ireland (Cramp et al. 1974), or 6.5% of the total. In 2000, FFC SPA held about 42,582 pairs of kittiwakes from a population of 415,995 pairs in Britain and Ireland (Mitchell et al. 2004), or 10% of the total. Now, FFC SPA holds about 51,535 pairs from a total in Britain and Ireland that is probably around 200,000 to 250,000 pairs based on the observed rate of decline and most recent colony counts (JNCC 2020), so FFC SPA now holds probably over 20% of the current total in Britain and Ireland.

### 2.2 Citation population size

The population citation is 44,520 pairs (2008-2011), according to the Natura 2000 Standard Data Form.

### 2.3 Conservation objectives

The site has conservation objectives, "to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site."

More detailed Supplementary Advice on Conservation Objectives (SACO) have since been added online, last updated 13 March 2020 (Natural England 2020). For kittiwake at FFC SPA these are:

- Restore the size of the breeding population at a level which is above 83,700 breeding pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Restore safe passage of birds moving between nesting and feeding areas;
- Restrict the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Restore the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding) at existing level;
- Restore the distribution, abundance and availability of key food and prey items (e.g. Sandeel, sprat, cod, squid, shrimps) at preferred sizes;
- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;

- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;
- Maintain water quality and specifically mean winter dissolved inorganic nitrogen (DIN) at a concentration equating to High Ecological Status (specifically mean winter DIN is < 12 µM for coastal waters), avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for FFC pSPA was published in February 2015 (Natural England 2015). That identified public access/disturbance as a threat to kittiwakes and identified prevention of disturbance as a responsibility of East Riding of Yorkshire Council, Natural England, RSPB, Scarborough Borough Council, Yorkshire Wildlife Trust, and Flamborough Management Scheme. The SIP also identified the decline in kittiwake abundance as requiring further research and identified investigation of the causes of decline as a responsibility of Natural England, North Eastern Inshore Fisheries Conservation Authority (IFCA), RSPB, Yorkshire Wildlife Trust, and Flamborough Management Scheme.

### 2.3.1 How each Conservation Objective (CO) might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are three main sources of impact on kittiwakes from offshore wind farm development: mortality due to collisions with operational turbines, displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of kittiwakes from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. That impact on survival may be a carry-over effect on reduced winter survival as birds are in poorer condition at the end of the breeding season than would have been the case in the absence of the wind farm. There is a known relationship between the condition (body mass) of kittiwakes at the end of the breeding season and their subsequent overwinter survival (see Oro and Furness 2002, Searle et al. 2017 and references therein).

The maintenance of the population of each of the qualifying features could be affected directly through collision mortality and indirectly through impact to energy budgets from displacement and barrier effects.

## 2.4 Location of compensation

Cook et al. (2011) defined six Ecological Assessment Areas (EAAs) for kittiwake (Figure 2). The FFC SPA occurs within EAA 3. Consequently, the hierarchy of the locations of compensation are:

1. FFC SPA;
2. EAA 3; and
3. All other EAAs in the UK.

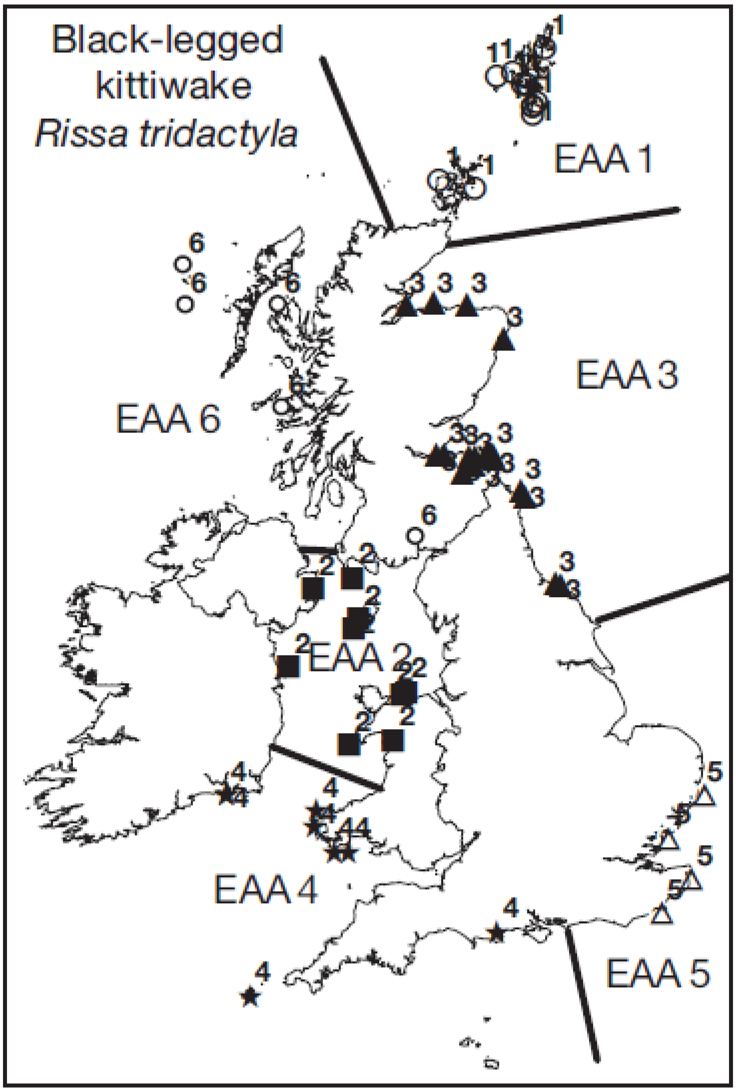


**Figure 2 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for kittiwake by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

## 2.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding kittiwakes were developed based on the seven potential compensation measures reviewed by Furness et al. (2013). Seven potential measures were listed:

1. Closure of sandeel and sprat fisheries in UK waters;
2. Provision of artificial structures for new kittiwake colonies;
3. Mink eradication;
4. Feral cat eradication;
5. Rat eradication;
6. Fencing out foxes from colonies; and
7. Exclusion of great skuas.

For each of these potential compensation measures a series of key biological questions directly related to the compensation of impacts on the FFC SPA breeding kittiwake population was identified. These help to clarify the nature of the evidence required to inform an assessment of the likelihood that any given compensatory measure may succeed in offsetting predicted impacts from offshore windfarms.

### 2.5.1 Closure of sandeel and sprat fisheries in UK waters

The key biological questions that would need to be answered in relation to closure of sandeel and sprat fisheries as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 5.

**Table 5 Key Biological Questions in assessing the potential for compensation through closure of sandeel and sprat fisheries.**

| No. | Key Biological question |
|---|---|
| 1 | Are sandeels important to kittiwake populations? |
| 1.1 | At FFC SPA? |
| 1.2 | At colonies in EAA 3? |
| 1.3 | At colonies in all other EAAs in the UK? |
| 2 | Do differences in the sandeel stock affect kittiwake demographics? |
| 2.1 | At FFC SPA? |
| 2.2 | At colonies in EAA 3? |
| 2.3 | At colonies in all other EAAs in the UK? |
| 3 | Would changing the sandeel stock change kittiwake populations? |
| 3.1 | At FFC SPA? |
| 3.2 | At colonies in EAA 3? |
| 3.3 | At colonies in all other EAAs in the UK? |
| 4 | Would closing UK waters to sandeel fisheries increase sandeel stocks? |
| 4.1 | At FFC SPA? |
| 4.2 | At colonies in EAA 3? |

| No. | Key Biological question |
|---|---|
| 4.3 | At colonies in all other EAAs in the UK? |

### 2.5.2 Provision of artificial structures for new kittiwake colonies

The key biological questions that would need to be answered in relation to the provision of artificial structures for new kittiwake colonies as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 6.

**Table 6 Key Biological Questions in assessing the potential for compensation through provision of artificial structures.**

| No. | Key Biological question |
|---|---|
| 1 | Do kittiwakes nest on artificial structures? |
| 1.1 | At FFC SPA? |
| 1.2 | In EAA 3? |
| 1.3 | In all other EAAs in the UK? |
| 2 | What is the productivity of breeding kittiwakes on artificial structures? |
| 2.1 | At FFC SPA? |
| 2.2 | In EAA 3? |
| 2.3 | In all other EAAs in the UK? |
| 3 | Would artificial colonies result in increased immigration to SPA colonies? |
| 3.1 | At FFC SPA? |
| 3.2 | In EAA 3? |
| 3.3 | In all other EAAs in the UK? |
| 4 | How large would the productivity of the artificial colonies need to be to result in a net export of fledglings available to immigrate into other colonies? |
| 4.1 | At FFC SPA? |
| 4.2 | In EAA 3? |
| 4.3 | In all other EAAs in the UK? |
| 5 | How many breeding seasons will it be before new artificial colonies are large enough to result in a net export of fledglings? |
| 5.1 | At FFC SPA? |
| 5.2 | In EAA 3? |
| 5.3 | In all other EAAs in the UK? |
| 6 | What are the potential effects of climate change on the ability of new artificial colonies to produce a net export of fledglings? |
| 6.1 | At FFC SPA? |
| 6.2 | In EAA 3? |
| 6.3 | In all other EAAs in the UK? |

### 2.5.3 Mink eradication

The key biological questions that would need to be answered in relation to mink eradication as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 7.

**Table 7 Key Biological Questions in assessing the potential for compensation through mink eradication.**

| No. | Key Biological question |
|---|---|
| 7 | Is there evidence that American mink (*Neovison vison*) predation occurs on kittiwake colonies in the UK? |
| 7.1 | At FFC SPA? |
| 7.2 | In EAA 3? |
| 7.3 | In all other EAAs in the UK? |

### 2.5.4 Feral cat eradication

The key biological questions that would need to be answered in relation to feral cat eradication as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 8.

**Table 8 Key Biological Questions in assessing the potential for compensation through feral cat eradication.**

| No. | Key Biological question |
|---|---|
| 8 | Is there evidence that feral cat predation occurs on kittiwake colonies in the UK? |
| 8.1 | At FFC SPA? |
| 8.2 | In EAA 3? |
| 8.3 | In all other EAAs in the UK? |

### 2.5.5 Rat eradication

The key biological questions that would need to be answered in relation to rat eradication as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 9.

**Table 9 Key Biological Questions in assessing the potential for compensation through rat eradication.**

| No. | Key Biological question |
|---|---|
| 9 | Is there evidence that rat predation occurs on kittiwake colonies in the UK? |
| 9.1 | At FFC SPA? |
| 9.2 | In EAA 3? |
| 9.3 | In all other EAAs in the UK? |

### 2.5.6 Fencing out foxes from colonies

The key biological questions that would need to be answered in relation to fencing out foxes from colonies as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 10.

**Table 10 Key Biological Questions in assessing the potential for compensation through fencing out foxes from colonies.**

| No. | Key Biological question |
|---|---|
| 10 | Is there evidence that fox predation occurs on kittiwake colonies in the UK? |
| 10.1 | At FFC SPA? |
| 10.2 | In EAA 3? |
| 10.3 | In all other EAAs in the UK? |

### 2.5.7 Exclusion of great skuas

The key biological questions that would need to be answered in relation to exclusion of great skuas from colonies as a compensation measure for impacts on the breeding kittiwake population at FFC SPA are shown in Table 11.

**Table 11 Key Biological Questions in assessing the potential for compensation through exclusion of great skuas from colonies.**

| No. | Key Biological question |
|---|---|
| 11 | Is there evidence that great skua predation occurs on kittiwake colonies in the UK? |
| 11.1 | At FFC SPA? |
| 11.2 | In EAA 3? |
| 11.3 | In all other EAAs in the UK? |
| 12 | Have great skuas reduced the size of kittiwake colonies |
| 12.1 | At FFC SPA? |
| 12.2 | In EAA 3? |
| 12.3 | In all other EAAs in the UK? |

## 2.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 2.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

In the case of kittiwakes, the additional evidence base for the potential for predator control measures as compensation was small, so these measures have been amalgamated into a single section (2.6.3).

### 2.6.1 Closure of sandeel and sprat fisheries in UK waters - new evidence

Furness et al. (2013) considered that the closure of the sandeel and sprat fisheries was highly likely to be effective with sufficient evidence to have a high confidence in that assessment. In this assessment only closure of the sandeel fishery has been considered as this has the strongest evidence base and most relevance to the FFC SPA.

Breeding kittiwakes at most colonies around the North Sea feed mainly on sandeels (Furness and Tasker 2000, Coulson 2011). The breeding success of kittiwakes has been shown to be strongly influenced by the abundance of sandeels (Frederiksen et al. 2004, Cury et al. 2011, Carroll et al. 2017, Christensen-Dalsgaard et al. 2018). Breeding success has also been shown to directly affect the population growth of colonies (Monnat et al. 1990, Cadiou et al. 1994, Coulson 2011, 2017). For instance, breeding success and colony size of kittiwakes in Shetland decreased substantially after the Shetland sandeel stock collapsed (Furness and Tasker 2000). Breeding success of birds nesting on Foula showed a strong relationship with the Shetland sandeel total stock biomass (Figure 3). Kittiwake breeding success was lower in most years of sandeel biomass below 40,000 tonnes but was higher in almost all years when sandeel biomass was above that level.

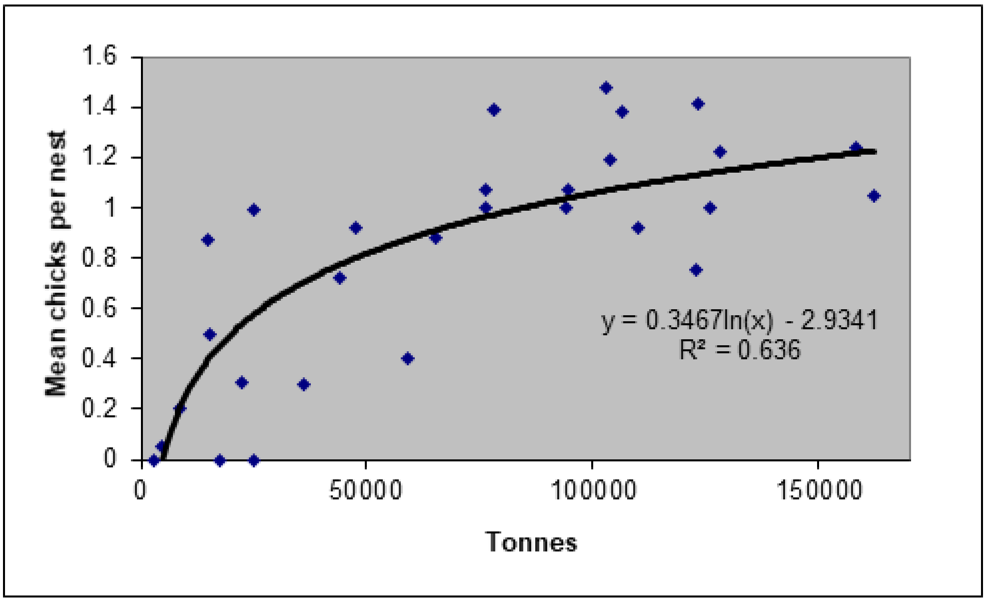


**Figure 3 Breeding success of black-legged kittiwake at Foula, Shetland, in relation to the Shetland sandeel total stock biomass for the years 1976 to 2004.**

The breeding success of kittiwakes on the Isle of May was also affected when the sandeel stock on the Wee Bankie was heavily fished (Frederiksen et al. 2004). This stock is distinct from those around Shetland or the southern North Sea (Frederiksen et al. 2005, ICES 2017, Olin et al. 2020). Over recent decades, sandeels (specifically *Ammodytes marinus*) were the target of what was the largest single-species fishery in the North Sea. That fishery concentrated on the Dogger Bank sandeel stock. Kittiwakes at FFC SPA forage over a large area including some of the most important sandbanks on the Dogger Bank. These areas support high densities of sandeels and the sandeel fishery (Carroll et al. 2017). It has been shown that the fishery has depleted the biomass of sandeels in this region (Lindegren et al. 2018), and that this has resulted in reduced productivity of kittiwakes at FFC SPA (Carroll et al. 2017). Reducing the fishing effort or closing the fishery in waters with connectivity with the colony could provide a compensation mechanism to improve breeding success of kittiwakes at FFC SPA. This could allow the recovery of the sandeel stock from the long-term high fishing mortality it has been experiencing. It has been predicted that a reduction in fishing mortality would result in a rapid, though likely incomplete, recovery of sandeel abundance (Lindegren et al. 2018). Sandeels are short-lived fish which only breed when one or two years old, with high reproductive potential. While kittiwakes feed on all age classes of sandeels, they particularly feed on one- and two-year-old fish. An increase in sandeel abundance would probably increase kittiwake productivity with a relatively short time lag of one or two years.

Success of kittiwakes at the Isle of May was on average 0.5 chicks per pair lower during years when sandeel fishing occurred in the area than it was in years with no sandeel fishing (Frederiksen et al. 2004; Figure 4).

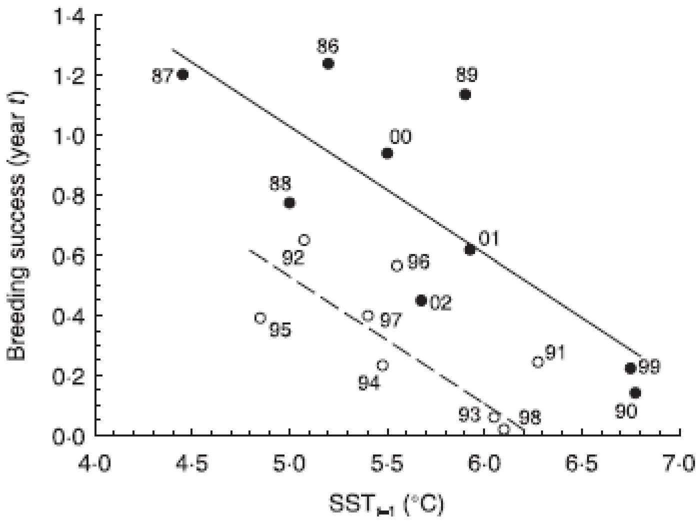


**Figure 4 Kittiwake breeding success at the Isle of May in relation to local Sea Surface Temperature in February-March of the previous year, and the presence (open circles and dashed line) or absence (black dots and solid line) of a sandeel fishery off east Scotland (From Frederiksen et al. (2004)).**

Due to the ongoing poor productivity of kittiwakes on the Isle of May (part of the Forth Islands SPA) an area off the east of Scotland was closed to sandeel fishing (the 'sandeel box'). Closure of the fishery resulted in an increase in sandeel stock biomass (Greenstreet et al. 2010) and an increase in kittiwake breeding success at colonies within the closed area compared to those outside (Daunt et al. 2008, Frederiksen et al. 2008). Thus, there is strong experimental evidence for the effect of closing the fishery. Since then, the sandeel fishing industry has lobbied for the reopening of the fishery within the box, but the regulator has maintained the closure. Fishing for sandeels has continued on the eastern edge of the closed area.

This sandeel fishery closure has affected the age structure of the sandeel population within the stock. When heavily fished, very few sandeels in the stock were older than two years. This meant there was a high variability of stock abundance between years because of the high variability of young fish production. After the fishery was closed, sandeels lived longer, with large cohorts remaining in the stock for up to six years (Peter Wright, pers. comm.). This longer life expectancy increased the biomass of the stock and reduced variability in fish abundance, which had been driven by variable recruitment. This improved kittiwake breeding success, as even in years of poor sandeel recruitment, the biomass of the stock was buffered by the older age classes of fish.

The abundance of sandeels in ICES Area 4 declined from 1993 to 2001 (Figure 5). This includes the sandeel no-take box off the east coast of Scotland. However, after the closure of the sandeel fishery in the box, this stock recovered (Figure 6).

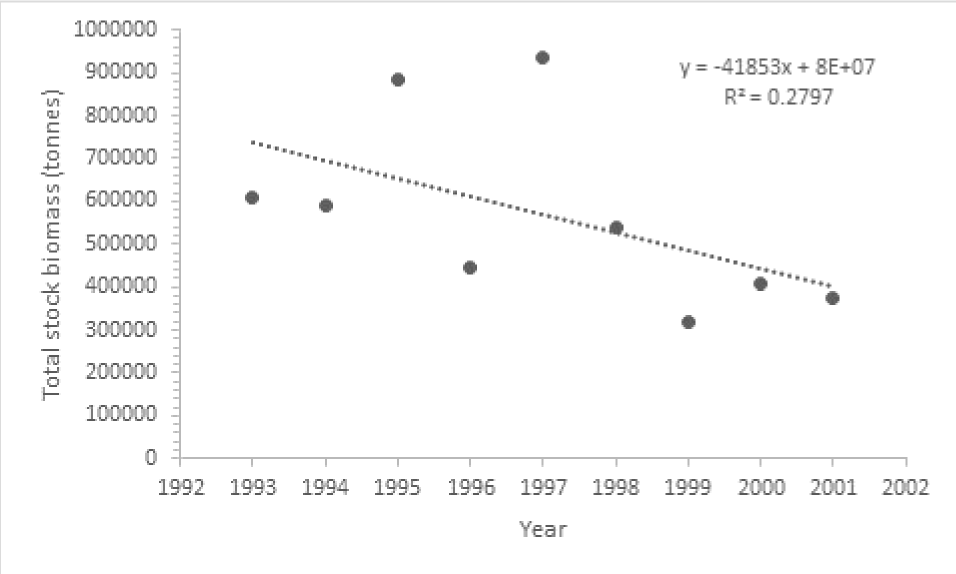


**Figure 5 Abundance (total stock biomass in tonnes) of sandeels in ICES area 4 from 1993 to 2001. Data from ICES (2020).**

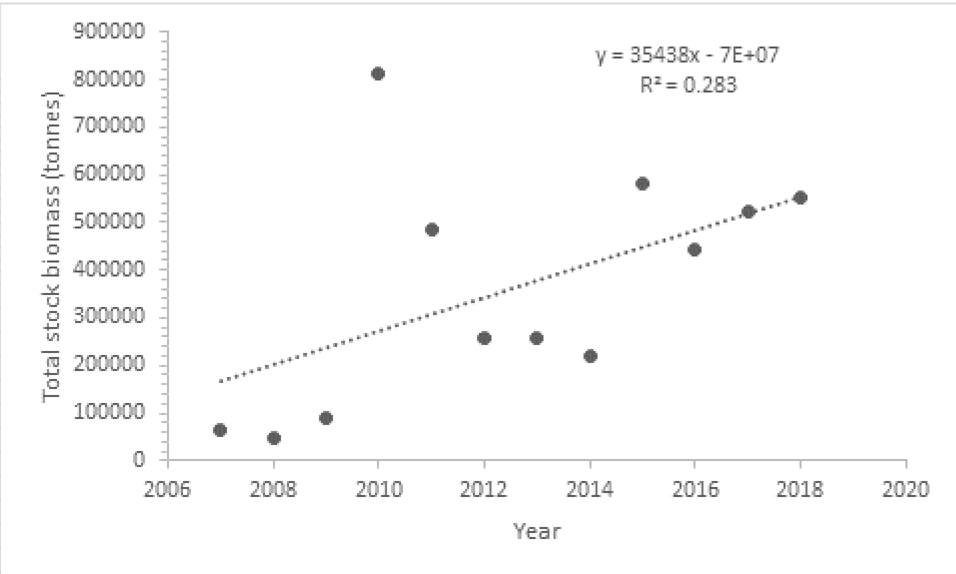


**Figure 6 Abundance (total stock biomass in tonnes) of sandeels in ICES area 4 from 2007 to 2018. Data from ICES (2020).**

The establishment of the sandeel box greatly reduced catches from ICES Area 4, but fishing continues in Area 4 outside the box. The depleted stock biomass in the remaining parts of Area 4 meant that the commercial catch was low between 2005 and 2012. This meant it was no longer commercially profitable to continue fishing in Area 4 and better catches could still be taken elsewhere, particularly Area 1r – the Dogger Bank. Commercial catches in Area 4 have increased considerably in recent years, as the stock has begun to recover (Figure 7). Therefore, the potential threat to kittiwake breeding success on the east coast of Scotland has returned, highlighting the importance of managing the fishery beyond closure areas.

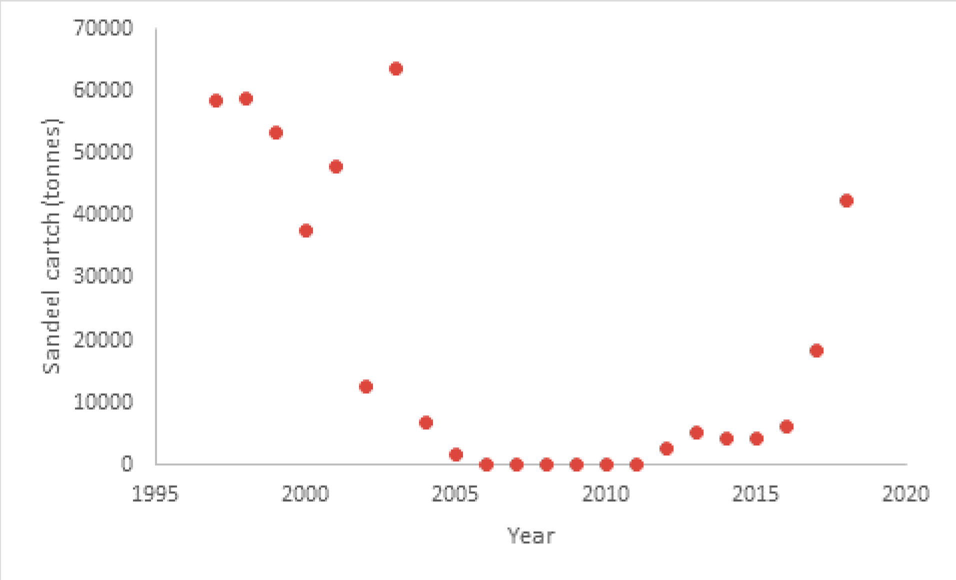


**Figure 7 Catch (tonnes) of sandeel by the commercial sandeel fishery in ICES area 4 from 1997 to 2018 (data from ICES 2020).**

Depletion of sandeel stocks not only affects breeding success of kittiwake, as demonstrated at Shetland, east Scotland and FFC SPA, but can also affect adult survival of kittiwakes if depletion of sandeel abundance is severe. Oro and Furness (2002) showed that kittiwake survival at Foula, Shetland, was related to sandeel abundance. Frederiksen et al. (2004) showed the same at the Isle of May. More recent data from the Isle of May also suggest this relationship. Return rates of breeding adult kittiwakes (these are the proportions of colour ringed adults in year x seen again in year x+1, so are not corrected for absences of individuals that subsequently return and so underestimate true survival slightly) correlate strongly with sandeel total stock biomass in ICES area 4 between 2004 and 2018, the latest year for which data are published (Figure 8). This figure has to be interpreted with caution, as data are autocorrelated, with poor years for kittiwake return rates occurring during a period of years with low sandeel stock biomass. Other drivers may be involved, and no causal relationship can be determined directly from this graph, but the strong correlation suggests that adult survival in this population has been subject to adverse effects from the depletion of the sandeel stock in the region and is consistent with evidence from Shetland. Unfortunately, there are no long-term data on kittiwake survival at FFC SPA, although RSPB have recently started colour ringing there, so such evidence should become available in future.

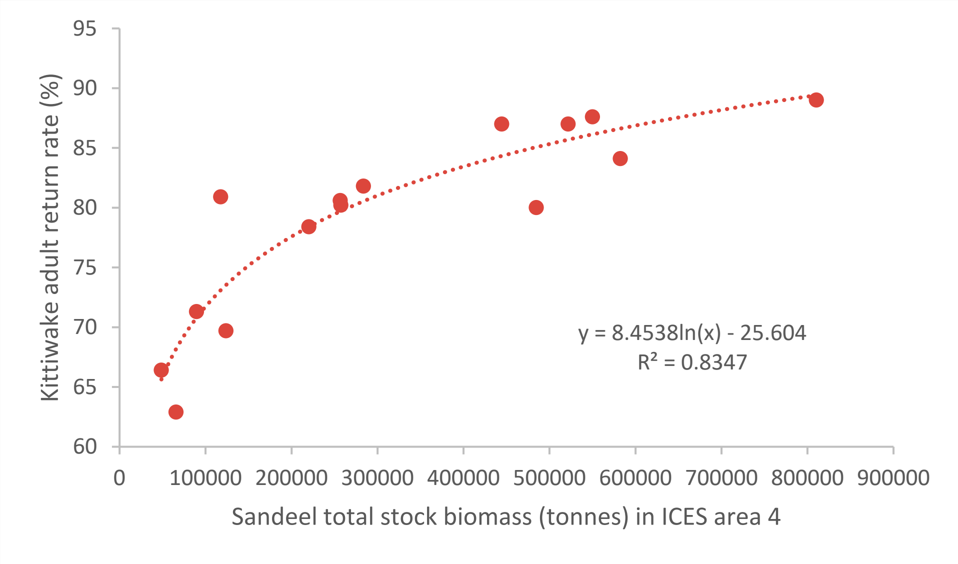


**Figure 8 Annual return rates of adult kittiwakes at the Isle of May (UKCEH data published in online annual reports on seabird monitoring) in relation to ICES estimates of annual sandeel total stock biomass (tonnes) in area 4 between 2004 and 2018. Data for 2019 and 2020 are not yet published.**

The kittiwake population at FFC SPA is currently above the citation population size, but well below the target population size outlined in the SACO. The productivity of kittiwakes in the SPA is currently below the minimum suggested productivity for a stable population of 0.8 chicks per pair (Coulson 2011). Carroll et al. (2017) found a strong relationship between kittiwake productivity at FFC SPA and the sandeel stock biomass in Area 1r, which includes the main foraging areas for breeding adult birds from the colony. This is similar to the findings from other studies (Furness and Tasker 2000, Oro and Furness 2002, Furness 2007) that kittiwake adult survival and productivity was correlated to sandeel stock biomass in Shetland. Frederiksen et al. (2004) also found this to be the case for kittiwakes breeding on the Isle of May, which was also affected by sea surface temperature.

Breeding success of kittiwakes around the Flamborough and Bempton cliffs has declined in recent years. It fell from an average of 1.2 chicks per pair in 1999-2001 to an average of just over 0.8 chicks per pair in 2006-2011, to around 0.5 chicks per pair in 2016-2019 (RSPB Annual Reports). This decline coincides with decline in sandeel abundance: total stock biomass in ICES area 1r (which includes the foraging grounds of kittiwakes breeding at FFC SPA) fell from an average of 995,624 tonnes in 1984-2002 to an average of 574,771 tonnes in 2003-2012 and fell further to an average of 460,023 tonnes in 2013-2018 (ICES 2020). The only year since 2000 in which breeding success exceeded 1 chick per pair (2010) was also the one and only year with anomalously high sandeels stock biomass (1.6 million tonnes) due to one year of exceptionally high recruitment (see Figure 7). However, that was a short-lived peak of sandeel abundance (Figure 7), in part because higher catches were taken from the stock by the fishery in 2010-2011 (ICES 2020).

Productivity of kittiwakes in the North Sea appears to be dependent on the availability of sandeels rather than other species of fish. A colony in Norway was found to have higher productivity in years when chicks were fed predominantly sandeels compared with years when chicks were mostly fed other species (Christensen-Dalsgaard et al. 2018).

Fisheries research on the Dogger Bank has assessed the consequences of high fishing mortality on the sandeel stock (Lindegren et al. 2018). It was estimated that if the fishing mortality (F) had been maintained at F=0.4, the spawning stock biomass would have been double the size it is currently. However, the fishing mortality levels have been much higher than this: F=0.8 to 1.2 between 1999 and 2009. These finding support the conclusion that the sandeel fishery has had a detrimental effect on the abundance of sandeels and this in turn has negatively affected kittiwake productivity and hence population size.

Lindegren et al. (2018) also suggested that recovery of the stock may be inhibited following reduction in fishing pressure by the effects of sea surface temperature on copepod abundance, the main prey source for sandeels. Long term trends on sea surface temperature as a result of climate change may therefore be a hindrance to sandeel recovery. Further constraints on sandeel stock recovery can occur when the biomass of the stock is reduced to very low levels, as the effect of natural mortality becomes a more important constraint (Saraux et al. 2020). The Dogger Bank stock is still subject to high fishing mortality (F=0.6) and remains below its long-term average at 10% of its highest historical level and below, albeit only slightly, the limiting spawning stock biomass. At this level ICES should recommend closure of the fishery due to an increased risk of recruitment failure in the stock (ICES 2020).

Analyses of multiple seabird-fisheries interaction case studies globally by Cury et al. (2011) resulted in the recommendation that fish stock should be kept above one third of their historic maximum biomass to provide sufficient forage fish for breeding seabirds to maintain suitable levels of productivity. The sandeel stock in the southern North Sea is below this level. The historic maximum biomass of sandeels in the Dogger Bank area (ICES Area 1r) was about two million tonnes in the 1980s (Lindegren et al. 2018). Based on the recommendations of Cury et al. (2011) the necessary stock biomass to maintain the productivity of seabird populations dependent on this stock, such as the kittiwakes at FFC SPA, would be 666,667 tonnes. However, ICES data show that this only occurred in three of the last 16 years between 2003 and 2018 (ICES 2020) (Figure 9).

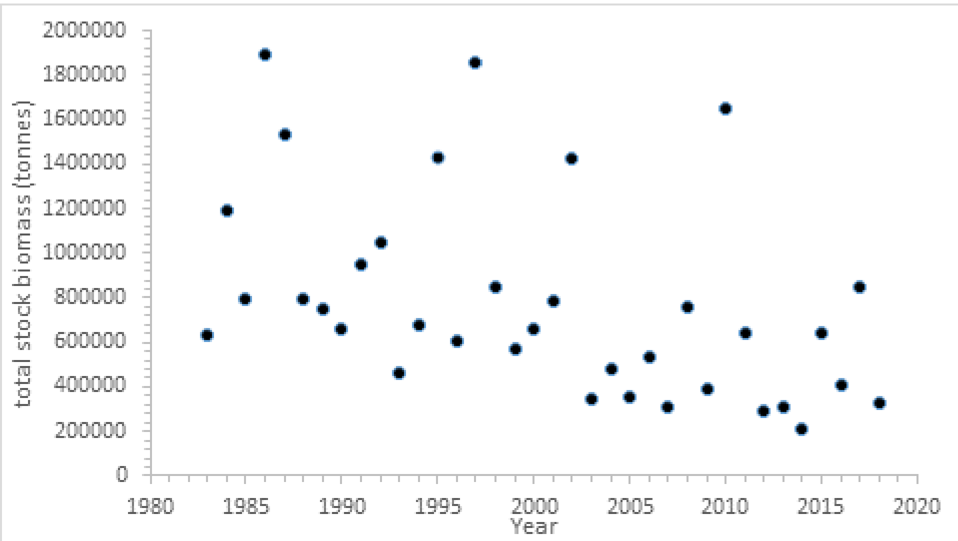


**Figure 9 Total stock biomass (tonnes) of sandeels in ICES area 1r (the Dogger Bank stock) between 1983 and 2018 (ICES 2020).**

The combined evidence described above demonstrates that the Dogger Bank sandeel fishery is limiting the productivity, and hence recovery, of the FFC SPA kittiwake population to the target population size. It is clear that the single most effective compensation measure for this population of kittiwakes, and likely other kittiwake populations on the North Sea coast of the UK, would be closure of the sandeel fishery (Carroll et al. 2017, Lindegren et al. 2018, Wright et al. 2018).

#### 2.6.1.1 *Answers to the key biological questions (2.5).*

The answers to the key biological question in relation to compensation through closure of sandeel and sprat fisheries are shown in Table 12.

**Table 12 Answers to Key Biological Questions in assessing the potential for compensation through closure of sandeel and sprat fisheries.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Are sandeels important to kittiwake populations? | |
| 1.1 | At FFC SPA? | Yes. Breeding success of kittiwakes is monitored every year by RSPB. The breeding success data can be split by area of the SPA and there is generally lower breeding success at Filey than elsewhere. The data from Filey cover a shorter run of years, and are from inconsistent numbers of sample plots, so are better left out of a long-term analysis (see Aitken et al. 2012, 2014, 2017, Babcock et al. 2015, 2016, 2018, Lloyd et al. 2019). Looking at the very extensive data from Flamborough Head and Bempton Cliffs, the productivity was higher earlier in the time series from 1986 than it has been recently, though with very high annual variation between 0.18 and 1.56 chicks per nest in different years (Figure 10). It is clear from Figure 10 that the breeding success is highly variable, and that it is necessary to consider the success in particular time periods, as breeding success was much higher in 1986-2001 than in 2002-2008 or in 2011-2019. The low breeding success in 2016-2019 coincides with ICES Area 1r sandeel stock falling to the lowest stock biomass reported over the past 40 years (ICES 2020). |
| 1.2 | At colonies in EAA 3? | Yes. Breeding kittiwakes at most colonies around the North Sea feed mainly on sandeels (Furness and Tasker 2000, Coulson 2011). |
| 1.3 | At colonies in all other EAAs in the UK? | Partly. Breeding success and colony size of kittiwakes in Shetland decreased substantially after the Shetland sandeel stock collapsed (Furness and Tasker 2000). Breeding success of birds nesting on Foula showed a strong relationship with the Shetland sandeel total stock biomass (Figure 3). These colonies are in EAA 1. There was no similar evidence from colonies in EAA 6, EAA2 and EAA 4, though there are no sandeel fisheries in the waters in these areas. |
| 2 | Do differences in the sandeel stock affect kittiwake demographics? | |
| 2.1 | At FFC SPA? | Yes. While there is no direct evidence from FFC SPA, the productivity of the kittiwake population has decline with time (Figure 9). During the same period total stock biomass in Area 1r has declined (Figure 9). The combined evidence described above demonstrates that the Dogger Bank sandeel fishery is limiting the productivity, and hence recovery, of the FFC SPA kittiwake population to the target population size. It is clear that the single most effective compensation measure for this population of kittiwakes, and likely other kittiwake populations on the North Sea coast of the UK, would be closure of the sandeel fishery (Carroll et al. 2017, Lindegren et al. 2018, Wright et al. 2018). |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2.2 | At colonies in EAA 3? | Yes. Evidence is presented above that sandeel stock declines negatively affects kittiwake productivity in Shetland and the Isle of May, which forage on two different sandeel stocks. Evidence is presented that the sandeel box increased sandeel abundance in Area 4, and kittiwake productivity increased on the Isle of May. |
| 2.3 | At colonies in all other EAAs in the UK? | Partly. There is only evidence for productivity being affected by sandeel stocks in EAA1. See the answers to 1.3 and 2.2. |
| 3 | Would changing the sandeel stock change kittiwake populations? | |
| 3.1 | At FFC SPA? | Yes. While there is no direct evidence for FFC SPA, the breeding success of kittiwakes has been shown to be strongly influenced by the abundance of sandeels (Frederiksen et al. 2004, Cury et al. 2011, Carroll et al. 2017, Christensen-Dalsgaard et al. 2018). |
| 3.2 | At colonies in EAA 3? | Yes. Direct evidence is presented above for increased population size on the Isle of May following fisheries closures in Area 1r. |
| 3.3 | At colonies in all other EAAs in the UK? | Partly. There is evidence, presented above, that kittiwake populations fell in Shetland following collapse of the sandeel fishery. However, the colonies in the other EAAs do not forage on sandeel stocks that are fished. |
| 4 | Would closing UK waters to sandeel fisheries increase sandeel stocks? | |
| 4.1 | At FFC SPA? | Yes. It has been shown that the fishery has depleted the biomass of sandeels in Area 1r (Lindegren et al. 2018). Reducing the fishing effort or closing the fishery in waters with connectivity with the colony, could provide a compensation mechanism to improve breeding success of kittiwakes at FFC SPA |
| 4.2 | At colonies in EAA 3? | Yes. Evidence has shown that closing the fishery in the sandeel box off the east coast of Scotland has increased the abundance of sandeels in the box. |
| 4.3 | At colonies in all other EAAs in the UK? | No. Collapse of the fishery in Shetland and the subsequent cessation of fishing for sandeel has not resulted in a recovery of the sandeel stock. It is hypothesised that the stock was sufficiently depleted that natural predation, particularly from recovering predatory fish populations, has resulted in a very slow recovery of the sandeel stock. |

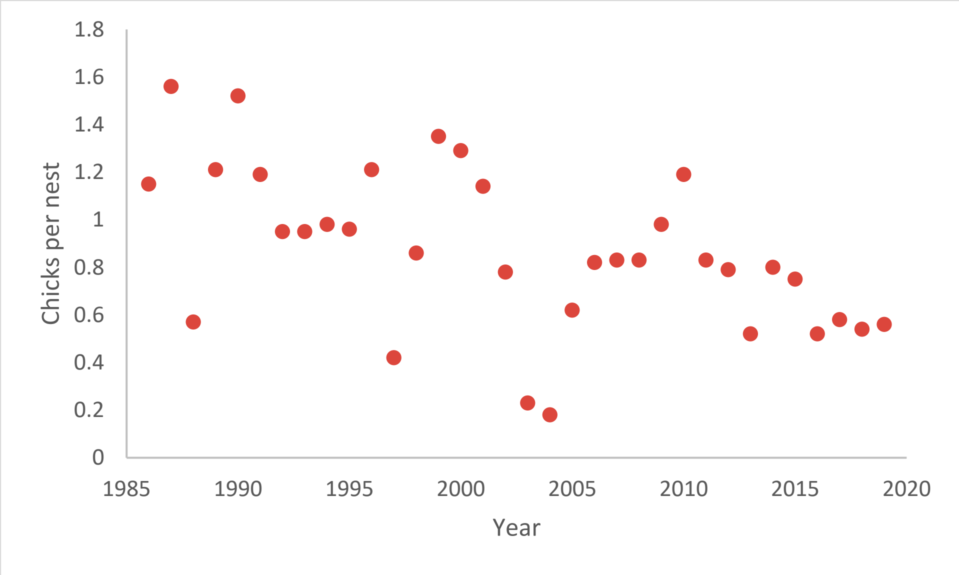


**Figure 10 RSPB data on breeding success of kittiwakes at Flamborough and Bempton from 1986 to 2019 (data from JNCC 2020 and from Aitken et al. 2012, 2014, 2017, Babcock et al. 2015, 2016, 2018, Lloyd et al. 2019).**

### 2.6.2 Provision of artificial structures for new kittiwake colonies - new evidence

Detailed reviews of artificial structures for breeding kittiwakes have recently been completed in relation to proposed compensation of kittiwakes by Ørsted (2020a, b, c, d, e, f). Kittiwakes are well known to nest on artificial structures, including buildings, such as the colony on the River Tyne in Newcastle which has been monitored over many decades.

Other structures used by nesting kittiwake include harbour walls, buildings, bridges, castles, churches, oil and gas platforms, power station water pipes, and purpose-made artificial colony sites. Breeding success on artificial structures can be at least as high as in natural colonies and can be higher where artificial sites are beyond the foraging range of large kittiwake colonies, close to food supplies and safe from predators (Christensen-Dalsgaard et al. 2019). Christensen-Dalsgaard et al. (2019) stated that, "increasing numbers of kittiwakes breeding on man-made structures both offshore and on the coast clearly provide a significant contribution of juveniles to the impoverished kittiwake population in Norwegian waters".

The Hornsea Three proposed constructing four new artificial colonies as compensation for predicted effects on the FFC SPA kittiwake population. Two structures have been proposed in Suffolk (Lowestoft to Sizewell) and two structures near the Tees Estuary, south of Seaham. Ørsted (2020b) stated that they may create "bespoke structure", or they may modify an existing structure (such as a building or seawall). They will also provide different structures when they are in the same area to "maximise the opportunity for kittiwake to colonise". Similarly, Norfolk Boreas have suggested a similar approach, should they be needed (Royal Haskoning DHV 2020).

While these approaches may be suitable for single projects, if compensation may be needed at a far larger scale using artificial colonies the *ad hoc* development of structures may result in a smaller overall level of compensation than needed to achieve the target population size of kittiwakes at FFC SPA. It is likely that further compensation will be needed for Round 4 developments, and for any potential future extension projects. Thus, a more strategic approach to development of kittiwake artificial colonies may be needed to deliver the level of compensation required. Here a range of potential impact and compensation scenarios were considered to encompass future development.

To provide successful compensation for FFC SPA any new artificial colonies will need to be colonised by kittiwakes and have high breeding success. Overall colony productivity will need to be about 0.6 chicks per nest to maintain the population at the new artificial site according to the simple population model applied here (see Section 2.7.1.2). So, in excess of 0.6 chicks per nest will be needed to provide potential compensation for offshore wind farm predicted impacts; any chicks above 0.6 per nest would be available to backfill losses that might be caused by offshore wind impacts. However, Coulson (2017) found that kittiwake colonies with productivity rates below 0.8 chicks per pair were declining, while those above 0.8 chicks per pair were increasing. The true value may be between 0.6 and 0.8 chicks per pair. A strategic approach to provide high-quality artificial colonies for kittiwakes could potentially perform much better than an *ad hoc*, project by project, approach. Since sustained high breeding success will be necessary for artificial colonies to provide adequate compensation at a strategic scale, the effects of protection from weather, from predators such as large gulls and mammals, and from disturbance by people, will need to be better known. In particular, it should be noted that some artificial structures created specifically for kittiwakes have not been successful. On the River Tyne, one of the "kittiwake towers" was never used by kittiwakes, so was eventually dismantled. Breeding success on the other kittiwake tower has been consistently lower than on some other structures in the area, such as the Tyne Bridge abutments.

Christensen-Dalsgaard et al. (2019) describe breeding by kittiwakes on six offshore oil platforms in Norwegian waters (five in the Norwegian Sea and one in the Barents Sea). The largest of these colonies was 674 nests on the oil platform Draugen, operated by OKEA, 75 km offshore. There were also 252 nests on Heidrun platform, operated by Equinor, 165 km offshore. Overall, they found over 1,200 pairs of kittiwakes nesting on these oil rigs in 2019 (exact numbers were not counted on two rigs so are not included in the total), and breeding success on the oil rigs was significantly higher than at coastal artificial colonies in the same part of the Norwegian coast (they list for comparison colony sizes and breeding success achieved at four artificial colonies on the Norwegian coast at fishing ports), and on average about four times higher than at natural colonies in the same part of Norway (they list for comparison colony sizes and breeding success at four neighbouring natural colonies). They suggest that the higher breeding success on oil rigs is likely to be due to higher food availability (the birds nesting offshore being at foraging grounds so not having to commute as far as birds that nest at the coast) and also to fewer predators at the oil rigs. They point out that predation on kittiwake nests on the oil rigs may not be zero. In particular, "kittiwakes breeding on the exposed parts of the rigs, had a lower productivity than those breeding on more sheltered parts of the rig". Christensen-Dalsgaard et al. (2019) suggest that this may be due to predation by large gulls, which are able to access nests that are in open areas but cannot access nests that are sheltered. However, the difference could potentially relate to exposure to rain and direct sunshine, which can also cause breeding failure of exposed nests.

It has been suggested that creation of artificial colonies for kittiwakes would only represent suitable compensation when carried out in regions where there is no available unoccupied natural habitat (as is the case along the east coast of England from FFC SPA to Kent). However, this is over-simplifying the situation. Where kittiwakes have large amounts of high-quality natural habitat with stable cliffs providing areas of narrow ledges, there could still be merit in providing some artificial colonies. Kittiwakes show strong competition for high quality nest sites (Coulson 2011, Acker et al. 2017) and there is clear evidence not only of density-dependent competition for nest sites at large kittiwake colonies (Acker et al. 2017) but also evidence of density-dependent competition for food in the waters around these large colonies (Wakefield et al. 2017). Breeding success may be reduced at large colonies because of increased effort (energy expenditure) required due to competition for resources. The evidence therefore indicates that creation of small breeding aggregations on artificial colonies in areas between large natural colonies could potentially result in higher breeding success if the artificial colonies provide conditions with less intra-specific competition and higher nest site quality. While smaller colonies would likely have higher productivity than larger colonies they are at a higher risk of extinction. However, this can be managed for artificial colonies through provision of suitably sheltered nest location, management of predators and *in extremis* artificial feeding of chicks where provisioning to chicks suggest that survival may be in doubt.

#### 2.6.2.1 *Answers to the key biological questions*

The answers to the key biological question in relation to compensation through provision of artificial structures for new kittiwake colonies are shown in Table 13.

**Table 13 Answers to Key Biological Questions in assessing the potential for compensation through provision of artificial structures for new kittiwake colonies.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Do kittiwakes nest on artificial structures? | |
| 1.1 | At FFC SPA? | No. The FFC is a natural colony with no birds nesting on artificial structures. |
| 1.2 | In EAA 3? | Yes. In EAA 3 there are kittiwake colonies on artificial structures along the River Tyne and Dunbar Castle. |
| 1.3 | In all other EAAs in the UK? | Yes. There are kittiwake colonies on artificial structures in several of the other EAAs in the UK. There is a colony in the harbour and surrounding town buildings in Lowestoft, on the pier at Sizewell power station and on gas platforms in Morecambe Bay. Artificial colonies also occur in France, Norway and Alaska. |
| 2 | What is the productivity of breeding kittiwakes on artificial structures? | |
| 2.1 | At FFC SPA? | Not applicable. The FFC is a natural colony with no birds nesting on artificial structures. |
| 2.2 | In EAA 3? | Breeding success of kittiwakes on artificial structures on the Tyne tends to be at least as high, or higher than at nearby natural colonies (JNCC 2020, Turner 2010, 2015, 2016, 2017, 2018, 2019). |
| 2.3 | In all other EAAs in the UK? | Data held by JNCC (JNCC 2020) show typically higher breeding success at artificial structures in Lowestoft than achieved by kittiwakes at FFC SPA. In addition, Christensen-Dalsgaard et al. (2019) showed that breeding success on the oil rigs was significantly higher than at coastal artificial colonies in the same part of the Norwegian coast. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 3 | Would artificial colonies result in increased immigration to SPA colonies? | |
| 3.1 | At FFC SPA? | Yes. There is no direct evidence of movement of fledged chicks from other colonies recruiting to breeding at FFC SPA. However, there is good evidence of the kittiwake population acting as a meta-population. Three chicks ringed at the Saltmeadows tower (Gateshead) were subsequently caught as adults in the colony at Boulogne, France (The Kittiwake Tower Local Nature Reserve - Gateshead Council) and many of the birds ringed on different structures along the River Tyne have moved elsewhere to breed (mentioned without details on web page of Northumbria Ringing Group under 'projects – kittiwake'). Birds ringed as chicks at the warehouse colony in North Shields were subsequently found breeding at colonies in France, Sweden and Germany, as well as in other UK colonies (Coulson 2011). Two chicks ringed at the artificial colony in Dunbar subsequently immigrated to breed at North Shields (Coulson 2011). Coulson (2011) estimated that 91% of female kittiwakes recruiting into a colony are immigrants from other colonies. By comparison, males show stronger philopatry, with 36.5% of recruit males being from the same colony. Nevertheless, even in males, the majority (63.5%) were immigrants from other colonies. Coulson (2011) also pointed out that some new kittiwake colonies form at considerable distances from the nearest established colony: examples include a new colony in Spain and a new colony in Denmark, both more than 500 km from the nearest existing colony. This shows that recruits must in some situations come from considerable distance. Analysis of kittiwake ring recovery data (Coulson and Neve de Mevergnies 1992) showed that recruits tend to originate from colonies within 50 km, but that some move as much as 1,000 km from their natal colony to where they recruit to breed. Coulson (2011) points out that new colonies of kittiwakes tend to grow fast in the years immediately following their establishment. However, kittiwakes are several years old when they breed for the first time, so the growth of new colonies must be due to immigration for several years. At North Shields and at Coquet, it took 9 and 7 years respectively before the first philopatric individuals bred in the colonies. Age of first breeding in the kittiwake is quite variable, from two to ten years of age (Coulson 2011, Table 11.6). Coulson (2011) noted that age of first breeding of males at the North Shields colony decreased from an average of 4.59 years in 1961-1970 to 3.69 years in 1981-1990. The lower age of first breeding in the 1980s coincided with a much-decreased adult survival rate in that decade, suggesting that competition for nest sites at the colony influenced age of first breeding. Presumably birds were able to recruit at a younger age when more vacancies were created by higher adult mortality. Coulson (2011) also noted that immature kittiwakes often take several years of attending the colony before they are successful in establishing themselves as breeders. This kind of density-dependent response has also been seen in other seabird species, such as wandering albatross (where age of first breeding became younger as bycatch mortality of adults at long-line fisheries increased (Croxall et al. 1990)). Similarly, Furness (2015) reported that great skua age of first breeding was younger at smaller colonies and increased when food availability declined. In the kittiwake, this kind of demographic flexibility will be likely to lead to more younger kittiwakes recruiting into new artificial colonies, because competition that otherwise limits their ability to recruit will be lower at such newly established colonies than at existing colonies. Kittiwakes breeding for the first time tend to have lower breeding success than experienced |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| | | birds (Coulson 2011). However, the improvement requires that experience, so birds starting to breed at an older age also have lower productivity in their first breeding attempt. Although experience strongly influences success from first breeding attempt to subsequent ones, actual age of the kittiwake has only a very small influence on success, if any. Coulson (2011) reported that females breeding for the first time at age 3 or 4 achieved 52% breeding success, whereas females breeding for the first time at age 5 or older achieved 48% breeding success. Percentage breeding success increased 10% from first attempt to second attempt but did not change in relation to calendar age at first breeding (Coulson 2011, page 229). Therefore, providing artificial nest sites that allow earlier recruitment into the breeding population will increase life-time reproductive success of birds that are able to recruit at a younger age, and will increase population productivity. |
| 3.2 | In EAA 3? | See the answer to 3.1 |
| 3.3 | In all other EAAs in the UK? | See the answer to 3.1 |
| 4 | How large would the productivity of the artificial colonies need to be to result in a net export of fledglings available to immigrate into other colonies? | |
| 4.1 | At FFC SPA? | Empirical evidence suggests that colonies producing more than 0.8 chicks per nest tend to grow, whereas colonies producing fewer than 0.8 chicks per nest tend to decline in breeding numbers (Coulson 2017). Demographic parameters for kittiwake suggest that about 0.8 chicks per nest would be required to maintain a stable colony size. Therefore, production in excess of about 0.8 chicks per nest would be required for an artificial colony to be self-sustaining and produce excess fledglings potentially available for recruitment into other colonies. Modelling described above suggests that productivity at 0.6 chicks per pair may result in more chicks fledging than needed to maintain colony size, but this assumed the population is closed. |
| 4.2 | In EAA 3? | See the answer to 4.1 |
| 4.3 | In all other EAAs in the UK? | See the answer to 4.1 |
| 5 | How many breeding seasons will it be before new artificial colonies are large enough to result in a net export of fledglings? | |

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| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 5.1 | At FFC SPA? | A new artificial colony could generate a net gain in the overall production of fledglings in a regional population from its first breeding season, because it may allow immature birds that cannot obtain a breeding site at an established colony to recruit into the new site and therefore start immediately to produce fledglings that would not otherwise have existed. However, that does not equate to being a net exporter of recruits into other colonies. The latter will depend on how fast the new colony grows, to what size, and how quickly its productivity rate exceeds that required simply to maintain itself. Until the new colony stops growing faster than nearby colonies, it will tend to have net immigration, especially if it achieves higher than average breeding success (which makes the site more attractive to potential recruits). It cannot stop having net immigration until it has been in existence for at least three or four years of breeding output, because most kittiwakes don't breed until at least three or four years old. Coulson (2011) provides empirical evidence that recruits do not normally start to return until about 7 to 9 years after a new colony is first established. During all those first years the new colony will be supported by a high level of net immigration (100%). It is unlikely that a new colony will switch to being a net source rather than a net sink until it has been producing fledglings for about ten to twenty years as a minimum. However, during that early period it will almost certainly have increased the total number of breeding pairs of kittiwakes in the region/meta-population (by allowing immature birds to breed) and therefore will also have increased the total number of fledglings produced by kittiwakes in the region/meta-population. |
| 5.2 | In EAA 3? | See the answer to 5.1 |
| 5.3 | In all other EAAs in the UK? | See the answer to 5.1 |
| 6 | What are the potential effects of climate change on the ability of new artificial colonies to produce a net export of fledglings? | |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 6.1 | At FFC SPA? | Kittiwakes breed far north and far south of the UK (Mitchell et al. 2004), so it is unlikely that climate change will have major direct effects on the ability of kittiwakes to live and breed at UK latitudes. While climate change may affect kittiwake distribution it is apparent there is considerable uncertainty in the effects on kittiwake populations (Mitchell et al. 2020). Increased storminess and increased intense rainfall during the breeding season may have impacts on kittiwake breeding success through egg and chick mortality. However, to date, storms and rain have been considered less influential on kittiwake breeding success than reduced sandeel availability (Mitchell et al. 2004, Furness et al. 2013, JNCC 2020). Sandeels are vulnerable to impacts of warming sea temperatures. Warmer seas may shift zooplankton communities from large lipid-storing copepods to small copepods that do not store so much lipid. This reduces food quality for sandeels. Warmer seas may also result in lower secondary production, reducing food quantity for sandeels. Warmer seas also increase metabolic rate of sandeel larvae, which can reduce survival of new cohorts so lead to lower production of new age classes of sandeels (Wright et al. 2018). However, those effects appear to be less than the effects of fishing mortality on sandeel abundance, as modelled by Lindegren et al. (2018). Climate change may reduce the extent to which sandeels can recover from stock depletion caused by high fishing mortality, but Lindegren et al. (2018) assessed that partial recovery can be expected despite climate change impacts. Since climate change appears to have a weaker effect in the Irish Sea, Celtic Sea and English Channel, kittiwake colonies in these areas may be less susceptible to climate change. Kittiwakes in these regions also rely on other prey species, such as sprat and herring, which are less affected by changes in sea surface temperature (Mitchell et al. 2020). Kittiwake colonies in Suffolk also rely on these prey species (Martin Kerby, pers. comm.) so may be less affected by climate change than those more reliant on sandeels. |
| 6.2 | In EAA 3? | See the answer to 6.1 |
| 6.3 | In all other EAAs in the UK? | See answer to 6.1 |

### 2.6.3 Control of predators on kittiwakes

No new evidence was found on the efficacy of controlling predators as a suitable compensation measure for impacts on kittiwake SPA colonies.

#### *2.6.3.1 Answers to the key biological questions*

The answers to key biological questions for all predators described in Section 0 to 2.5.7 and shown in Table 14.

**Table 14 Answers to Key Biological Questions in assessing the potential for compensation through control of predators on kittiwake colonies.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 7 | Is there evidence that American mink predation occurs on kittiwake colonies in the UK? | |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 7.1 | At FFC SPA? | **No**. While American mink are present in almost all areas in the UK (Mathews et al. 2018), there is no evidence of mink predation on kittiwakes at FFC SPA. |
| 7.2 | In EAA 3? | **Yes**. Coulson (2011) states 'predation by mammals on kittiwakes is extremely rare'. Furness et al. (2013) found two instances of mink predation affecting kittiwake breeding success at the many colonies monitored by JNCC over many years, both at St Abb's Head to Fast Castle SPA (1999 and 2001). |
| 7.3 | In all other EAAs in the UK? | **No**. Only two instances of predation have been recorded, and only in EAA3. |
| 8 | Is there evidence that feral cat predation occurs on kittiwake colonies in the UK? | |
| 8.1 | At FFC SPA? | **No**. Feral cats are present in almost all urban areas in the UK so it is highly likely that feral cats will be present in the general area. However, there are no records of feral cat predating kittiwakes at FFC SPA. |
| 8.2 | In EAA 3? | **No**. Furness et al. (2013) found a single case of feral cat predation in UK kittiwakes colonies and this was not in EAA 3. |
| 8.3 | In all other EAAs in the UK? | **Yes**. Furness et al. (2013) found a single case of feral cat predation in EAA 4 (Isles of Scilly). |
| 9 | Is there evidence that rat predation occurs on kittiwake colonies in the UK? | |
| 9.1 | At FFC SPA? | **No**. While rats may occur in the general areas around FFC SPA there is no evidence of rat predation on kittiwakes from the SPA. |
| 9.2 | In EAA 3? | **No**. Furness et al. (2013) found a single case of rat predation in UK kittiwake colonies, and this was not in EAA 3. |
| 9.3 | In all other EAAs in the UK? | **Yes**. Furness et al. (2013) found a single case of rat predation in EAA 4 (Isles of Scilly). |
| 10 | Is there evidence that fox predation occurs on kittiwake colonies in the UK? | |
| 10.1 | At FFC SPA? | **No**. While foxes likely occur in the general areas around FFC SPA there is no evidence of fox predation on kittiwakes from the SPA. |
| 10.2 | In EAA 3? | **No**. Furness et al. (2013) found a single case of fox predation in UK kittiwake colonies, and this was not in EAA 3. |
| 10.3 | In all other EAAs in the UK? | **Yes**. There is only one record of fox predation impacting breeding success of kittiwakes, and that is an unusual case where foxes have been able to access some kittiwake nests on artificial structures at Lowestoft (Furness et al. 2013, JNCC 2020), in EAA 5. |
| 11 | Is there evidence that great skua predation occurs on kittiwake colonies in the UK? | |
| 11.1 | At FFC SPA? | **No**. Great skuas only breed in north and west Scotland. None breed in England (Mitchell et al. 2004). |
| 11.2 | In EAA 3? | **No**. Great skuas only breed in north and west Scotland. None breed in England (Mitchell et al. 2004). |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 11.3 | In all other EAAs in the UK? | **Yes**. Furness et al. (2013) stated, "Several kittiwake colonies are affected by great skua depredations (Votier et al. 2004, 2007, 2008). Evidence indicates that the great skuas that kill kittiwakes tend to be birds nesting close to kittiwake colonies (Furness 1987, Votier et al. 2007)." |
| 12 | Have great skuas reduced the size of kittiwake colonies? | |
| 12.1 | At FFC SPA? | **No**. Great skuas only breed in north and west Scotland. None breed in England (Mitchell et al. 2004). |
| 12.2 | In EAA 3? | **No**. Great skuas only breed in north and west Scotland. None breed in England (Mitchell et al. 2004). |
| 12.3 | In all other EAAs in the UK? | **Yes**. Great skuas reduced one population of kittiwakes 54–85% between 1981 and 1995 in Shetland (Heubeck et al. 1999). They have also negatively affected adult survival of kittiwakes on Foula SPA (Oro & Furness 2002). |

## 2.7 Population level assessment

### 2.7.1 Flamborough and Filey Coast SPA

All the population level assessments for FFC SPA kittiwakes were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult survival rate. For a population size of 51,268 pairs and an adult survival rate of 0.854, a 1% increase in baseline mortality would be 150 additional birds being killed per annum. The high impact scenario was based on the current in-combination impact on the population from all offshore wind farms (approximately 590 birds killed per annum) pro-rated to the 2050 net zero target of 100GW of installed capacity. This is an additional 74GW of additional capacity compared to the current level of installed, consented or planned capacity (26GW). This results in an additional mortality of 1,679 adult birds per annum, or a 11.2% increase in adult mortality rate. The medium impact scenario was based on the ratio of low to medium impact derived for other species (specifically gannet and razorbill). The medium scenarios for those species were based upon the pro-rata increase in installed capacity for Round 4 (additional 7GW). The pro-rata increases for kittiwake using this approach was only an additional 9 birds per annum compared with the "low" impact scenario (i.e. 159 vs 150). This was considered to be too small a difference to be useful in determining levels of compensation needed for a level of impact between the "low" and "high" scenarios. Impact levels are summarised in Table 15.

**Table 15 Values for low, medium and high impact scenarios for kittiwakes at FFC SPA.**

| Impact scenario | "Low" | "Medium" | "High" |
|---|---|---|---|
| Additional mortality (no. birds) | 150 | 861 | 1,679 |
| Additional mortality (as a % of baseline mortality rate) | 1% | 5.7% | 11.2% |

#### 2.7.1.1 *Sandeel closure*

As described in 2.6.12.6.1, the sandeel fishery off the east coast of Scotland resulted in a decrease in productivity of kittiwakes on the Isle of May of 0.5 chicks per pair. Therefore, it was assumed that this would be the maximum possible increase in productivity achievable at FFC SPA from closure of the Dogger Bank sandeel fishery. This assumption may be conservative, but there is no evidence for this. Alternatively, the effects of increasing SST from climate change may reduce the benefit from fisheries closure as the recovery of sandeels is hampered by a decline in copepods. Therefore, in addition, it was assumed that the increase in productivity could be lower than 0.5 chicks per pair, so it was tested at arbitrary increases of half this size (i.e. 0.25 chicks per pair) and half this size again (i.e. 0.125 chick per pair). It was therefore assumed that confidence in the compensation measure delivering a given increase in chick productivity increased the lower the anticipated increase in productivity assumed to result from implementation of a sandeel fishery closure measure (i.e. there was higher confidence that 0.125 chick per pair would be realised than 0.5 chicks per pair). Note that these scenarios did not include any likely impact on adult survival, so are conservative in that regard.

These increases in productivity were tested using the Seabird PVA Tool. The inputs for each scenario are summarised in Table 16.

**Table 16 PVA input parameters for sandeel closure compensation scenarios.**

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | 0.5 chicks per pair | 0.25 chicks per pair | 0.125 chicks per pair | |
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 4 | | | PVA app default |
| upper constraint on productivity | 2 chicks per pair | | | PVA app default |
| Initial population size | 51,268 pairs in 2016 - 2017 | | | Aiken et al. (2017) |
| Productivity rate per pair | mean: 0.58, sd: 0.0353 | | | Aiken et al. (2017) |
| Adult survival rate | mean: 0.854, sd: 0.077 | | | PVA app "National" default value |
| Age class 0 to 1 | mean: 0.79, sd: 0.077 | | | PVA app "National" default value |

<table>
<tr><th rowspan="2">Model parameter</th><th colspan="3">Compensation level</th><th rowspan="2">Source</th></tr>
<tr><th>0.5 chicks per pair</th><th>0.25 chicks per pair</th><th>0.125 chicks per pair</th></tr>
<tr><td>Age class 1 to 2</td><td colspan="3">mean: 0.854, sd: 0.077</td><td>PVA app “National” default value</td></tr>
<tr><td>Age class 2 to 3</td><td colspan="3">mean: 0.854, sd: 0.077</td><td>PVA app “National” default value</td></tr>
<tr><td>Age class 3 to 4</td><td colspan="3">mean: 0.854, sd: 0.077</td><td>PVA app “National” default value</td></tr>
<tr><td>Years impacts begin and end</td><td colspan="3">2020 to 2050</td><td>n/a</td></tr>
<tr><td colspan="5">Impact scenarios</td></tr>
<tr><td>Impact on productivity rate</td><td colspan="3">0</td><td>n/a</td></tr>
<tr><td rowspan="2">Impact on adult survival rate</td><td>Low</td><td>Medium</td><td>High</td><td rowspan="2">Calculated as above</td></tr>
<tr><td>0.001462901, se: NA</td><td>0.008397051, se: NA</td><td>0.01637474, se: NA</td></tr>
<tr><td colspan="5">Sandeel fishery closure scenarios</td></tr>
<tr><td>Impact on productivity rate</td><td>-0.5</td><td>-0.25</td><td>-0.125</td><td>0.5 chicks per pair – Frederiksen et al. (2004), remainder assumed</td></tr>
<tr><td>First year to include in outputs</td><td colspan="3">2020</td><td>n/a</td></tr>
<tr><td>Final year to include in outputs</td><td colspan="3">2050</td><td>n/a</td></tr>
<tr><td>Target population size</td><td colspan="3">83,700 pairs</td><td>SACO TPS for FFC SPA</td></tr>
</table>

**PVA results**

The baseline population projection was compared with the three impact scenarios (Table 15). The projections all showed a decline in the population size with time for baseline (unimpacted) and all three impact scenarios (Figure 11).

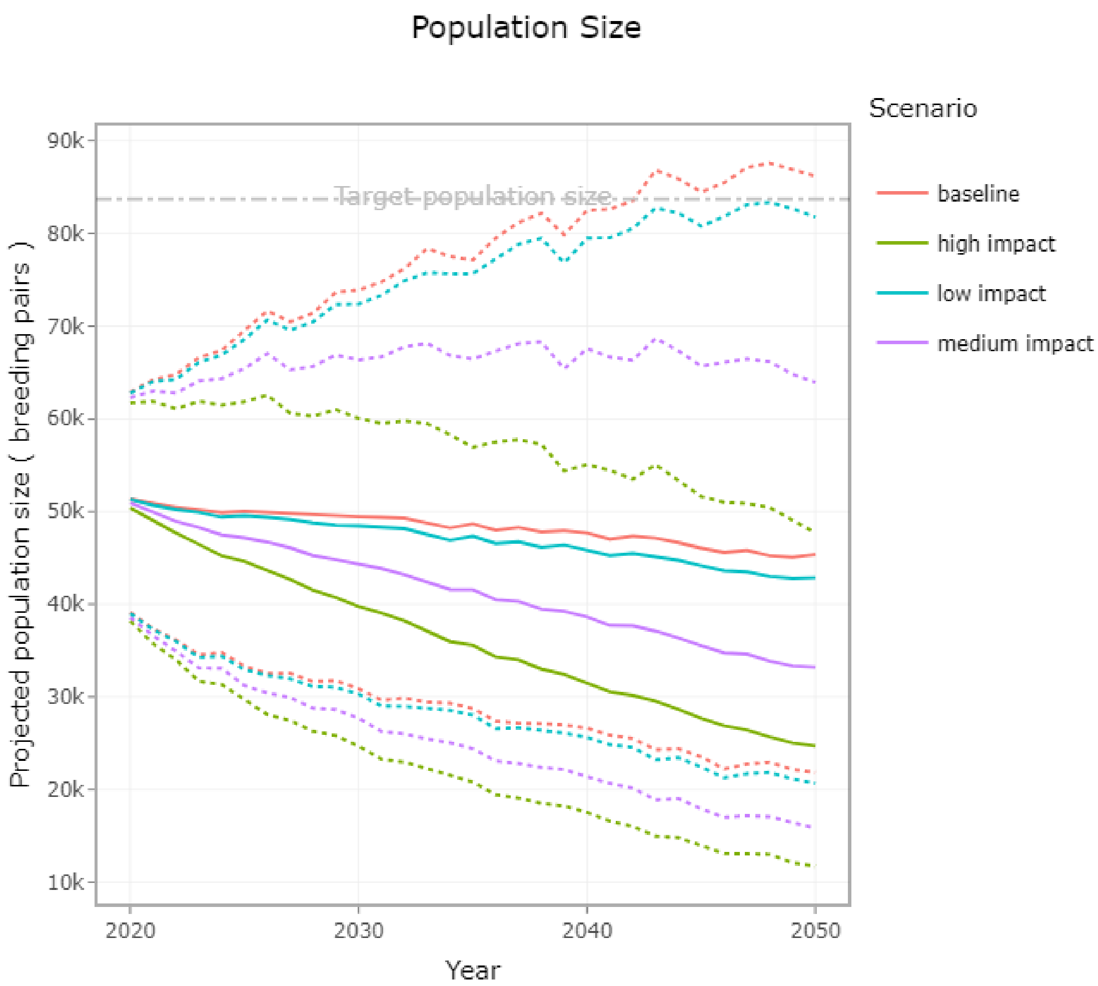


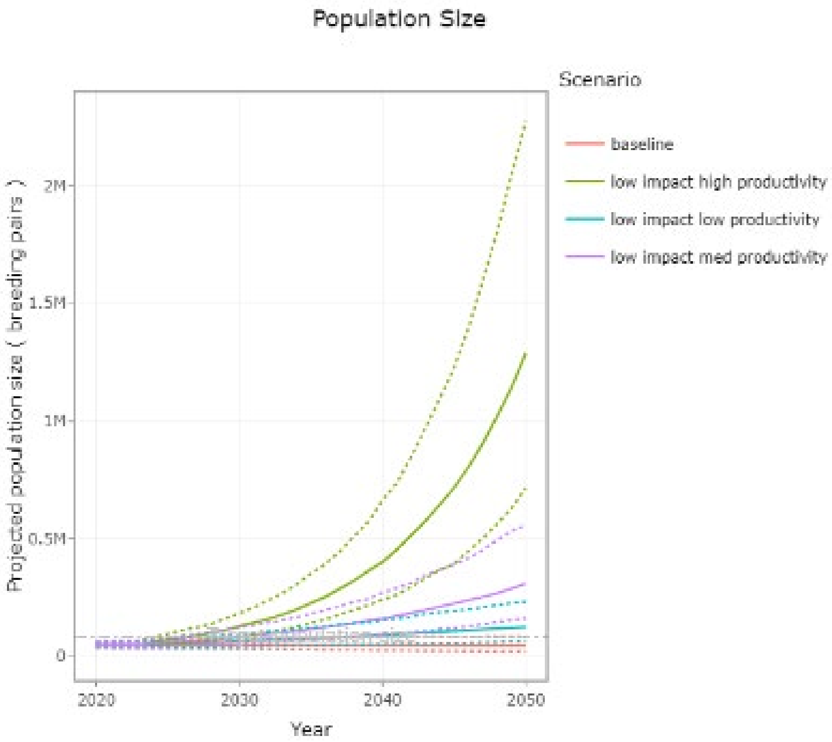

**Figure 11 Population projections for the baseline and low, medium and high impact scenarios.**

In none of the projected scenarios, including the baseline, was the target population size exceeded. The counterfactuals of both population size and growth rate from the projected population in 2050 showed relatively high levels of impact on the FFC SPA kittiwake population.

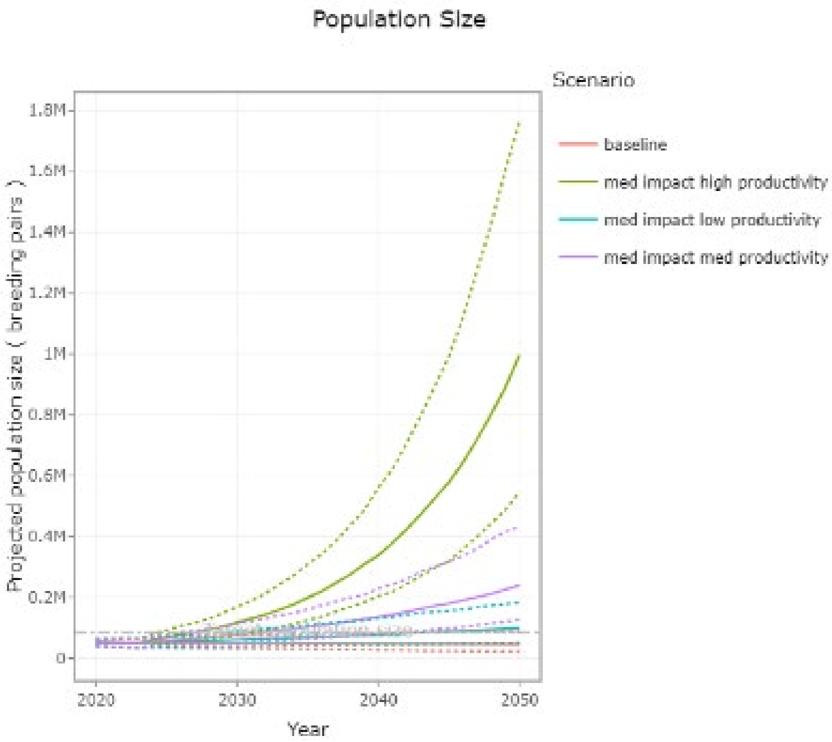
**Table 17 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9474 | 0.9474 (0.9310 – 0.9647 | 0.9982 | 0.9983 (0.9977 – 0.9988) |
| Medium | 0.7341 | 0.7337 (0.7187 – 0.7470) | 0.9901 | 0.9901 (0.9894 – 0.9906) |
| High | 0.5452 | 0.5450 (0.5310 – 0.5579) | 0.9806 | 0.9806 (0.9787 – 0.9813) |

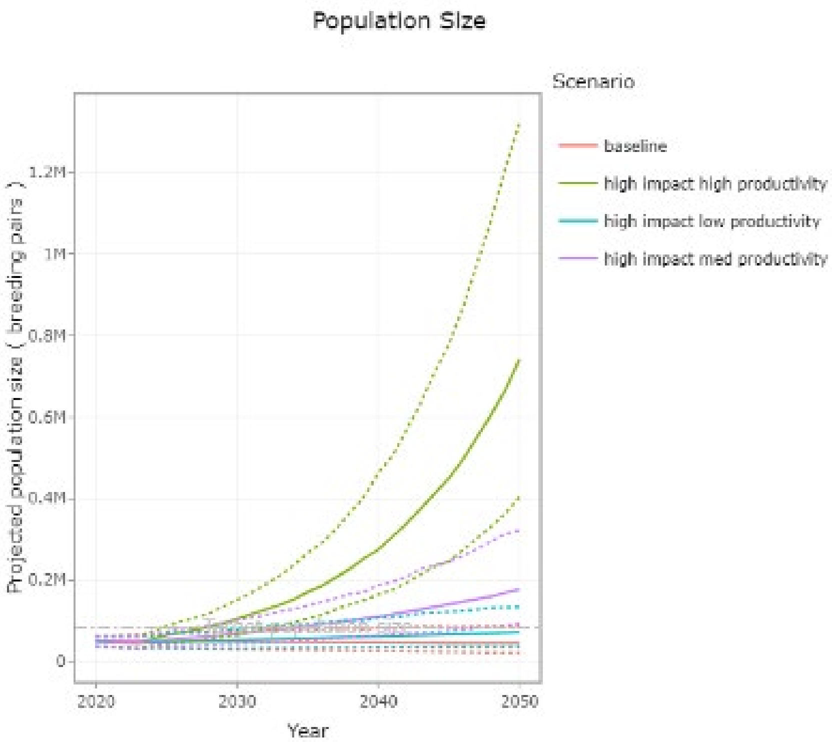
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on productivity (i.e. additional 0.125 (low), 0.25 (medium) or 0.5 (high) chicks per pair) (Figure 12).




Low impact




Medium Impact




High Impact

**Figure 12 Projected population size of breeding kittiwakes (pairs) at FFC SPA comparing baseline with the low, medium and high impact scenarios combined with low, medium and high compensation scenarios.**

In all scenarios the additional productivity resulted in the projected population size increasing (Table 18). The population size increases shown are all likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, all three scenarios suggested that the sandeel closure is likely to result in substantial increases in the kittiwake population at FFC SPA relative to a scenario in which productivity remains at its current low level.

**Table 18 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 0.996 |
| Low | Low | 1.0296 |
| Low | Medium | 1.0595 |
| Low | High | 1.1095 |
| Medium | Low | 1.0211 |
| Medium | Medium | 1.0507 |
| Medium | High | 1.1004 |
| High | Low | 1.0114 |

MacArthur Green

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| High | Medium | 1.0407 |
| High | High | 1.0898 |

In all but one impact and combination scenario the target population size was exceeded within the time span of the population projection (30 years). It was only for the high impact and low productivity (0.125 chicks per annum) where the target population size was not achieved within 30 years. The PVA showed that the population was more sensitive to the scale of change in compensation than the scale of the impacts modelled.

**Table 19 Year in which the projected median population size exceeded the target population size for each combination of impact and compensation scenario.**

| Impact | Compensation | Year target population size exceeded |
|---|---|---|
| Low | Low | 2038 |
| Low | Medium | 2030 |
| Low | High | 2027 |
| Medium | Low | 2044 |
| Medium | Medium | 2032 |
| Medium | High | 2028 |
| High | Low | Not achieved |
| High | Medium | 2035 |
| High | High | 2028 |

Examination of the counterfactuals of population size showed a larger difference in the counterfactual of population size (CPS) between the increase in productivity from the medium to high scenarios than from the low to medium scenarios (Figure 13). This is likely due to the fact that projected population sizes were not bound by density dependent processes preventing continuous growth. As such, the comparison between the CGR values is of more value in this case.

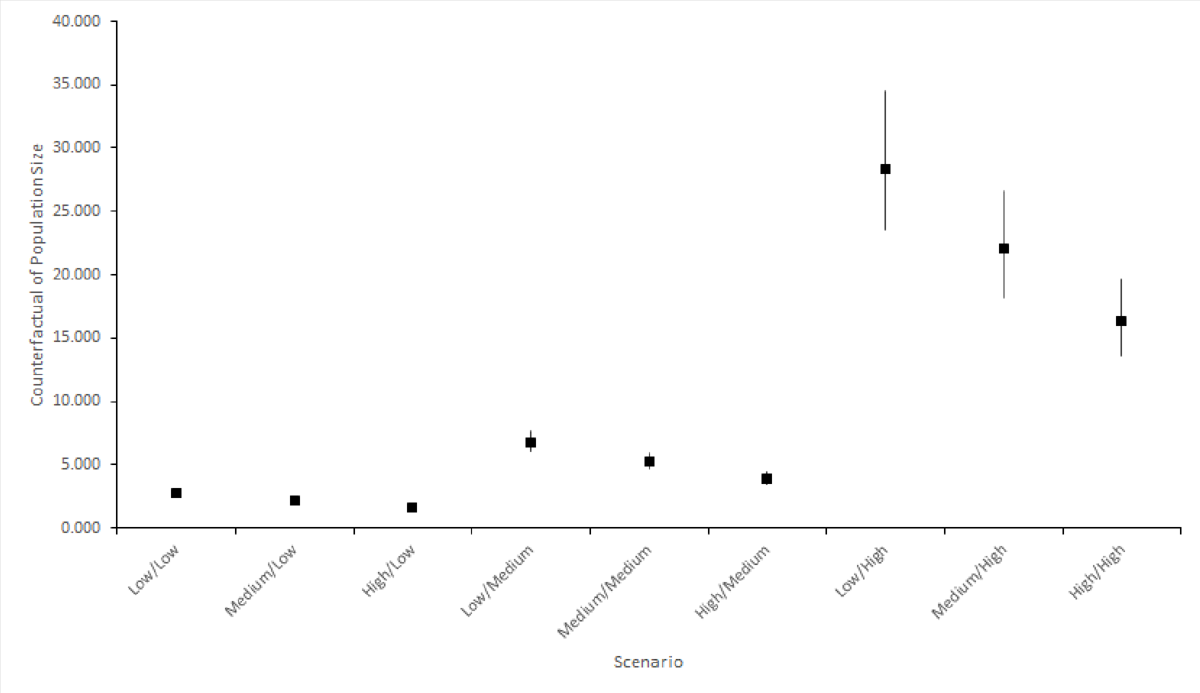



**Figure 13 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the counterfactual of growth rate (CGR) also showed a larger difference between the three compensation level scenarios than between the three impact level scenarios (Figure 14). The CGR was smallest for the low compensation scenarios and largest for the high compensation scenarios. The CGRs for all the high compensation scenarios were larger than those for the low or medium compensation scenarios.

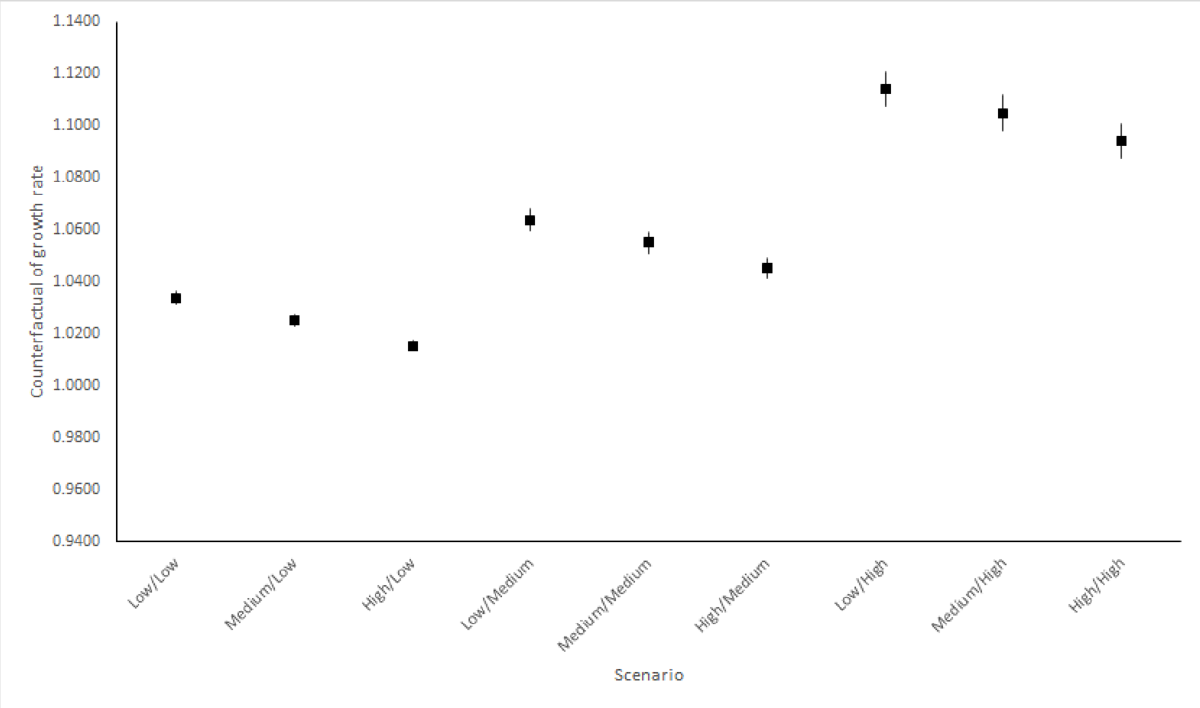



**Figure 14 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

The comparison of mean CGR values (and 95% confidence intervals) does suggest than even if the improvement in productivity is a quarter of that at the Isle of May there is still likely to be a positive outcome on the FFC SPA kittiwake population, and that this would be sufficient to result in population growth for all but the most pessimistic scenario (high impact and low compensation results).

#### 2.7.1.2 *Artificial colonies*

The aim of creating artificial colonies is to produce enough additional chicks over and above what would otherwise be produced without the artificial colonies such that the predicted losses of breeding adult birds from FFC SPA from individual projects are replaced, preferably at the impacted SPA itself or at least within the SPA network, so as to maintain the coherence of that Network. One component of success would be that artificial colonies produce a sufficient number of **additional** fledglings each year (dependent upon the magnitude of the impact being compensated for and any compensation ratios applied) to make it likely that enough reach breeding age each year to offset the **additional** annual mortality being compensated for. This must be coupled with evidence of some of those being seen to recruit to kittiwake colonies in the same region as the impacted colony, and ideally of some recruiting directly back to that impacted SPA. However, given that proving that exactly the required number recruit back to the impacted colony (or indeed to any colonies) is likely to be nigh impossible, all aspects of appropriate measures of success and of compensation ratios must be given due consideration in devising suitable compensation plans.

We created a simple, deterministic, population model to calculate the effects of different input parameters (in particular productivity) on the differences in the number of recruits produced. Two versions were produced, a 5-age class version and a 4-age class version, the latter corresponding to the default Seabird PVA tool formulation and for which the following outputs are discussed. The model was a single sex one, therefore the number of adults also equals the number of pairs.

Firstly, we used the model to calculate the minimum level of productivity for a closed population to sustain itself (i.e. a population growth rate of 1). Using the default "national" adult survival rate of 0.854 applied to age classes 1-2, 2-3, 3-4+, and 0.79 applied to age class 0-1, we varied the colony productivity (chicks per pair) until we had a population growth rate of 1.0 (or very close to this). This productivity was about 0.6 chicks per pair, varying slightly depending on the colony size. A colony with a productivity at or below this (all else being equal) would not produce any excess chicks to replace the losses of adults at any impacted colony (based on the adult survival rate).

We defined four potential levels of productivity. "Very low" productivity (0.8 chicks per pair) was the observation from Coulson (2017) of the level of productivity below which colonies were observed to be declining. This value was used here, rather than the value of 0.6 determined through population modelling, as the observation from Coulson (2017) does not rely on the population being 'closed' (i.e. no immigration or emigration) and other assumptions about population demographics. "Low" productivity (0.96 chicks per pair) was determined from Coulson (2017) as the recent "low" productivity for the River Tyne colony. "Medium" productivity (1.07 chicks per pair) was determined from the colonies on offshore platforms in the Norwegian sector of the North Sea (Christensen-Dalsgaard et al. 2019). "High" productivity (1.23 chicks per pair) was based on the highest level of productivity from the River Tyne colony in the 1950s (Coulson 2017). See Table 1.

We defined three impact levels: "low", "medium" and "high". Low impact was 1% of the baseline mortality level (150 birds from the FFC SPA), medium impact for kittiwake was based on the ratio of low to medium impact from other species (specifically razorbill and gannet), and high impact was based on a pro-rata level from 100GW of installed capacity based on the mortality per GW occurring at present (see Table 20).

**Table 20 Levels of low, medium and high productivity and impact tested using the model.**

| Impact scenario | Productivity (chicks per pair) | Impact (adults killed per year) |
|---|---|---|
| Very low | 0.8 | n/a |
| Low | 0.96 | 150 |
| Medium | 1.07 | 861 |
| High | 1.23 | 1,679 |

The model was constrained by the number of breeding females (i.e. pairs) to replicate the limits of space at an artificial structure. The number of excess adult birds produced by the model was tracked, and the difference between the 'excess' adults and the colony size used an indication of the number of birds which the colony could export (e.g. to offset losses elsewhere). It should be noted that as a single sex model the outputs are effectively doubled in terms of mortalities (i.e. a modelled export of 25 adults is equivalent to 50 of both sexes). The figures below have taken this into account, with the colony size required for each combination of impact and productivity rate doubled.

The approximate artificial colony sizes required by the model to offset the low, medium and high mortality at the three productivity levels were estimated (Table 21).

**Table 21 Predicted numbers of pairs in artificial colonies needed to compensate for low, medium and high impacts based on low, medium and high productivity for three levels of compensation ratio (1:1, 1:5 & 1:10). Cells shaded grey are larger than the current Tyne colony on artificial structures.**

| Impact level | Low | | | Medium | | | High | | |
|---|---|---|---|---|---|---|---|---|---|
| Compensation ratio | 1:1 | 1:5 | 1:10 | 1:1 | 1:5 | 1:10 | 1:1 | 1:5 | 1:10 |
| Very low productivity | 433 | 2,165 | 4,330 | 2,477 | 12,385 | 24,770 | 4,830 | 24,150 | 48,300 |
| Low productivity | 122 | 610 | 1,220 | 698 | 3,490 | 6,980 | 1,362 | 6,810 | 13,620 |
| Medium productivity | 94 | 470 | 940 | 536 | 2,680 | 5,360 | 1,048 | 5,420 | 10,480 |
| High productivity | 70 | 350 | 700 | 402 | 2,010 | 4,020 | 785 | 3,925 | 7,850 |

These results indicate that for the low impact scenario (1% of baseline mortality) relatively few birds would be needed in artificial colonies to compensate for predicted impacts, at least at a 1:1 ratio. As the compensation ratio is increased, the size of artificial colonies required to compensate for the low level of impact increased but only exceeded 1,000 pairs if productivity achieved was low or very low. However, with very low productivity and under a ratio of 1:10 the colony size required i.e. 4,330 pairs was estimated to be larger than the maximum size of the River Tyne colony in the last ten years, which was 1,889 pairs in 2021 (Dan Turner pers. comm.).

Under the medium impact scenario, colony sizes required for any combination of productivity and compensation ratio were greater than for the equivalent combination under the low impact scenario. With the exception of a 1:1 ratio and productivity that exceeded "very low", the required colony sizes were all in excess of 1,000 pairs and exceeded the size of the largest known kittiwake colony on artificial structures.

Under the high impact scenario, a colony size of more than 1,000 pairs was estimated to be required under all combinations except if productivity was assumed to be high and a ratio of 1:1 was considered acceptable. If productivity achieved was very low, even with a ratio of 1:1 the required colony size would be several thousand pairs and so exceed the maximum size of the River Tyne colony in the last ten years. If compensation ratios of 1:5 or 1:10 were considered appropriate, colony sizes would need to reach several thousand pairs and often far exceed the maximum recorded size of the colony on the River Tyne, particularly if productivity was low/very low or a ratio of 1:10 was applied.

#### 2.7.2 EAA3

##### 2.7.2.1 *Sandeel closure*

Population level assessment was not completed for any other SPAs in the EAA 3 area. The following SPAs designated for their breeding kittiwake populations also occur within EAA 3:

- East Caithness Cliffs;
- Troup, Pennan and Lion's Heads;
- Buchan Ness to Collieston Coast;
- Fowlsheugh;
- Forth Islands;
- St Abb's Head to Fast Castle; and
- Farne Islands.

All of these colonies would be expected to benefit from a closure of UK waters to sandeel fisheries, as all of these colonies occur within foraging range of sandeel stocks that are currently or previously depleted through fishing. The sandeel box off the east coast of Scotland has likely positively affected the kittiwake populations at Buchan Ness to Collieston Coast SPA, Fowlsheugh SPA, Forth Islands SPA, and St Abb's Head to Fast Castle SPA. It is important to note that the sandeel box is a fisheries management tool, not a seabird conservation tool, and that current management of the stock that includes the sandeel box allows a take of the stock that could deplete the stock despite the presence of the box. This stock has not experienced the same level of take since the box was put in place, though fishing has occurred within the stock in recent years (Figure 7).

#### 2.7.2.2 *Artificial colonies*

The locations of artificial colonies within the EAAs are less important than the likelihood of creating potential recruits in to the FFC SPA or the wider SPA network. Given the distribution of kittiwake colonies, and the potential for competition with other large colonies described above, it is likely that artificial colonies to south of the FFC SPA would be more successful, so in EAA 5.

### 2.7.3 All other EAAs in the UK.

#### 2.7.3.1 *Sandeel closure*

The closure of UK waters to sandeel fisheries would be expected to mainly benefit colonies that exploit sandeel resources in areas that are currently, or have previously been, subject to sandeel fisheries. For kittiwakes this would (in addition to EAA3) be within EAAs one and five. There are no SPAs designated for their kittiwake populations within EAA 5, but there are 12 within EAA 1 (Calf of Eday, Copinsay, Fair Isle, Foula, Hermaness, Saxa Vord and Valla Field, Hoy, Marwick Head, North Caithness Cliffs, Noss, Rousay, Sumburgh Head and West Westray). The SPAs within EAAs two, four and six are less likely to benefit from closure of UK waters to sandeel fisheries, as there is much less fisheries pressures on sandeel populations in these areas.

#### 2.7.3.2 *Artificial colonies*

As described above, the location of artificial colonies to compensate for impacts to FFC SPA are likely best placed in EAA 5. Colonies in other EAAs would be less likely to benefit FFC SPA, but if placed away from larger existing colonies (to reduce competition for food) they could benefit other SPAs in the UK.

## 2.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the two recommended compensation measures were undertaken for the sandeel compensation approach and the artificial colony approach. The summary table for the confidence in the sandeel fishery closure compensation is shown in Table 22 and the population assessment is shown in Table 23. The confidence in the artificial colony compensation is shown in Table 24 and the assessment is shown in Table 24. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 26 and Table 31 (sandeel fisheries closure) and in Table 29 (artificial colonies).

**Table 22 Assessment of confidence in the recommended compensation method of sandeel fishery closure.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Effect of sandeel stock on kittiwake productivity | Change in kittiwake productivity with sandeel stock | Furness and Tasker 2000, Frederiksen et al. 2004, Carroll et al. 2017. | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Effect of sandeel stock on kittiwake adult survival | Change in kittiwake adult survival with sandeel stock | Oro and Furness 2002, Frederiksen et al. 2004. | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| FFC SPA kittiwakes forage on Dogger Bank sandeel stock | Tracking of adult kittiwakes from Bempton cliffs | Carroll et al. 2017. | Robust | Robust | Medium | Robust | ROBUST | HIGH | VERY HIGH |
| Dogger Bank sandeel stock is depleted | Change in Dogger Bank sandeel stock over time | Lindegren et al. 2018 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Sandeel fishery is depleting the Dogger Bank stock | Change in fisheries landings with time | Lindegren et al. 2018 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Closing the fishery would result in recovery of the stock | Change in sandeel stock with time | Lindegren et al. 2018, Greenstreet et al. 2010 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Closing the fishery would increase kittiwake productivity | Change in kittiwake productivity with sandeel stock | Daunt et al. 2008, Frederiksen et al. 2008, ICES 2020. | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Productivity of kittiwake at FFC SPA has been declining | Change in productivity with time | RSPB Annual Reports | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Fishing mortality of sandeels in the Dogger Bank stock | 0.8 – 1.2 | Lindegren et al. 2018 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Level of sandeel stock to maintain seabird productivity | 666,667 tonnes | Cury et al. 2011, Lindegren et al. 2018 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 23 Assessment of confidence in the inputs to PVA assessing sandeel fishery closure.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 4 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 2 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 51,268 pairs in 2016 - 2017 | Aiken et al. (2017) | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | mean: 0.58, sd: 0.0353 | Aiken et al. (2017) | Medium | n/a | Medium | Robust | MEDIUM | HIGH | MEDIUM |
| Adult survival rate | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 0 to 1 | mean: 0.79, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age class 2 to 3 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Impact on productivity rate of sandeel fishery closure | High = 0.5 chicks per pair | Frederiksen et al. (2004) | Robust | n/a | Low | Robust | MEDIUM | MEDIUM | MEDIUM |
| | Medium = 0.25 chick per pair | n/a | n/a | n/a | n/a | n/a | n/a | n/a | HIGH |
| | Low = 0.125 chicks per pair | n/a | n/a | n/a | n/a | n/a | n/a | n/a | VERY HIGH |
| Target population size | 83,700 pairs | NE FFC SPA SACO | Robust | Medium | Medium | Robust | MEDIUM | HIGH | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 24 Assessment of confidence in recommended compensation method of artificial colony creation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Productivity required to export chicks – observation | 0.8 chicks per pair | Coulson 2017 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Productivity required to export chicks – model | 0.6 chicks per pair | Bespoke population model (see 2.7.1.2) | Medium | n/a | n/a | Robust | MEDIUM | HIGH | MEDIUM |
| Competition for high quality nest sites | n/a | Coulson 2011, Acker et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Density-dependent competition for nest sites at large kittiwake colonies | n/a | Acker et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Density-dependent competition for food | n/a | Wakefield et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 25 Assessment of confidence in inputs to assessing artificial colony creation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 4 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 2 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 51,268 pairs in 2016 - 2017 | Aiken et al. (2017) | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | Low = 0.96 | Coulson 2017 | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | Medium = 1.07 | Christensen-Dalsgaard et al. 2019 | Robust | n/a | Low | Robust | MEDIUM | HIGH | HIGH |
| | High = 1.23 | Coulson 2017 | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Adult survival rate | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 0 to 1 | mean: 0.79, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 2 to 3 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.854, sd: 0.077 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | MEDIUM |

**Table 26 Narratives justifying the evidence and applicability scores for metrics used in recommended compensation method from sandeel fisheries management.**

| Metric | Narrative |
|---|---|
| Effect of sandeel stock on kittiwake productivity | There were multiple publications from high quality studies that kittiwake productivity is negatively affected by reductions in sandeel stock, including one study from FFC SPA. |
| Effect of sandeel stock on kittiwake adult survival | There were multiple publications from high quality studies that kittiwake survival is negatively affected by reductions in sandeel stock. However, with no study from FFC SPA applicability could not be high. **The studies were from similar sites, so confidence was assessed as medium.** |
| FFC SPA kittiwakes forage on Dogger Bank sandeel stock | From the geographic location of the FFC SPA the nearest sandeel stock is the Dogger Bank stock. This is confirmed through GPS tracking of birds. Only 11 birds were tracked, so the sample size is quite low resulting in a Quantity of Evidence score of Medium. Despite this, the overall evidence score is robust, as the connection with the Dogger Bank sandeel stock is quite clear. The evidence is from the FFC SPA, so the Applicability score is High. This results in an overall score of Very High. |
| Dogger Bank sandeel stock is depleted | It is clear from the evidence in Lindegren et al. 2018 that the sandeel stock has been declining, and depleted, for some time. |
| Sandeel fishery is depleting the Dogger Bank stock | It is clear from the evidence in Lindegren et al. 2018 that the sandeel stock is depleted due to the sandeel fishery. |
| Closing the fishery would result in recovery of the stock | While modelling of the sandeel stock by Lindegren et al. 2018, and evidence from closure of the Wee Bankie stock in Greenstreet et al. 2010, shows robust evidence with high applicability concerns about recovery due to climate change and sandeel predator populations have the potential to reduce the recovery of the stock. **As such the assessment of confidence as very high should be reduced to high.** |
| Closing the fishery would increase kittiwake productivity | High quality evidence from studies on the Isle of May provide robust evidence. There is no direct evidence from the FFC SPA population, so applicability was Medium, resulting in an overall confidence assessment of HIGH that kittiwake productivity would increase with closure of the fishery. |
| Productivity of kittiwake at FFC SPA has been declining | Monitoring of the breeding success of kittiwakes as the FFC SPA has been of a high quality and shown that the productivity of nesting pairs has declined with time. |
| Fishing mortality of sandeels in the Dogger Bank stock | High quality evidence from Lindegren et al. 2018 shows that the Dogger Bank stock had a high fishing mortality. |
| Level of sandeel stock to maintain seabird productivity | With evidence from the modelling by Lindegren et al. 2018 showing stock recovery is possible with reduced fishing mortality, and a general concept that one third of the available stock should be available for natural predators from Cury et al. 2011 the evidence was assessed as Robust. While the evidence from Lindegren et al. 2018 applies directly to the stock needed for kittiwakes at FFC SPA, the evidence from Cury et al. 2011 was not from the FFC SPA kittiwake population directly, so the applicability was assessed as Medium. This resulted in a recommended confidence score of High |

| Metric | Narrative |
|---|---|
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence scores for the metrics were high or very high, even after applying some mitigating factors. As such an overall score of high was given. |

**Table 27 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from sandeel fisheries management.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of kittiwakes is well established and not in question. It is not variable between populations and is directly applicable to the FFC SPA population |
| Upper constraint on productivity | Kittiwakes lay two eggs and so productivity cannot be above this. Vary rare occasions where more than two eggs have occurred in nests is likely due to egg dumping. |
| Initial population size | These data are based on recent counts using accepted and standardised methods from the colony being investigated. |
| Productivity rate per pair | These data are based on recent colony productivity plots using accepted and standardised methods. Only samples were obtained from the colony, rather than a whole colony count, so the amount of evidence was assessed as medium. However, they are quantitative data so are robust. It is possible that the samples are too low to give either accurate productivity data, or representative data on the variability of productivity within the colony. The data are highly applicable as they are from the colony being studied. Due to some of the limitation on these data giving a medium evidence score, a medium confidence score has been given despite the high applicability score. |
| Adult survival rate | The adult survival rate evidence is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. However, they are based on national (UK) scale data so there is some applicability to the colony being studies. A medium score was therefore given. A low score would only have been given had the data been from a colony remote from the UK or from a different sea area (e.g. North Pacific). With robust evidence and medium applicability, the confidence score was high. |
| Impact on productivity rate | The evidence for an increase in productivity of 0.5 chicks/pair is based on a study published in a peer reviewed journal by established scientists. Thus, the quality of evidence is Robust. However, with only a single study there can be no score on the consistency of the evidence and the amount of evidence is low. These are quantitative data, so the type of evidence is robust. With robust quality and type of evidence but only a single study, albeit a high-quality study, the overall evidence score was medium. It's applicability for this study was medium as it is not based on evidence from the colony being studies here, but it was from a similar colony on the North Sea coast of the UK. With a medium evidence score and a medium applicability |

| Metric | Narrative |
|---|---|
| | score the confidence assessment in the increase of 0.5 chicks/pair was medium. |
| | The evidence score was not applicable for either medium or low productivity impact, as these were values based on halving of the evidence-based value. These values were chosen to provide additional levels of precaution to the assessment, since the amount of evidence score was low. However, by reducing the potential benefit to productivity from the proposed compensation scenario the confidence that these levels might be achieved increases. Thus, the medium scenario (increase of 0.25chicks/pair) was scored with a high confidence and low scenario (increase of 0.125 chicks/pair) was scored with a very high confidence. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the population consequences of both the impact scenarios negatively affecting the population combined with the compensation scenarios positively affecting the population an overall score of high was given. |

**Table 28 Narratives justifying the evidence and applicability scores for metrics used in in recommended compensation method of artificial colony creation.**

| Metric | Narrative |
|---|---|
| Productivity required to export chicks – observation | Coulson (2017) collated and analysed data from a wide variety of kittiwake colonies across the UK. This was used to show that those colonies that were declining had productivity values below 0.8, while those that were increasing had productivity values above 0.8. So, the evidence was robust, but was not directly about the FFC SPA so the applicability was medium. Therefore, the overall confidence was high. However, we do not know if an artificial colony with a productivity below 0.8 would fail to be self-sustaining as other demographic factors may be acting on the population in the artificial colony. **The overall confidence was therefore reduced to medium.** |
| Productivity required to export chicks – model | The bespoke population model used to assess the productivity level above which the artificial colony would export birds used robust input data and a robust model structure. The applicability was high as it referred specifically to the population of interest. The overall confidence score was medium. However, the model is very simple and based on several important assumptions, not least the assumption that the population is closed. **Confidence in the model output should be modified to low**. However, this does not result in the model not being useful and it indicated that it is possible for an artificial colony with a productivity below 0.8 to still export chicks. |
| Competition for high quality nest sites | There was robust evidence from two sources that there is competition for nest sites in large colonies, including from the FFC SPA colony. Overall confidence was very high. |
| Density-dependent competition for nest sites at large kittiwake colonies | There was robust evidence of density dependence in nest sites in large colonies, though this was not from the FFC SPA colony, so the overall confidence was high. |
| Density-dependent competition for food | There was robust evidence of density dependence in competition for food at large colonies, though this was not from the FFC SPA colony, so the overall confidence was high. |

| Metric | Narrative |
|---|---|
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the evidence that artificial colonies could be a viable compensation measure ranged from medium to very high. An overall score of high was given, but this is very dependent on many factors, and this is a novel approach for this application. **It is therefore prudent to reduce confidence to medium for this approach**. |

**Table 29 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from artificial colonies.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of kittiwakes is well established and not in question. It is not variable between populations and is directly applicable to the FFC SPA population |
| Upper constraint on productivity | Kittiwakes lay two eggs and so productivity cannot be above this. Vary rare occasions where more than two eggs have occurred in nests is likely due to egg dumping. |
| Initial population size | These data are based on recent counts using accepted and standardised methods from the colony being investigated. |
| Productivity rate per pair | The low productivity rate for an artificial colony was based upon empirical evidence from the artificial colony on the River Tyne. This was a long-term study with data published in many peer reviewed journals over decades of research, therefore the quality of evidence was high. A single value from a single study was chosen, rather than a mean of multiple studies, so the consistency is not applicable and was not scored. The evidence was based on large sample sizes of quantitative evidence, so the amount and type of evidence was robust. With only robust scores for all categories, the overall evidence score was also robust. The study was from an artificial colony on the east coast of England, but not one designed or managed specifically for kittiwakes. Thus, the applicability was medium. A high score would only have been given to results from a study of a specifically created and managed artificial colony. **So, the score was reduced to medium.** |
| | The medium productivity rate was based on published estimates from a single study, albeit across several colonies on offshore platforms. As such the quality and type of evidence were robust. However, since it was a single study of a limited number of colonies in a single breeding season the amount of evidence was assessed as low. As with the low (and high) values a single study was chosen, so the consistency of evidence was not relevant. With robust quality and type of evidence but low amount of evidence the overall evidence score was medium. Further evidence from colonies on offshore platforms would allow this metric to be improved, and consistency of evidence assessed. Applicability was considered high, as the offshore oil platforms the colonies were on have been suggested as a potential practical compensation measure. With a medium evidence score and a high applicability score, the confidence was assessed as high. |

| Metric | Narrative |
|---|---|
| | The high productivity rate was also based on the long-term study of the River Tyne colony, so the scores were considered to be the same as the low productivity rate. **Thus, the score was reduced to medium.** |
| Adult survival rate | The adult survival rate evidence is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. However, they are based on national (UK) scale data so there is some applicability to the colony being studies. A medium score was therefore given. A low score would only have been given had the data been from a colony remote from the UK or from a different sea area (e.g. North Pacific). With Robust evidence and medium applicability, the Confidence score was High. |
| **OVERALL CONFIDENCE SCORE** | The confidence scores were mostly high with a few very high scores. The approach however is relatively novel and there are very few relevant examples of artificial colonies that have been created and managed specifically for kittiwakes. **Thus, rather than giving an overall confidence score of high based on the scores for each metric, the score of medium was given to reflect the uncertainty in using novel approaches.** |

With an overall assessment of High in the approach of closing UK sandeel fisheries and a confidence of High in the assessment method (PVA), the assessment of confidence in the proposed compensation methods against the three impact scenarios need to be carefully considered. It was apparent from the population level assessment of the sandeel closure scenarios that the medium and high compensation levels were able to address all the impact levels (see 2.7.1.1), with CPS and CGR values all well above one and populations projected to grow. While CPS and CGR values were also above one for the low compensation scenarios, they were much closer to one than either the medium or high compensation scenarios (Figure 13 & Figure 14). In all cases the population growth rate exceeded one (Table 18). The low compensation scenario was able to provide adequate compensation for the low, medium and high impact scenarios, so was assessed as a medium confidence. For the medium and high compensation scenarios there was high confidence that the impact at low, medium and high levels would be compensated for. While confidence in both the compensation approach and the assessment method was High, it was thought that a Very High confidence was not merited, due to the potential for climate change and large predatory fish populations to suppress the recovery of the Dogger Bank stock. Confidence in this compensation approach could be increased through total reform of the ICES management of the fishery, ensuring adequate stocks for seabirds across the North Sea.

**Table 30 Assessment of confidence in impact and sandeel compensation scenarios.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | MEDIUM | MEDIUM | MEDIUM |

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Medium compensation | HIGH | HIGH | HIGH |
| High compensation | HIGH | HIGH | HIGH |

The confidence in the artificial colony approach to compensation was initially assessed as High, but this was reduced to medium based on the narrative, particularly due to the novel nature of the approach. The population level assessment of the artificial colonies approach had a high confidence, that was reduced to medium after the narratives were considered. At a 1:1 ratio the number of pairs required in artificial colonies was relatively small (Table 21) compared with the existing size of the River Tyne colony. However, with the exception of the low impact scenario and a 1:1 ratio, at the very low productivity level the size of colonies required at any impact level or ratio, being larger than the current size of population on artificial structures on the River Tyne, were too large to compensate. At the high productivity level the low impact scenario could be compensated at all ratios, but for the medium and high impact scenarios only the 1:1 ratio could provide compensation. Assimilating all of these factors results in High confidence in using artificial colonies only when the impact was low, and if the compensation level was low to high i.e. productivity achieved exceeds the very low value modelled of 0.8 chicks/pair (Table 31). It may even be reasonable to conclude that the confidence in compensating Medium and High impacts was low, as both the 1:5 and 1:10 ratios could not be compensated even with high productivity levels without assuming unrealistically large colonies being attracted to artificial structures. In general, artificial colonies are likely best considered as a project specific measure, rather than a strategic level approach.

**Table 31 Assessment of confidence in impact and artificial colony compensation scenarios (1:1) scenario.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Very low compensation | MEDIUM | LOW | LOW |
| Low compensation | HIGH | MEDIUM | MEDIUM |
| Medium compensation | HIGH | MEDIUM | MEDIUM |
| High compensation | HIGH | MEDIUM | MEDIUM |

## 2.9 Future monitoring and adaptive management

### 2.9.1 Closure of sandeel fisheries

If compensation through closure of the sandeel fishery in UK waters was applied, it would be important that suitable monitoring is put in place to demonstrate that this has been effective at:

- Increasing the sandeel stock available to the kittiwake population at FFC SPA;
- Increasing the provision of sandeels to chicks at the FFC SPA;
- Improving the productivity of kittiwakes at FFC SPA; and
- Increasing the survival of adult kittiwakes at FFC SPA.

Methods for suitable monitoring of sandeel stocks would need to be established with experts in this field and is beyond the scope of this study. Monitoring would need to determine overall abundance of the stock and perhaps also the stock within the foraging range of kittiwakes from the FFC SPA to ensure that the compensation measure is having the desired effect on the prey resource for the population at FFC SPA.

In addition to assessing the stock at both the relevant ICES stock area and within the foraging range of the FFC SPA kittiwake population, monitoring of the provision of food to chicks would be important. Collecting data on the species, number and size of fish being fed to chicks in the SPA before compensation was applied as well as during the closure of sandeel fishery could be used to determine whether there had been a change in chick provisioning. These data would be used to determine whether the closure of the sandeel fishery has resulted in a positive change in the provision of sandeels to chicks at the FFC SPA colony.

Whether a change in the provision of sandeels to chicks results in a change in productivity would also need to be monitored. In addition to continuing the current SMP monitoring plots at the SPA more careful monitoring of productivity of nests where fish provisioning is monitored would be necessary to show that any change to productivity was due to changes in sandeel provision.

Finally, monitoring of the return rate of adult kittiwakes to the FFC SPA colony would be useful as a proxy for monitoring of adult survival, as a change in return rate would be sufficient to indicate that adult survival had changed.

These monitoring measures need to be connected to adaptive management decision making. The proposed monitoring needs to be considered together when adapting the management to the results of the monitoring. The aim of this proposed compensation measure is to increase the productivity and/or adult survival of kittiwakes at the FFC SPA colony specifically, with an overall objective of maintaining or increasing the population size. However, it is important to note that compensation is required to maintain the integrity of the SPA network. Monitoring of sandeel stocks is needed to determine whether recovery of the stock was as expected, below the level expected or above the level expected. Similarly adaptive management will need to consider whether action is necessary if the change in productivity or adult survival are above or below the expected value. Ultimately the need to adapt management actions will need to be based on whether the population size at the SPA changes as a result of the proposed compensation method. Adaptive management actions will need to consider the pattern of change in all the monitored elements before deciding whether, and what type, of corrective action is needed.

Even if the population size does not increase in the FFC SPA, no further action may be appropriate if the other elements have been shown to have positively changed and the kittiwake population in the UK network of SPAs can be shown to have increased. This could occur where the FFC SPA becomes a source population, exporting birds in excess of that needed for intrinsic growth, or it could be as a result of competition from other species that also benefit from the closure of the sandeel fishery. Where closure of the fishery can be shown not to have had a positive effect on the FFC SPA kittiwake population it will be important to understand why this is before deciding on subsequent management actions. It may be necessary to move to other compensation mechanisms should the closure of the sandeel fishery ultimately prove to be unsuccessful. However, if climate change results in no recovery, or decline, of the sandeel stock there are unlikely to be short term solutions for compensating for the kittiwake population as it should decline regardless of management to a level where it meets the carrying capacity of the environment.

There were no significant gaps in knowledge needing to be filled. However, further research on the effects of climate change on sandeel prey and the effects of predatory fish populations on sandeel recovery would be useful.

#### 2.9.2 Provision of artificial structures

For any artificial colonies created to provide compensation of the FFC SPA kittiwake population there will need to be careful monitoring of the colonies. This should aim to determine whether the colonies are providing a net benefit to the wider kittiwake meta-population. As such the key elements to monitor would be:

- Population size of the colony;
- Productivity of the colony;
- Natal philopatry;
- Emigration from the colony; and
- Return rate of breeding adults.

Monitoring of the colony size and productivity should be undertaken annually. This would be to ensure that the colony is still functioning as expected and that growth occurs to the capacity of the structure. It would also be important to determine that the productivity across an average of multiple years is enough to allow colony growth through natal philopatry, and then an excess of chicks available to recruit into the wider population. Ringing of all chicks from the colony would likely be important in determining whether growth is from natal recruitment or immigration. As the colony grows ringing of chicks would also be important to determine whether excess chicks successfully recruit into other colonies, including FFC SPA. This would best be achieved through individual colour marks that could be read in the field (i.e. coloured engraved Darvic rings). Some efforts to resight colour ringed birds at other colonies would also be beneficial, as well as encouragement of reporting by bird watchers. Ringing of adults would also be important to determine their return rate to the colony, either as a proxy for adult survival or to model adult survival based on resighting probabilities.

These monitoring results will need to inform adaptive management decisions. Where the population size of the colony either does not increase to fill the capacity of the structure, or is very slow to grow, the reasons for this should be investigated and where possible the structure should be adapted to improve the suitability for nesting birds. There may also need to be some monitoring and control of predators if they are shown to negatively affect the population size or growth of the colony. Structures should be designed to minimise the accessibility of nest sites to terrestrial mammalian predators and the effectiveness of those designs should be monitored. Where predatory birds kill adult birds, chicks or eggs, suitable methods to limit predators' access to nests through design changes may be needed. Failing that, control measures may be needed, varying from lethal control (e.g. for crows) to diversionary feeding (e.g. for species where lethal control is illegal or undesirable). Where productivity of the colony is too small to either maintain the population or export excess birds into the wider population the reason for this should be investigated and suitable management action taken. The type of management taken will likely depend on the reasons for the low productivity. For instance, if weather related effects are resulting in chicks on part of the colony dying then the design of this part of the colony should be adapted to improve the shelter to the birds in those areas. Supplementary feeding of chicks to increase productivity has been successfully applied for research purposes (Benowitz-Fredericks et al. 2013), so could be used to increase productivity where this can be shown to be limited by adult provisioning of chicks.

Where the combination of population size and productivity from an artificial colony remains lower than desired then increasing the size of the colony or creating further colonies could be a suitable management action to address the export of excess chicks. However, care may be needed as larger colonies can result in lower productivity through competition where foraging resources are limiting. If a colony is being limited by food supply, then either supplementary feeding or a further colony at another location where competition would not occur may be a suitable management action.

### 2.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for kittiwakes at FFC SPA: the closure of sandeel fisheries in the Dogger Bank area (ICES Area 1r) and creation of artificial nesting colonies. Both of these compensation methods could increase the population size at FFC SPA. New evidence was found in relation to both of these compensation approaches. Newer information on sandeel stocks in the Dogger Bank area (ICES Area 1r), assessments of their fishing mortality and the clear link to kittiwake productivity was summarised. More recent information on the use of artificial structures by kittiwakes, the productivity of colonies on these structures and the colonisation of offshore structures were summarised.

Overall, it was apparent that strategic level compensation to kittiwakes at FFC SPA could be achieved through closure of the sandeel fishery in UK waters and ecology focused management of the sandeel fishery across the North Sea. This approach has been shown to be successful in improving the conservation status of kittiwakes in the Forth Islands SPA (Isle of May). Population modelling aided in concluding that the recovery of the FFC SPA population could be rapid and that achieving the target population size was possible, even if the closure was less successful in increasing kittiwake productivity than the measures taken for the Forth Islands SPA colony. It was also shown that this compensation measure would likely also provide sufficient compensation for impacts from future projects, potentially even for 2050 targets. Confidence was high in both the compensation method (Table 26) and the assessment method (Table 27). Since both the high and medium compensation scenarios, regardless of the level of impact being assessed, showed both CPR (Figure 13) and CGR (Figure 14) values well in excess of one, confidence should be high that these scenarios would deliver more than adequate compensation. For the low compensation scenario, where only one quarter of the improvement shown in the empirical data from the Isle of May could be achieved, confidence was medium for the low and medium impact scenarios. While there was higher confidence that the lower compensation level could be achieved through closure of the UK sandeel fishery, the ability for this to overcome the impact scenarios was much lower. For the high impact scenario there was low confidence in the ability of the low compensation scenario to be sufficient. Implementation of these measures would require ongoing adaptive management alongside on-going Habitats Regulations Assessments of future offshore wind development plans. This could be used to ensure that the capacity of the kittiwake populations benefitting from fisheries closures were not exceeded. Adding various adaptive management and monitoring plans would help reduce uncertainty in the benefits of the compensation measures to kittiwake populations, including the colony at FFC SPA.

Construction of artificial structures for kittiwake has at least some potential to have a positive effect on the growth rate of the FFC SPA kittiwake population, but only at low impact levels for compensation ratios greater than 1:1. Well designed, situated and managed artificial colonies appear to have the potential to be beneficial at relatively small impact scales, but would need to be large to provide compensation at higher impact levels even at a ratio of 1:1 and would need to be unrealistically large if ratios greater than 1:1 were required. Whether these colonies would directly benefit the FFC SPA is unknown, but as the SPA colony is part of the wider kittiwake meta-population in northern Europe it seems likely some birds would be recruited into the SPA. It is important to note that the sizes of the impact being compensated for are very small relative to the overall size of the meta-population and even the FFC SPA colony. As such, monitoring of the FFC SPA colony is unlikely to be able to demonstrate a positive effect. Ongoing monitoring and research on immigration and emigration within the meta-population (e.g. through tracking and/or colour ringing) would likely be needed to determine whether artificial colonies are beneficial to the FFC SPA directly.

Overall, the ability for artificial colonies to compensate for impacts to the FFC SPA at the scales tested here was relatively limited. Practical and financial costs were not considered here. It seems likely that this compensation approach, while having some merit for strategic scale compensation, is perhaps best used as an individual project solution.

For both the compensation and the impact scenarios that have been assessed here it is important to take account of the scenario testing approach used here when interpreting the results. The three impact scenarios are based on predicted impact levels, and these may differ from the realised impacts from constructed offshore wind farms. Future research is likely to change key elements of those assessments (e.g. avoidance rates applied to collision risk modelling), and realised impact may be higher, or lower, than those predicted. The medium and high impact scenarios are also based on assumptions about the level of impact increasing in direct proportion with the installed capacity of the turbines. However, it is likely that installed capacity will not have a linear relationship with predicted impact. For instance, collision risk increases linearly with the number of turbines, if they are all of equal size and rotational speed. However, as turbine nameplate capacity increases, fewer turbines are needed to reach any given level of installed capacity. A general rule of thumb is that fewer larger turbines will have a lower predicted collision risk than more, smaller, turbines for the same level of installed capacity. The high impact scenario also does not take into account likely changes in predicted impacts from decommissioning. At least some of the wind farms that contribute to the overall current in-combination impact may be decommissioned by 2050, thus reducing the predicted impact level.

There were also various assumptions and uncertainties in the compensation scenarios. While the high compensation scenario from sandeel fisheries closure in UK waters was based on empirical evidence from the closure of the fishery in the sandeel box on the east coast of Scotland, the proposed closure here is for the whole of the UK fishery. This may result in a better response from breeding kittiwakes than found on the Isle of May, but there is no evidence for this. However, in testing scenarios where the increase in productivity is halved and halved again gives confidence that this proposed compensation measure would very likely result in improvements to the productivity of the FFC SPA kittiwake population. However, it was not possible to account quantitatively for the potential effects of climate change on copepod abundance, a key food source for sandeels. It is possible that long term effects of climate change will reduce the sandeel abundance in the North Sea and therefore the ability of this compensation measure to increase the productivity of the FFC SPA kittiwake population. If climate change could result in impacts to sandeel abundance this highlights the importance of both reducing or eliminating the fishing pressure on the sandeel stock in the North Sea and the need to transition to low carbon electricity generation. The potential for climate change to reduce the efficacy of this compensation measure strengthens the argument for applying this measure, rather than reducing it.

# 3 FLAMBOROUGH AND FILEY COAST SPA – BREEDING GANNET

## 3.1 Conservation status of gannet

The global gannet population was estimated at 390,000 pairs around 2000 (Mitchell et al. 2004), of which 220,000 pairs bred in Great Britain (Wanless et al. 2005). Colonies have been increasing at a rate of about 2% per annum for several decades (JNCC 2020). JNCC (2020) estimated that the UK population in 2015 was about 293,200 pairs, representing 55.6% of the global population.

The UK population was amber listed in Birds of Conservation Concern (BOCC) 4 (Eaton et al. 2015) and BOCC3. JNCC SMP data show a long-term increase in the population index for the UK from 100 in 1986 to almost 200 in 2018 (JNCC 2020). It is protected under the Birds Directive as a migratory species. The SPAs in Great Britain were estimated to hold 95.9% of the Great Britain breeding population of gannets present in 2000 (Stroud et al. 2016).

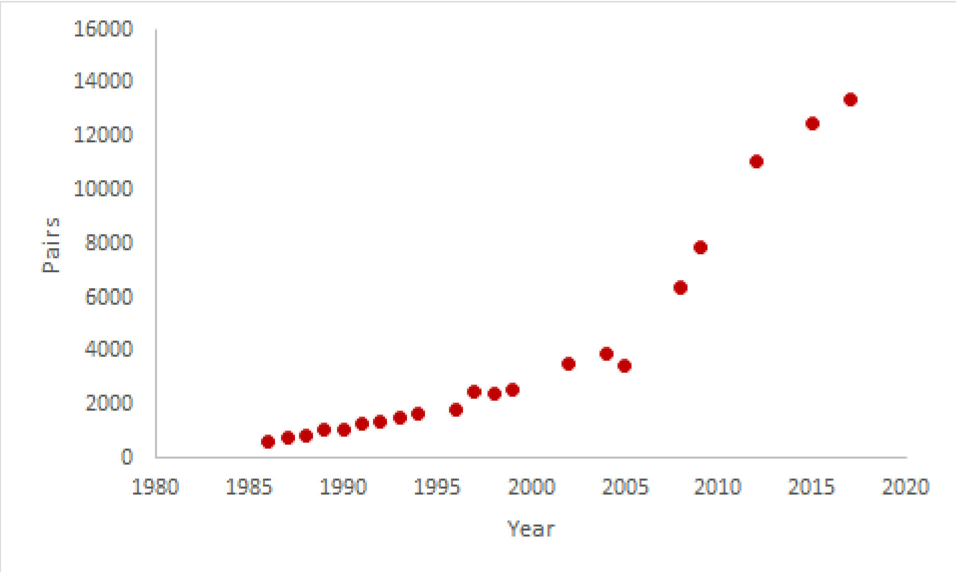



**Figure 15 Counts of numbers of pairs of gannets at Flamborough and Filey SPA (data from JNCC SMP database and RSPB; Aitken et al. 2017).**

There are eight sites in Scotland with breeding gannet listed in the citation as an SPA feature. There is one in England (FFC SPA), and one in Wales. In the most recent assessment of site condition, the conservation status of the breeding gannet at all sites in the UK was classified as Favourable Maintained. Overall, the Natura suite for breeding gannets should be considered at present to be in Favourable conservation status. The population of the colony in the FFC SPA (at Bempton) has increased since the mid-1980s, with a sudden rise in the rate of increase of the population after 2015 (Figure 15). The population in 2017 was about 13,392 pairs (Aitken et al 2017). It is the only gannet colony in England.

## 3.2 Citation population size

The population citation is 8,469 pairs (5 year average 2008-2012), according to the Natura 2000 Standard Data Form.

## 3.3 Conservation objectives

The site has conservation objectives "to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site."

More detailed Supplementary Advice on Conservation Objectives (SACO) have since been added online, last updated 13 March 2020 (Natural England 2020). For gannet at FFC SPA these are:

- Maintain the size of the breeding population at a level which is above 8,469 pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Maintain safe passage of birds moving between nesting and feeding areas;
- Restrict the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding) at: current extent;
- Maintain the distribution, abundance and availability of key food and prey items (e.g. herring, mackerel, sprat, sandeel) at preferred sizes;
- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;

- Maintain water quality and specifically mean winter dissolved inorganic nitrogen (DIN) at a concentration equating to High Ecological Status (specifically mean winter DIN is < 12 μM for coastal waters), avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for FFC pSPA was published in February 2015 (NE 2015). That identified public access/disturbance as a threat to gannets and identified prevention of disturbance as a responsibility of East Riding of Yorkshire Council, Natural England, RSPB, Scarborough Borough Council, Yorkshire Wildlife Trust, and Flamborough Management Scheme. No other threats or pressures affecting gannets at FFC SPA were specifically identified as requiring management in the SIP.

#### 3.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are three main sources of impact on gannet from offshore wind farm development: mortality due to collisions with operational turbines, displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of gannets from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. However, studies on gannets have found no apparent carry over effects of migratory distance (Pelletier et al. 2020) or non-breeding foraging strategy (Grecian et al. 2019) on breeding season demography or body mass.

The maintenance of the population of each of the qualifying features could be affected directly through collision mortality and indirectly through impact to energy budgets from displacement and barrier effects.

### 3.4 Location of compensation

Cook et al. (2011) defined two Ecological Assessment Areas for gannet (Figure 16). The FFC SPA occurs within EAA 1. Consequently, the hierarchy of the locations of compensation are:

1. FFC SPA;

2. EAA 1; and
3. EAA 2.

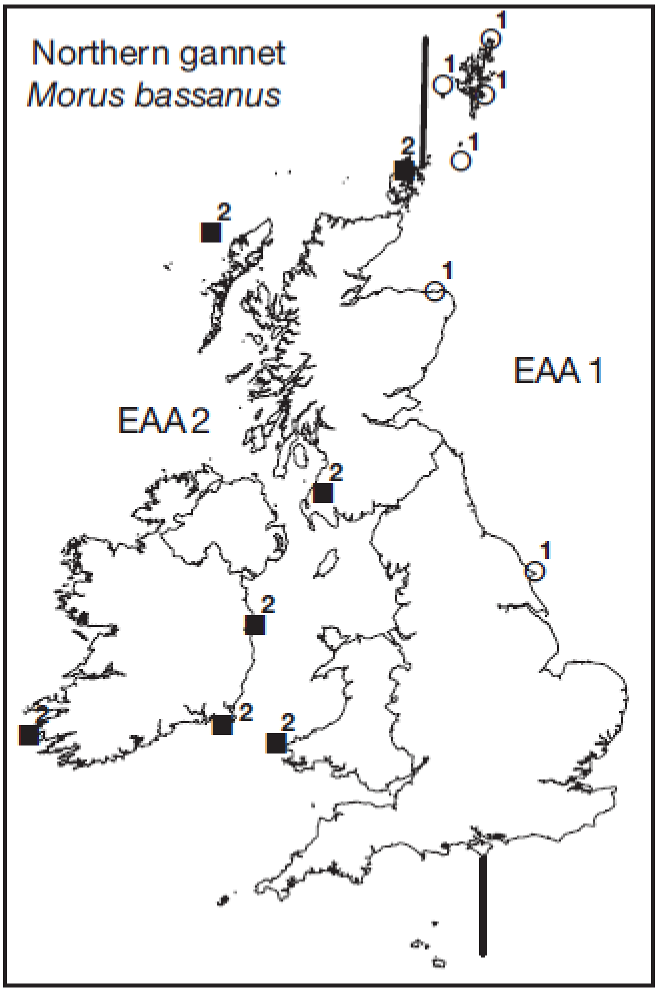



**Figure 16 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for gannet by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

### 3.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding gannets were developed based on the potential compensation measures reviewed by Furness et al. (2013):

- End harvest of chicks;
- Encourage establishment of new colonies; and
- Reduce bycatch in fisheries.

Furness et al. (2013) concluded that only ending the harvest of gannet chicks was highly likely to be effective, with high confidence based on available evidence. Subsequently a PVA has been used to assess the impact of harvesting gannet chicks that is licenced to be carried out at Sula Sgeir (North Rona & Sula Sgeir SPA). There have also been further studies since 2013 examining the fisheries bycatch of seabirds that provides better evidence of benefits that could be possible for gannet populations through changes to fisheries practices to reduce bycatch. Finally, there is also recent evidence of changes in gannet conservation status at some colonies outside the British Isles that may be important in predicting future population trends of colonies in the UK.

For each of these potential compensation measures a series of key biological questions directly related to the compensation of impacts on the FFC SPA breeding gannet population was identified. These help to clarify the nature of the evidence required to inform an assessment of the likelihood that any given compensatory measure may succeed in offsetting predicted impacts from offshore windfarms.

**Table 32 Key Biological Questions in assessing the potential for compensation of breeding gannet at FFC SPA.**

| No. | Key Biological question |
|---|---|
| **End harvest of chicks** | |
| 1 | Is the harvest of chicks from Sula Sgeir limiting the population? |
| 2 | How many adult birds would be added to the UK gannet population through the ending of harvesting of chicks at Sula Sgeir? |
| **Encourage establishment of new colonies** | |
| 1 | Is there evidence that new gannet colonies can be established through management actions? |
| 2 | Could new gannet colonies could be established at: |
| 2.1 | FFC SPA? |
| 2.2 | Colonies in EAA 1? |
| **Reduce bycatch in fisheries** | |
| 1 | Does bycatch in UK waters affect gannets from: |
| 1.1 | FFC SPA? |
| 1.2 | SPAs in EAA 1? |
| 1.3 | SPAs in EAA 2? |
| 2 | Does the migration of gannets put birds at risk from bycatch in non-UK waters? |
| 2.1 | From FFC SPA? |
| 2.2 | From SPAs in EAA 1? |
| 2.3 | From SPAs in EAA 2? |
| 3 | Are there suitable mitigation measures for the types of fisheries with gannet bycatch off the Iberian Peninsula and north-west Africa in winter? |

## 3.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 3.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

The majority of the global gannet population breeds in Britain and Ireland, so this species is of very high conservation value in the UK. While most of the UK population are qualifying features of SPAs, and all of these sites are in favourable condition with population sizes considerably above citation levels, due to the high conservation value compensation may still be needed in some situations.

### 3.6.1 Changing ecological conditions

Gannets have recently colonised new breeding areas in the Barents Sea (Barrett et al. 2017). Their spread to Bear Island in 2011 is thought to be associated with a warming of the Barents Sea and northward spread of prey fish, especially mackerel and herring. Although some colonies in that region have been growing rapidly, some recently established colonies have declined or been abandoned. Declines were attributed to disturbance from an increasing population of white-tailed eagles (Barrett 2008; Pettex et al. 2015). The increasing numbers of white-tailed eagles in Norway was anticipated to cause further declines at some gannet colonies (Barrett et al. 2017). White-tailed eagles are increasing in Scotland after their re-introduction as a breeding species and are currently being reintroduced in England, and while they have not yet been seen to affect gannet breeding colonies, this possibility clearly exists in future.

A large decline in the return rates of tagged breeding gannets at Rouzic, France, was reported by Gremillet et al. (2020). Return rates changed from 100% in 2006–2007 to less than 30% after 2015. This decline was consistent with a decrease in inter-annual survival probabilities for ringed adult gannets, from >90% in 2014–2015 to <60% in 2018–2019. This occurred during population decline of the Rouzic gannet breeding colony. This was most likely caused by a combination of strong fishing pressure on pelagic fish stocks, increased fishery bycatch, and intentional harvesting of adult gannets by fishermen as food (Gremillet et al. 2020). Most gannets that breed at Rouzic spend the winter in the Canary Current off West Africa, an area that is thought to have especially high incidence of bycatch and intentional harvest of gannets, as well as high fishing mortality imposed on pelagic fish stocks (Gremillet et al. 2020). Some adult gannets from UK colonies also winter in that area (Fort et al. 2012) and so there may be impacts on UK birds too.

There was an unprecedented abandonment of nests and breeding failure by gannets at colonies in eastern Canada in late summer 2012 (Montevecchi et al. 2021). This was related to historically low levels of the main prey fish of gannets in eastern Canada in 2012, suggesting that birds may have been food stressed. However, abandonment occurred during a marine heat wave and intense thunderstorms. Low breeding success is exceptionally unusual in gannets, so this event appears to have been driven by exceptional climate conditions during a year of exceptionally low food abundance. This does suggest that increasing weather extremes from global climate change may result in larger and/or more frequent breeding failure in UK gannet populations, including FFC SPA.

### 3.6.2 Harvesting of gannet chicks

The colony at Sula Sgeir has shown an 8% increase from SPA designation to 2013; less growth than in any other UK SPA with breeding gannet as a feature. Sula Sgeir is the only colony in the UK at which a harvest of gannet chicks is allowed. A licence to take up to 2,000 fully grown chicks per year has been provided annually by NatureScot. The reported numbers taken each year are close to this limit (an average of 1,917 per year from 2004 to 2014; Trinder 2016). Disturbance in the colony from the harvest is likely to also reduce the breeding success of some pairs which are not harvested. Thus, productivity is likely reduced by even more than the absolute number of chicks actually removed by the harvest. Population modelling (Trinder 2016) showed that the harvest of chicks at Sula Sgeir appeared to have reduced the rate of population growth below the level predicted in the absence of a harvest. Trinder (2016) suggested that this probably also reduced the growth rate of other gannet colonies in the region, since there is strong evidence from population modelling of natal emigration of birds between colonies. There is, however, very little empirical evidence of the rate of natal dispersal among colonies, so the strength of the meta-population relationships among gannet colonies is uncertain. Trinder (2016) calculated that the Sula Sgeir population required a little more than 270 breeding age immigrant recruits per annum to achieve the level of growth observed. This suggested that the Sula Sgeir population is a population sink for emigrant gannets from other colonies, and therefore if the legal harvest of chicks was to end, there would likely be increased growth rates of neighbouring colonies, as well as that on Sula Sgeir. Trinder (2016) stated that "*The Sula Sgeir population grew at a rate of 2.2% over the last decade, which is below the Scottish rate of 2.9%. However, this colony also appears to have been supported through recruitment from other colonies, since without immigration the estimated growth would have been less than 1%. Given that the analysis presented here indicates exchange between Scottish colonies, the removal of individuals from one colony would seem very likely to have effects on other connected colonies. While it remains possible that the level of estimated immigration to Sula Sgeir may not be affected by the magnitude of harvest experienced, it does seem likely that the reduction in internal recruitment (i.e. by chicks hatched at Sula Sgeir) presents increased opportunities for external recruitment. Thus, the interchange between colonies indicates that harvesting from Sula Sgeir has in the past, and likely will in future, also have effects on other populations*".

While Sula Sgeir is the only UK gannet colony where a chick harvest occurs, there are also harvests of gannet chicks at colonies in the Faroe Islands and Iceland. Reducing the harvest at those colonies could also represent compensation, since the northeast Atlantic gannet population is a meta-population with natal dispersal occurring between all colonies. Ending the harvesting of gannet chicks is highly likely to be effective in compensating impacts on the gannet population generally and therefore on the network of SPAs in the UK, with a high confidence based on evidence.

#### *3.6.2.1 Answers to the key biological questions (3.5).*

The answers to the key biological question in relation to compensation through harvesting of gannet chicks are shown in Table 33.

**Table 33 Answers to Key Biological Questions in assessing the potential for compensation through harvesting of gannet chicks.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is the harvest of chicks from Sula Sgeir limiting the population? | **Yes.** Population modelling (Trinder 2016) showed that the harvest of chicks at Sula Sgeir appeared to have reduced the rate of population growth below the level predicted in the absence of a harvest. |
| 2 | How many adult birds would be added to the UK gannet population through the ending of harvesting of chicks at Sula Sgeir? | The ending of the annual harvest of an average of 1,917 gannet chicks from Sula Sgeir (Trinder 2016) would result in an additional 495 adult birds available to recruit into the UK gannet population each year. This is based on the compound survival rate from fledging to age of first breeding of 0.258 from the age specific survival rates used in the PVA (3.7). With approximately 95.9% of the UK gannet population in SPAs approximately 474 adult gannets would be available to recruit into the UK SPA network each year (assuming that the current harvest causes |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| | | no reduction in productivity except that due to removal of harvested chicks; in practice some losses due to disturbance could also be reduced and so compensation by terminating the harvest is likely to exceed the equivalent of 474 adult gannets per year). This also assumes that all birds would recruit into UK gannet colonies. It is likely that some would recruit into colonies in the Faroe Islands, Iceland and Ireland. |

### 3.6.3 Encourage establishment of new colonies

No new evidence could be found on the establishment of new gannet colonies. The conclusions of Furness et al. (2013) remain. It is apparent that there have been two attempts to encourage establishment of new northern gannet colonies that have both been unsuccessful, although similar actions have been successful with the closely related gannet that occurs in Australia and New Zealand.

#### *3.6.3.1 Answers to the key biological questions (3.5).*

The answers to the key biological question in relation to compensation through establishment of new gannet colonies are shown in Table 34.

**Table 34 Answers to Key Biological Questions in assessing the potential for compensation through establishment of new gannet colonies.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that new gannet colonies can be established through management actions? | **No**. No new evidence could be found. It appears that establishing new gannet colonies through management actions is relatively unlikely to be successful, despite the fact that breeding numbers are increasing, and some natural colonisation of new sites is occurring. |
| 2 | Could new gannet colonies could be established at: | |
| 2.1 | FFC SPA? | **No**. There is an existing colony at FFC SPA. |
| 2.2 | Colonies in EAA 1? | **No**. See the answer to question 1. |
| 2.3 | Colonies in EAA 2? | **No**. See the answer to question 1. |

### 3.6.4 Fisheries bycatch mortality

Bradbury et al. (2017) produced a GIS tool which showed the relative risk of UK seabirds to bycatch from fisheries in UK waters. These authors identified gannet at risk of bycatch in fisheries in UK waters, with the risk higher in summer than other seasons and higher in inshore waters of Scotland than other geographic locations. Miles et al. (2020) reported a preliminary assessment of seabird population response to potential bycatch mitigation in the UK-registered fishing fleet. It was concluded that bycatch mortality of gannets in this fishery represented slightly more than 1% of the annual natural mortality. Using data from Northridge et al. (2020), Miles et al (2020) estimated that the annual bycatch of gannets by UK-registered fishing vessels was between 25 and 764 birds per year. Bycatch of gannets in UK waters may be relatively small compared to bycatch of UK gannets occurring outside the breeding season in wintering areas, including the Bay of Biscay, Iberian shelf waters and off West Africa, with apparently high and increasing take of gannets off West Africa.

Gannet was the seabird most frequently caught as bycatch taken by Portuguese mainland coastal fisheries, particularly on demersal long-lines and in set nets, but also taken in purse-seine catches (Oliveira et al. 2015). These fisheries overlap with the main wintering area of UK gannets, so will be catching some birds from UK SPA populations. These limited data from Oliveira et al. (2015) suggest that the bycatch from fisheries in southern Europe may kill more gannets each year than the predicted precautionary estimates of collision mortality at offshore wind farms in the UK. However, sampling intensity of bycatch in the fisheries was low, and there was uncertainty about the bycatch taken when observers monitoring this were not on-board vessels. This implies that the bycatch may be even larger than reported.

Calado et al. (2020) reported that gannet was frequent in the bycatch taken by fisheries in the Atlantic Iberian coastal waters, especially in long-lines. Gannet bycatch occurred throughout the year, with bycatch in summer mainly being immature gannets. Immature birds remain in southern European waters while adults have returned to breeding colonies, so it is not surprising that immature birds occurred in larger proportions at that time of year. These authors concluded that the scale of the bycatch could have significant impacts on the whole gannet population.

A large bycatch and substantial harvest of gannets for food in West African waters was reported by Gremillet et al. (2020), but the scale of this problem was unclear. Mauritanian authorities confiscated eight containers of frozen seabirds from fishing boats in early 2013 (thought to contain tens of thousands of birds, including many gannets) destined for shipping to Asia and intended for human consumption (EU 2020). This practice is illegal and may have increased recently and now represents a significant threat to gannet populations and appears to represent by far the highest level of anthropogenic additional mortality imposed on gannets. Reducing that bycatch and harvest would therefore provide considerable scope for compensation. However, the unregulated nature and lack of monitoring of this impact may make it difficult to address. Regulation within West Africa may be especially difficult. However, landing frozen seabirds in countries in Asia could possibly be regulated.

In contrast to the information above, Clark et al. (2020) investigated behavioural responses of breeding adult gannets in Iceland to fishing vessels using GPS tracking. Fishery discarding is illegal in Iceland and gannets in Iceland did not switch from travelling to foraging when they came close to fishing vessels. Foraging trips by gannets were relatively short, suggesting high availability of preferred food (presumed to be pelagic fish). It was concluded that the lack of an association between gannets and fishing boats in Iceland was due to a combination of high availability of pelagic forage fish and a lack of discarding by Icelandic fishing boats providing an alternative food source. This implied less risk of bycatch from fisheries in Iceland, so shows a potential management approach to reduce bycatch.

ICES (2013) identified the following fishing methods as having the potential to by-catch northern gannets:

- Bottom otter trawl;
- Midwater otter trawl;
- Midwater pair trawl;
- Purse-seine;

- Trammel net; and
- Set longlines.

The Bycatch Management Information System[2] lists potential mitigation measures for different fisheries. It only list methods for purse seine, gillnet and longlines. It was assumed that methods for purse seines could potentially be applied to bottom and pelagic trawls, as the bycatch risks from these nets are likely to be relatively similar from the gannet bycatch perspective. Trammel nets are a type of gillnet, so the advice for gillnet bycatch mitigation was assumed to be applicable. The potential bycatch mitigation method for each fishing type is summarised in Table 35.

**Table 35 Mitigation measures recommended by the BMIS (shaded grey) for the fisheries identifed by ICES as potential sources of gannet bycatch. Methods of potential value to gannet bycatch mitigation are summarised as Y (Yes), U (unknown) and N (No).**

| Mitigation measure | Fishery | | | | | |
|---|---|---|---|---|---|---|
| | Bottom otter trawl | Midwater otter trawl | Midwater pair trawl | Purse-seine | Trammelnet | Set longlines |
| Auditory deterrents and attractors | N | N | N | N | N | N |
| Bait size and condition, hooking technique | | | | | | Y |
| Chemical deterrents or attractants | N | N | N | N | | N |
| Decoys | | | | | U | |
| Double-weight branchlines | | | | | | Y |
| Dyed bait | | | | | | U |
| Gear configuration – other | Y | Y | Y | Y | | Y |
| Hook Shielding Devices | | | | | | Y |
| Hookpod | | | | | | Y |
| Illumination of gillnets | | | | | U | |
| Light Cues - attractors & deterrents | | | | | U | |
| Line weighting & bait sink rate | | | | | | Y |
| Management of abandoned, lost, discarded fishing gear (ALDFG) | Y | Y | Y | Y | Y | Y |
| Management of offal discharge | | | | | | Y |
| Night / day setting | | | | | Y | Y |
| Safe handling & release | Y | Y | Y | Y | Y | Y |
| Seabird Saver | U | U | U | U | | U |
| Side-setting | | | | | | Y |
| Sliding Leads (Lumo Leads) | | | | | | Y |

[2] https://www.bmis-bycatch.org/

| Mitigation measure | Fishery | | | | | |
|---|---|---|---|---|---|---|
| | Bottom otter trawl | Midwater otter trawl | Midwater pair trawl | Purse-seine | Trammelnet | Set longlines |
| Smart Tuna Hook | | | | | | Y |
| Spatial & temporal measures | Y | Y | Y | Y | Y | Y |
| Streamer (tori) lines | | | | | | Y |
| Sub-surface gillnets | | | | | | |
| Underwater setting techniques | | | | | | Y |
| Vessel-specific management | U | U | U | U | | |
| Water cannon or fire hose | | | | | | N |

It is important to note that there is no published evidence of the application of these mitigation measures for bycatch of gannet specifically. Most publications have been focused on other seabirds, particularly albatrosses and petrels. However, bycatch mitigation, including bird-scaring lines, night setting and line-weighting, have been successfully applied as bycatch mitigation to seabirds in South Africa, including Cape gannet *Morus capensis* (Rollinson et al. 2016). The demersal hake longline fishery in Namibia had one of the highest seabird mortality rates in the world (Petersen et al. 2009). However, the use of scaring lines and line weights to increase the sink rate were shown to be successful in reducing bycatch (Paterson et al. 2019). The use of scaring lines and water sprayers have been shown to mitigate bycatch in trawl nets in Australia, which included Australian gannet *Morus serrator* (Koopman et al. 2018),

#### *3.6.4.1 Answers to the key biological questions (3.5).*

The answers to the key biological question in relation to compensation through fisheries bycatch mitigation are shown in Table 36.

**Table 36 Answers to Key Biological Questions in assessing the potential for compensation through fisheries bycatch mitigation.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Does bycatch in UK waters affect gannets from: | |
| 1.1 | FFC SPA? | **Probably.** Using data from Northridge et al. (2020), Miles et al (2020) estimated that the annual bycatch of gannets by UK-registered fishing vessels was between 25 and 764 birds per year. |
| 1.2 | SPAs in EAA 1? | See the answer to question 1. |
| 1.3 | SPAs in EAA 2? | See the answer to question 1. |
| 2 | Does the migration of gannets put birds at risk from bycatch in non-UK waters? | |
| 2.1 | From FFC SPA? | **Yes.** Gannets from the FFC SPA are highly likely to migrate to waters off the coast of the Iberian Peninsula and north-west Africa in winter. There is good evidence of gannet bycatch in these water in winter. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2.2 | From SPAs in EAA 1? | **Yes.** Gannets from SPA colonies in EAA 1 are highly likely to migrate to waters off the coast of the Iberian Peninsula and north-west Africa in winter. There is good evidence of gannet bycatch in these water in winter. |
| 2.3 | From SPAs in EAA 2? | **Yes.** Gannets from SPA colonies in EAA 2 are highly likely to migrate to waters off the coast of the Iberian Peninsula and north-west Africa in winter. There is good evidence of gannet bycatch in these water in winter. |
| 3 | Are there suitable mitigation measures for the types of fisheries with gannet bycatch off the Iberian Peninsula and north-west Africa in winter? | **Yes.** Suitable mitigation measures are identified in Table 35. |

## 3.7 Population level assessment

### 3.7.1 Flamborough and Filey Coast SPA

All the population level assessments for FFC SPA gannets were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult survival rate. For a population size of 13,391 pairs (Aitken et al. 2017) and an adult survival rate of 0.919 a 1% increase in baseline mortality would be 21.7 additional birds being killed per annum.

The medium impact scenario was based on a pro-rata increase in the current in-combination impact on the FFC population from all offshore wind farms (439 birds killed per annum) based on the current capacity of the developments included in that in combination assessment (26GW) and the anticipated capacity of proposed Round 4 offshore wind farms (an additional 7GW). This resulted in an estimated additional mortality of 118 birds per annum associated with that 7GW increase in installed capacity.

The high impact scenario was based on the current in-combination impact on the FFC population from all offshore wind farms (439 birds killed per annum) pro-rated to the 2050 net zero target of 100GW of installed capacity. This is an additional 74GW of additional capacity compared to the current level of installed, consented, or planned capacity (26GW). This results in an additional mortality of 1,249 adult birds per annum, or 57.6% increase in adult mortality rate. Impact levels are summarised in Table 37.

**Table 37 Values for low, medium and high impact scenarios for gannets at FFC SPA.**

| Impact scenario | Low | Medium | High |
|---|---|---|---|
| Additional mortality (birds) | 21.7 | 118 | 1,249 |
| Additional mortality (rate) | 1% | 5.4% | 57.6% |

#### 3.7.1.1 *Fisheries bycatch mortality*

The baseline population was compared against the low, medium and high impact scenarios using the Seabird PVA Tool with the input parameters shown in Table 38. The only compensation scenarios that could be parameterised and included in a PVA model of the FFC gannet population were for reductions in the bycatch of adult gannets from UK fishing vessels. Miles et al. (2020) estimated a bycatch of 25 to 764 gannets per annum. From a total UK breeding population size of 291,328 pairs (JNCC 2020) and 95.9% of this population within SPAs (Stroud et al. 2016) it was calculated that the proportion of the UK breeding population in FFC SPA was 4.6%. Thus, from a low bycatch estimate of 25 birds from Miles et al. (2020), 1 individual should be from the FFC SPA, which was used as the low compensation scenario. The upper bycatch estimate from Miles et al. (2020) was therefore 35 birds from FFC SPA, used as the high compensation scenario. The medium scenario therefore needed to be some point near the middle of these two values. Northridge et al. (2020) estimated that approximately 300 birds per year were taken as bycatch from UK fishing vessels, this was close to the mid-point between the upper and lower values from Miles et al. (2020), so was used as a medium scenario. This equated to 14 birds per year from the FFC SPA gannet population.

**Table 38 PVA input parameters for fishery bycatch in UK waters by UK fishing vessels compensation scenarios.**

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 5 | | | PVA app default |
| upper constraint on productivity | 1 chick per pair | | | PVA app default |
| Initial population size | 13,391 pairs in 2017 | | | Aiken et al. (2017) |
| Productivity rate per pair | mean: 0.81, sd: 0.0679 | | | Aiken et al. (2017) |
| Adult survival rate | mean: 0.919, sd: 0.042 | | | PVA app "National" default value |
| Age class 0 to 1 | mean: 0.424, sd: 0.045 | | | PVA app "National" default value |

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| Age class 1 to 2 | mean: 0.829, sd: 0.026 | | | PVA app "National" default value |
| Age class 2 to 3 | mean: 0.891, sd: 0.019 | | | PVA app "National" default value |
| Age class 3 to 4 | mean: 0.895, sd: 0.019 | | | PVA app "National" default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Impact on adult survival rate | Low | Medium | High | Calculated as above |
| | 0.000810246, se: NA (21.7 birds) | 0.004405944, se: NA (118 birds) | 0.0466358, se: NA (1,249 birds) | |
| **UK fishery bycatch elimination** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Adults from FFC SPA not killed | 1 | 14 | 35 | Miles et al. (2020) |
| First year to include in outputs | 2020 | | | n/a |
| Final year to include in outputs | 2050 | | | n/a |
| Target population size | 8,469 pairs | | | SACO TPS for FFC SPA |

For each impact scenario the adult survival rate used in the PVA for each combination of impact scenario and compensation scenario was calculated as a single adjusted survival rate, which is summarised in Table 39.

**Table 39 Adjusted adult survival rate for each combination of low, medium and high impact with low, medium and high compensation for gannets at FFC SPA.**

| | Adjusted adult survival rate | | |
|---|---|---|---|
| Compensation level | Low (1) | Medium (14) | High (35) |
| Low impact | 0.0007729 | 0.000287507 | -0.000496602 |
| Medium impact | 0.004368606 | 0.003883205 | 0.003099096 |
| High impact | 0.046598462 | 0.046113061 | 0.045328952 |

**PVA results**

The baseline population projection was compared with the three impact scenarios (Table 38). The projections showed an increase in the population size with time for baseline (unimpacted) and the low and medium impact scenarios (Figure 17). Only the high impact scenario was projected to result in population decline.

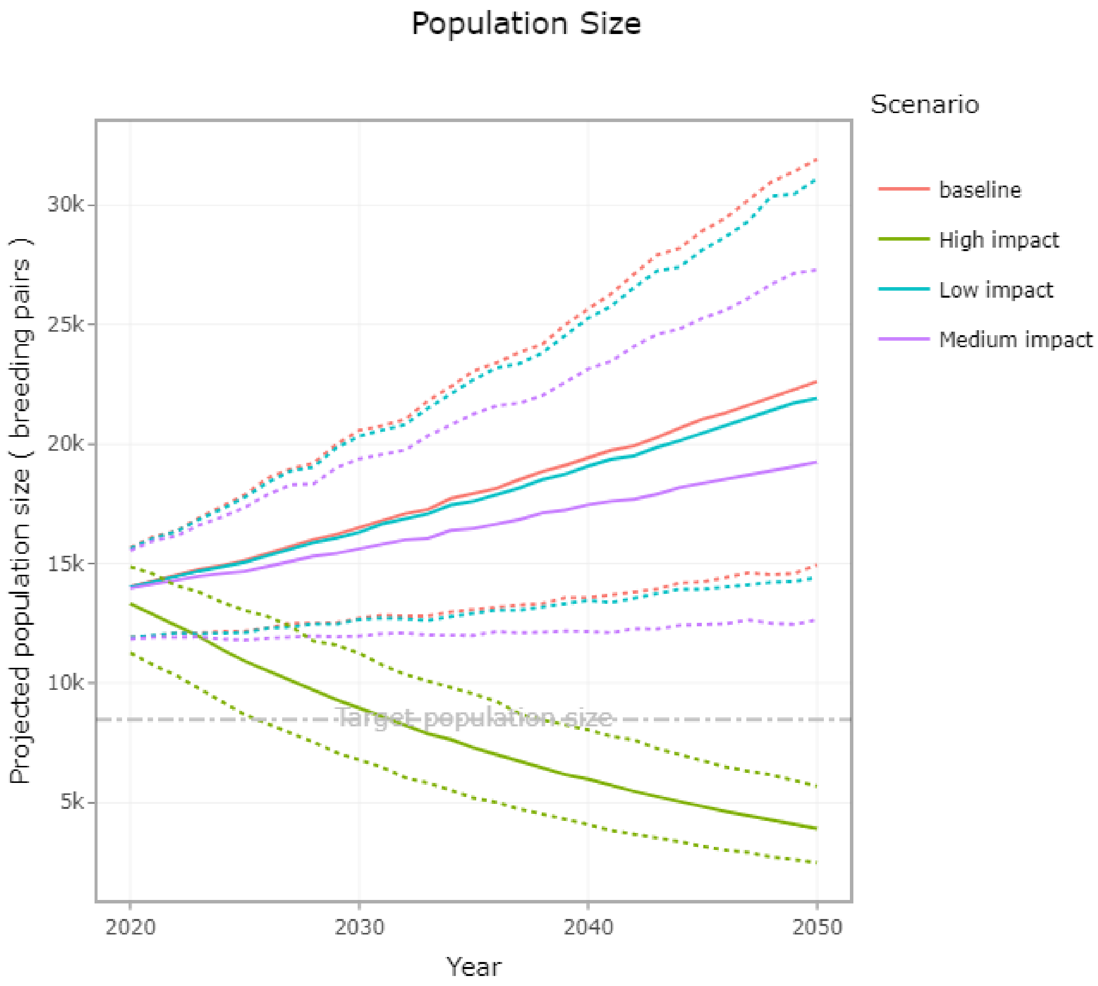


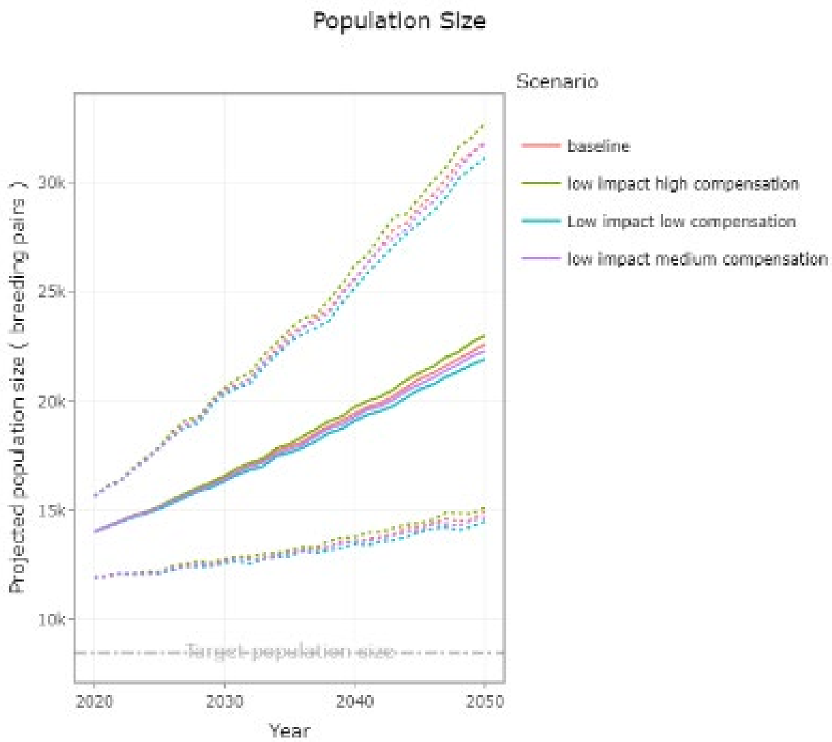

**Figure 17 Population projections for the baseline and low, medium and high impact scenarios.**

The starting population for all scenarios was above the target population size. The baseline population projection, and the low and medium impact scenarios, remained above the target population size. Only the high impact scenario predicted population decline with the population declining below the target population size between the late 2020s and late 2030s. The counterfactuals of both population size and growth rate from the projected population in 2050 showed a wide range of effects on the FFC SPA gannet population. The low impact scenario had relatively little effect on the population, with the population still growing at a very similar rate to the baseline population. The medium impact scenario showed a more important effect on the population, but the population would still increase, albeit at a lower rate. The high impact scenario showed a large and very important effect on the population, with a strong decline and a much lower population size in 2050 compared with the baseline population projection.

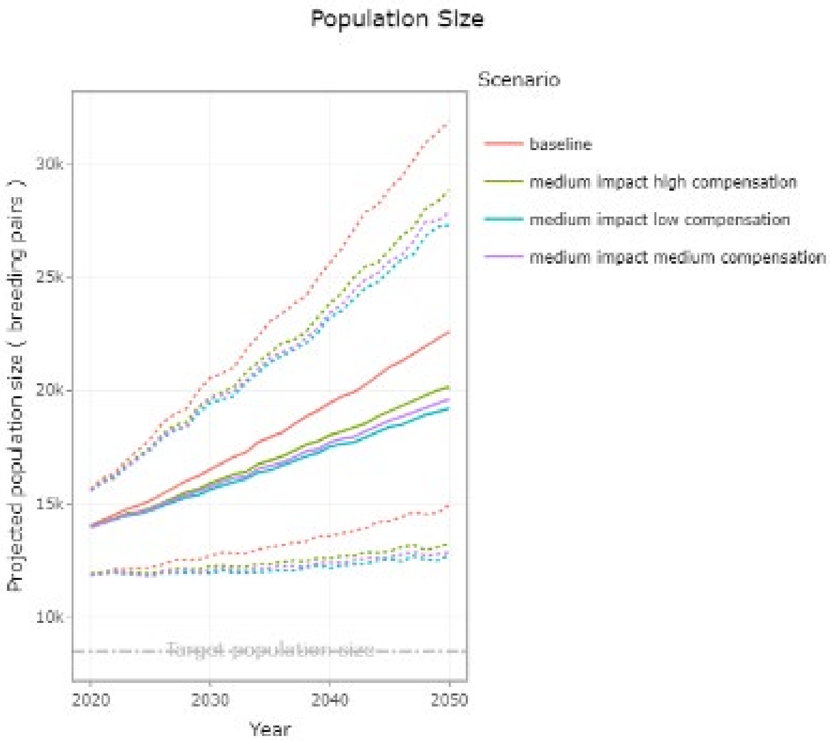
**Table 40 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9712 | 0.9712 (0.9486 - 0.9950) | 0.9991 | 0.9991 (0.9983 - 0.9998) |
| Medium | 0.8509 | 0.8509 (0.8290 - 0.8724) | 0.9948 | 0.9948 (0.9940 - 0.9956) |
| High | 0.1729 | 0.1729 (0.1652 - 0.1805) | 0.9450 | 0.9449 (0.9436 - 0.9463) |

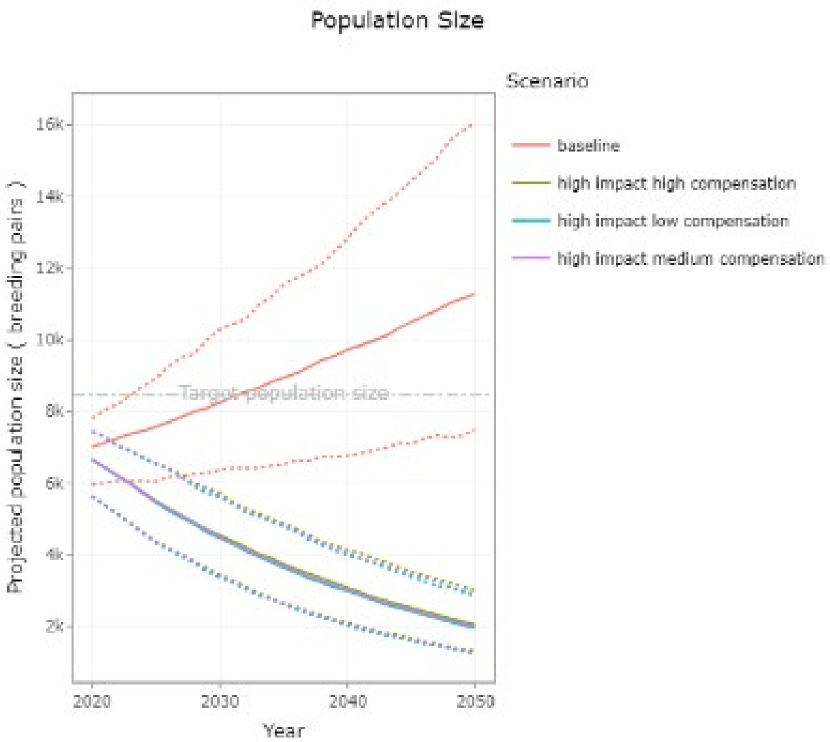
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on adult survival (Figure 18).




Low impact




Medium Impact




High Impact

**Figure 18 Projected population size of breeding gannets (pairs) at FFC SPA comparing baseline with the low, medium and high impact scenarios combined with low, medium and high compensation scenarios.**

In the low and medium impact scenarios the compensation resulted in the projected population size increasing for all compensation scenarios (Table 41). However, the high impact scenarios all showed population decline at all compensation levels. The population size increases shown are likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, for the low and medium impact scenarios the suggested levels of compensation through implementing measures that successfully reduce bycatch of gannets in the UK fishery is likely to result in increases in the gannet population at FFC SPA.

**Table 41 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario. Values of the median annual growth rate above the baseline value are shaded grey.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 1.0163 |
| Low | Low | 1.0153 |
| Low | Medium | 1.0159 |
| Low | High | 1.0168 |
| Medium | Low | 1.0110 |
| Medium | Medium | 1.0116 |
| Medium | High | 1.0125 |
| High | Low | 0.9603 |
| High | Medium | 0.9611 |
| High | High | 0.9619 |

Examination of the counterfactuals of population size showed a large difference in the CPS between the low and medium impact scenarios and high impact scenarios regardless of the level of compensation (Figure 19). However, the only combination of impact and compensation scenarios that raised the CPS above one was the low impact high compensation scenario.

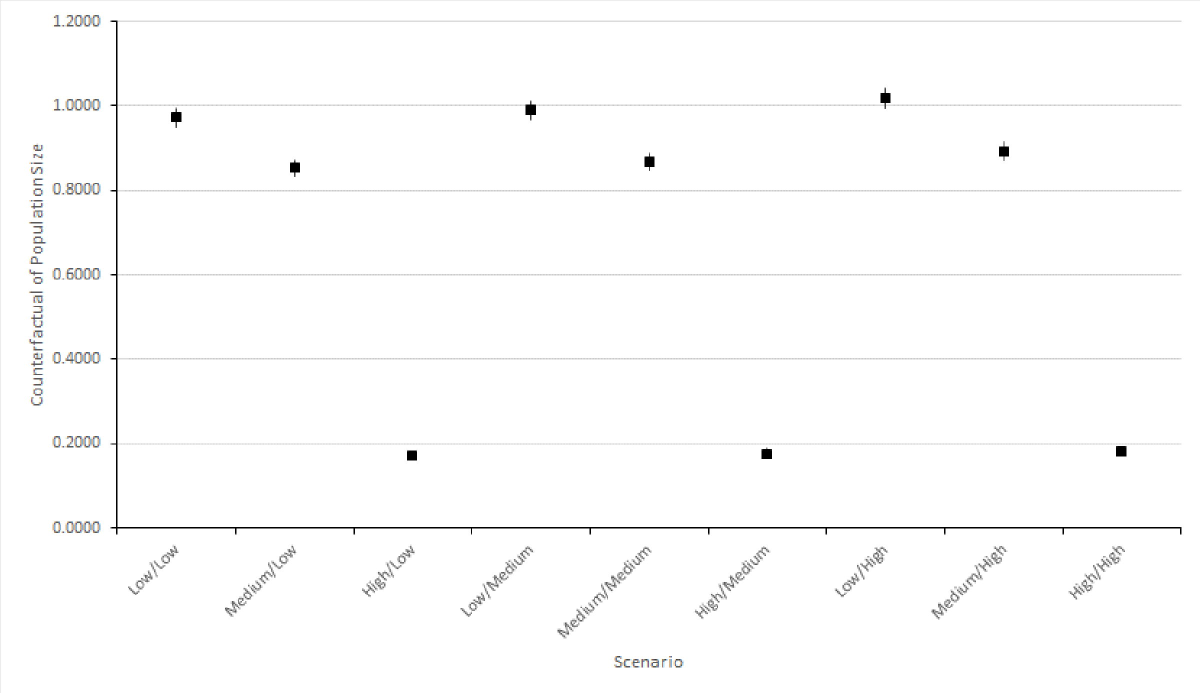



**Figure 19 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed a large difference between the low and medium impact scenarios and high impact scenarios regardless of the level of compensation (Figure 20). The only CGR with most of the range from the lower to upper 95% confidence interval above one (i.e. the population was projected to grow at a faster rate than the baseline population) was the low impact high compensation scenario.

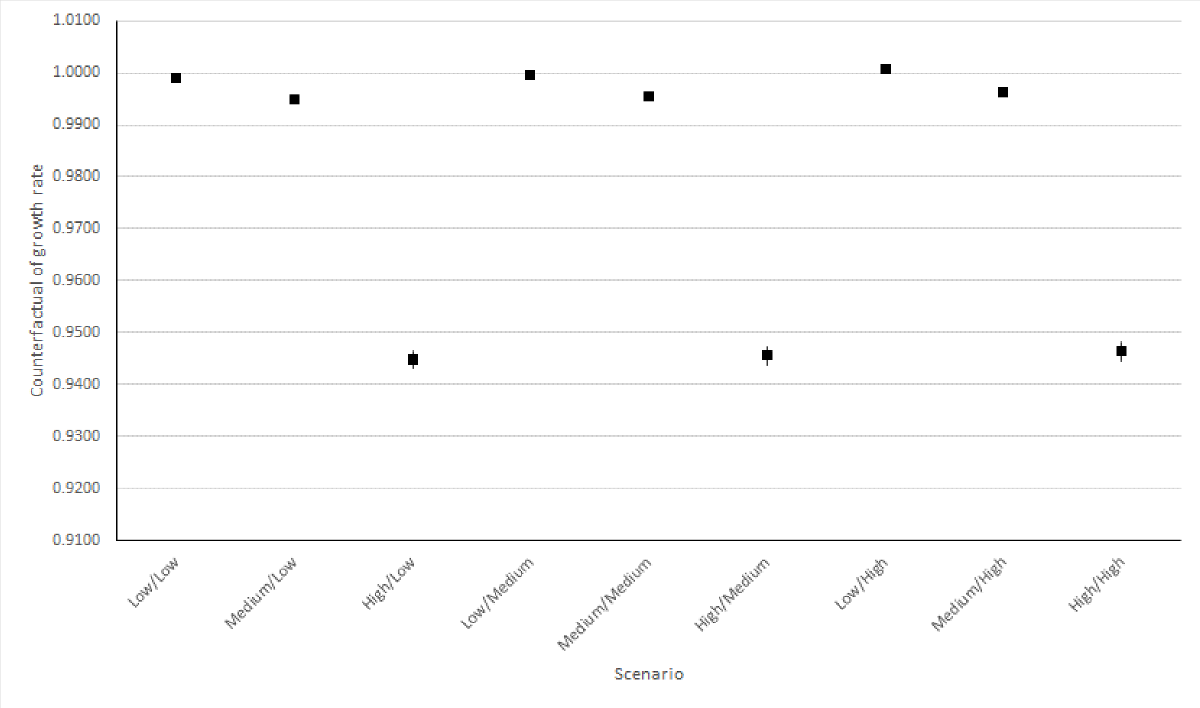



**Figure 20 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Overall, this assessment suggests that the compensation available from eliminating gannet bycatch from the UK fishery is relatively limited. The range of possible gannet bycatch levels is high, and this is a reflection of the high levels of uncertainty in the number of birds killed. However, calculation of the number of birds from the FFC SPA population that would not be killed if compensation were successful was very simple and assumed the FFC SPA was a closed population and that birds from FFC mix evenly across UK waters with those from all other UK gannet colonies and so suffer by-catch proportionately with colony size. This also assumes all the gannet bycatch is of gannets from UK colonies, which is unlikely to be true as the population of gannets in the north-east Atlantic is in reality a meta-population. Relative to the overall population size the bycatch is very small though, and it is clear from the PVA that the different impact levels tested were much more important than the levels of compensation applied. The literature suggests that bycatch is much higher in the wintering areas for UK breeding gannets, including the birds from the FFC SPA, but there was insufficient evidence to provide a quantitative assessment of the benefits of reducing or eliminating bycatch in these areas. Further research would be needed to address this evidence gap if a quantitative assessment were to be undertaken in the future.

#### *3.7.1.2 Compensation ratios*

The above PVA assessment assumed that the three compensation scenarios were based on a 1:1 ratio. The number of additional birds that would need to be prevented from being killed as bycatch from the UK fishery for ratios of 1:3 and 1:6 is also shown in Table 42. Even assuming the worst-case scenario presented by Miles et al (2020) of a by-catch of 764 gannets per annum of which 35 might be attributable to FFC SPA, only the low impact scenario at a ratio of 1:1 might be compensated for by ending by-catch. All other combinations of impact level and compensation ratios applied to these cannot be delivered by ending by-catch by UK vessels because they do not seem to catch enough gannets that might be FFC birds.

**Table 42 Number of FFC gannets prevented from being killed in bycatch required to compensate at higher compensation ratios. Cells shaded grey have a larger predicted impact than the largest possible bycatch mitigation effect in UK waters.**

| Compensation ratio | Low impact | Medium impact | High impact |
|---|---|---|---|
| 1:1 | 22 | 118 | 1,249 |
| 1:3 | 65 | 354 | 3,747 |
| 1:6 | 132 | 708 | 7,494 |

### 3.7.2 EAA 1

The FFC SPA gannet population is within the EAA 1 area. This area includes four additional SPAs designated due to their breeding gannet populations (Forth Islands, Fair Isle, Noss and Hermaness, Saxa Vord and Valla Field). These sites would likely benefit from the same bycatch mitigation measures as the FFC SPA, and this would be expected to be in proportion to their population size.

### 3.7.3 EAA 2

There are five SPAs within the EAA 2 area (Ailsa Craig, Grassholm, North Rona and Sula Sgeir, St Kilda and Sule Skerry and Sule Stack). These sites would also benefit from the same bycatch mitigation measures as FFC SPA. As such the fisheries bycatch mitigation methods suggest above would not only benefit the FFC SPA but all of the other gannet colonies in the UK SPA network. This would increase resilience to both natural and human caused pressures in the whole UK gannet meta-population.

## 3.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the method and assessment of efficacy of the two recommended compensation measures was considered for the end to chick harvesting compensation approach and the fisheries bycatch reduction approach. The summary table for the end to chick harvesting compensation method is shown in Table 43 and in the assessment is shown in Table 44. The summary table for the fisheries bycatch reduction compensation method is shown in Table 45 and in the assessment is shown in Table 46. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 47 and Table 48 (end of chick harvesting) and in Table 49 and Table 50 (fisheries bycatch reduction).

**Table 43 Assessment of confidence closure of chick harvesting method to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Sula Sgeir population growth | 2.0% | Trinder (2016) | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Sula Sgeir as a population sink | n/a | Trinder (2016) | Medium | Robust | Medium | Robust | MEDIUM | HIGH | HIGH |
| Natal dispersal rate | n/a | Trinder (2016) | Robust | Robust | Low | Limited | LOW | LOW | VERY LOW |
| Effect of ending guga hunting on UK gannet population in SPAs | n/a | Trinder (2016) | Robust | Robust | Medium | Robust | MEDIUM | LOW | LOW |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | MEDIUM |

**Table 44 Assessment of confidence closure of chick harvesting approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Number of gannet chicks harvested | 1,917 | Trinder (2016) | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Survival from fledging to first breeding | 0.258 | Based on age specific survival rates from PVA app "National" default values | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Proportion of gannets in UK SPAs | 95.9% | Stroud et al. (2016) | Medium | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 45 Assessment of confidence in fisheries bycatch reduction method to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Total additional bycatch mortality from UK fishing | 1% | Miles et al. 2020 | Robust | Robust | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Annual bycatch of gannets by UK-registered fishing vessels | 25 - 764 birds per year | Miles et al. 2020 | Robust | Medium | Medium | Robust | MEDIUM | MEDIUM | MEDIUM |
| Bycatch off Iberian Peninsula | n/a | Oliveira et al. 2015, Calado et al. 2020 | Robust | Robust | Medium | Medium | MEDIUM | MEDIUM | MEDIUM |
| Occurrence of UK gannets (including birds from FFC SPA) off the Iberian coast in winter | n/a | Fort et al. 2012 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Fisheries with gannet bycatch | See 3.6.4 | ICES 2013 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Types of mitigation available for gannet bycatch | See Table 35 | Bycatch Management Information System | Robust | Medium | Robust | Medium | MEDIUM | MEDIUM | MEDIUM |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | MEDIUM |

**Table 46 Assessment of confidence in fisheries bycatch reduction approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 5 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 1 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 13,391 pairs in 2017 | Aiken et al. (2017) | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | mean: 0.81, sd: 0.0679 | Aitken et al. (2017) | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Adult survival rate | mean: 0.919, sd: 0.042 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 0 to 1 | mean: 0.424, sd: 0.045 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.829, sd: 0.026 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 2 to 3 | mean: 0.891, sd: 0.019 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age class 3 to 4 | mean: 0.895, sd: 0.019 | PVA app "National" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 47 Narratives justifying the evidence and applicability scores for metrics used in the recommended compensation method from closure of the gannet chick harvest at Sula Sgeir.**

| Metric | Narrative |
|---|---|
| Sula Sgeir population growth | Trinder (2016) described the change in population size of the Sula Sgeir gannet colony over time using robust data collected using agreed, standardised, methods. This was considered highly applicable to the colony of interest (Sula Sgeir). Therefore, confidence was very high. |
| Sula Sgeir as a population sink | Trinder (2016) estimated that the Sula Sgier population was acting as a population sink. The type of evidence was robust as it was based on numerical modelling, and the consistency was also robust, as there are no publications casting doubt on this finding. However, since this metric was based on a model result, rather than an empirical observation, the overall evidence score was considered medium. Applicability was high as this was estimated for the colony of interest (Sula Sgeir) directly, giving an overall confidence score of high. **However, given that this is a model output, it has been reduced to medium**. |
| Natal dispersal rate | The natal dispersal rate was a model derived value with no empirical observations of this value. Other important demographic rates (particularly emigration and immigration across age classes) also remain unknown. Therefore, overall confidence was very low. |
| Effect of ending guga hunting on UK gannet population in SPAs | Trinder (2016) provides a robust approach to modelling the effects of ending the guga harvest, giving an overall evidence score of medium. However, the applicability to the FFC SPA directly is very low. Compensation at the UK network level is likely much higher than this, so an overall level of low was given, resulting in an overall confidence of low. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was low, medium or robust and the applicability was high or low, so an overall score of medium was given. **However, if this was only applied to compensation to the FFC SPA population confidence would be low. If the compensation was to the UK network, particularly the SPAs in the north of Scotland then the confidence would be high**. |

**Table 48 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from closure of the gannet chick harvest at Sula Sgeir.**

| Metric | Narrative |
|---|---|
| Number of gannet chicks harvested | The average number harvested was based on information provided to SNH (now NatureScot) and reported in Trinder (2016). While these data are now at least five years old, the maximum harvest is 2,000 chicks. So, this value is likely to be fairly accurate. |
| Survival from fledging to first breeding | The compound survival rate to age at first breeding was calculated from the age specific survival rates provided in the Seabird PVA Tool. While these are robust data from published studies, they are not from the FFC SPA itself. Therefore, the applicability was medium. |
| Proportion of gannets in UK SPAs | The proportion of gannets in UK SPAs was based on the JNCC Third SPA review so would have been an accurate assessment at that time |

| Metric | Narrative |
|---|---|
| | using empirical data from standardised counts. These data are now a little old and the gannet population in UK colonies is still increasing, so this value may have changed a little since it was reported. Hence the quality of evidence was assessed as medium. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was high or medium, so an overall score of high was given. |

**Table 49 Narratives justifying the evidence and applicability scores for metrics used in the recommended compensation method from fisheries bycatch reduction.**

| Metric | Narrative |
|---|---|
| Total additional bycatch mortality from UK fishing | Miles et al. (2020) calculated the change in mortality based on robust numerical data. The overall evidence score was robust, even with a medium score for the amount of evidence. With medium applicability the overall confidence score was high. |
| Annual bycatch of gannets by UK-registered fishing vessels | Miles et al. (2020) estimates of gannet bycatch use robust methods, but the sample sizes are not very large, and the range of the estimate is relatively large. So, the overall evidence score was medium. With no way of knowing whether the birds being taken as bycatch are from the FFC SPA the applicability was medium. Low was not chosen as it is likely that some birds do originate from the FFC SPA. This gave an overall confidence score of medium. |
| Bycatch off Iberian Peninsula | The evidence for bycatch off the Iberian Peninsula is clear, but the sample sizes were relatively small, and many reports did not provide numbers of birds killed. Thus, an overall evidence score of medium was chosen. Applicability was medium as it is reasonable to assume that FFC SPA gannets occur in the area of the bycatch generally, however it is unknown whether FFC SPA gannets are part of the bycatch. Thus, an overall score of medium was given. |
| Occurrence of UK gannets (including birds from FFC SPA) off the Iberian coast in winter | It is very clear that UK gannets winter in the Atlantic Ocean west of Spain and Portugal (and off the Atlantic coast of North Africa), based on multiple, high quality, tagging studies. So overall confidence in this metric was very high. |
| Fisheries with gannet bycatch | It is clear than gannets are caught by various fisheries, even if the number of birds may be poorly known. The applicability was medium (rather than high) as there is no direct evidence of bycatch from the FFC SPA population, though it is highly likely. Thus, an overall confidence of medium was given. |
| Types of mitigation available for gannet bycatch | While there is a wealth of information on bycatch mitigation methods, there are few species-specific data. Thus, an overall evidence score of medium was given. Applicability was medium as there is no direct evidence of FFC SPA gannets being caught in fisheries, although this seems likely. An overall score of medium was given. The absence of species-specific evidence for gannets could be used to justify reducing this score to low, but in several instance closely related species have been shown to benefit from these types of mitigation, so a medium score seems justifiable. |

| Metric | Narrative |
|---|---|
| **OVERALL CONFIDENCE SCORE** | The confidence scores were mostly medium with one high and one very high score. The applicability scores were mostly medium. Thus, an overall confidence score of medium was considered appropriate. |

**Table 50 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from fisheries bycatch reduction.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of gannets is well established and not in question. It is not variable between populations and is directly applicable to the FFC SPA population. |
| Upper constraint on productivity | Gannets lay a single egg and so productivity cannot be above this. |
| Initial population size | These data are based on recent counts using accepted and standardised methods from the colony being investigated. |
| Productivity rate per pair | These data are based on recent counts using accepted and standardised methods from the colony being investigated. |
| Adult survival rate | The survival rate evidence for all age classes is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. However, they are based on national (UK) scale data so there is some applicability to the colony being studied. A medium score was therefore given. A low score would only have been given had the data been from a colony remote from the UK or from a different sea area (e.g. western Atlantic). With robust evidence and medium applicability, the confidence score was high. |
| Age class 0 to 1 | |
| Age class 1 to 2 | |
| Age class 2 to 3 | |
| Age class 3 to 4 | |
| **OVERALL CONFIDENCE SCORE** | The confidence scores were mostly high with a few very high scores. The applicability scores were high or medium. Thus, an overall confidence score of high was considered appropriate. |

The confidence in the chick harvest closure method was medium, but this was dependent on the need to compensate for the FFC SPA directly, which would have low confidence, or to the UK SPA network, which would have a high confidence. The confidence was high in the assessment method itself. The assessment of confidence in the proposed compensation methods against the three impact scenarios also needs to be carefully considered. Ending the harvest on Sula Sgeir was predicted to add 495 adult birds to the UK gannet meta-population (Tinder 2016). If the UK network level was the aim of the compensation level, then there would be high confidence that this method would compensate for the low and medium impact scenarios, but there would be low confidence for the high impact scenario. If the compensation was for the FFC SPA directly, then the lack of information on meta-population dynamics of the UK gannet population results in a low confidence for any impact scenario.

It was apparent from the population level assessment of the fisheries bycatch scenarios that the high impact scenario could not be compensated for through elimination of the UK gannet bycatch (see 3.7.1.1), with population growth below one for all scenarios tested. For both the low and medium impact scenarios the CPS value only exceeded one for the low impact/high compensation scenario. The CGR values also only exceeded one for the low impact/high compensation scenario. The CGR values for the other low and medium impact scenarios were all larger than 0.99, but all the high impact scenario values were less than 0.95. Thus, with medium confidence in the method and high confidence in the assessment method there was low confidence that the high impact scenarios could be compensated for, medium confidence that the medium impact scenarios could be compensated for and medium confidence for the low impact scenarios combined with the low and medium compensation but high for the high compensation scenario (Table 51).

**Table 51 Assessment of confidence impact and fisheries bycatch reduction compensation scenarios.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | MEDIUM | MEDIUM | LOW |
| Medium compensation | MEDIUM | MEDIUM | LOW |
| High compensation | HIGH | MEDIUM | LOW |

## 3.9 Future monitoring and adaptive management

### 3.9.1 Harvesting of gannet chicks

In the event that the harvesting of chicks from Sula Sgeir could end, it would be necessary to maintain the ongoing monitoring of gannet colony size at this colony and at nearby colonies. This would be used to determine whether the cessation of harvesting resulted in both growth of the Sula Sgeir colony and whether the other colonies in the region also respond through increase population growth.

It is unlikely that detailed monitoring of birds at the Sula Sgeir colony would be logistically possible due to the remote location of the colony, the difficulty of accessing the island and the absence of safe accommodation on the island. Only remote monitoring would be possible, so more frequent aerial surveys of the island between national surveys could be commissioned to provide improved temporal resolution on the size of the gannet population.

It is important to note that the harvest of gannet chicks (guga) by the community at Ness on Lewis, is a cultural activity licensed by NatureScot on behalf of Scottish Government, not a commercial one. It is not clear whether ending the harvest would be acceptable to the community at Ness.

### 3.9.2 Fisheries bycatch mortality

Methods to prevent gannet bycatch from fisheries either in the UK or south-west Europe would require monitoring of the efficacy of any mitigation measures applied to the fishing technique. This would require fisheries observers on board ships to report on any changes to bycatch, at least initially. There would also need to be an assessment of proportion of vessels in the fishery deploying the mitigation measures. In addition, monitoring of UK SPA gannet colonies would need to be undertaken using the method used by Wanless et al. (2015).

#### 3.9.3 Future research

There are key gaps in knowledge of the meta-population structure of gannets in the UK. Further research on emigration and immigration rates would be useful in determining the effects of compensation measures aimed at one colony on other colonies (e.g. the ending of the guga harvest on Sula Sgeir).

There has been little reported work on species specific efficacy of bycatch mitigation methods, particularly for trawl fisheries. This could be undertaken as part of a recommended compensation measure and its adaptive management to both ensure an appropriate level of compensation occurs, and to allow future compensation methods to be more targeted.

### 3.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for gannets at FFC SPA: end harvest of chicks, encourage establishment of new colonies, and reduce bycatch in fisheries. Evidence showed little success in encouraging the establishment of new colonies for northern gannet in the North Atlantic, so this approach was not recommended. Since Furness et al. (2013) was published additional PVA analysis on the effects of the harvest of chicks at Sula Sgeir has provided new evidence on the potentially beneficial effects of ending this harvest on both the harvest colony and other gannet colonies in the north of Scotland. Taking a wider network coherence there is a high degree of confidence that ending chick harvest on Sula Sgeir would effectively compensate for the low and medium impact levels considered but no confidence that it could do so in respect of the high impact level. When focussing on the ability of the measure to deliver a benefit directly back to the gannet population at FFC SPA, there is only low confidence that ending chick harvest on Sula Sgeir could compensate for the low level of impact considered and no confidence that it could do so in respect of the medium and high impact levels considered. There was much more new evidence available on the effects of fisheries bycatch on gannet population since Furness et al. (2013). This evidence provided sufficient information to allow some assessment of compensation on the FFC SPA gannet population.

PVA suggested that the FFC SPA should be increasing and that this increase would remain likely with low and medium impact levels. However, the high impact scenario resulted in a rapid projected decline in the FFC SPA population. Three levels of compensation measure were then assessed against the three levels of impact. The mitigation of bycatch mortality on adult gannets that was tested reduced the level of adult mortality in the PVA model. The results projected that there was limited capacity to compensate for the impact scenarios through eliminating the UK bycatch mortality.

Confidence in the assessment process was high, but with limited bycatch occurring the confidence in the compensation measures were mostly low (all high impact scenarios) or medium (most low and medium impact scenarios). Only the low impact, high compensation scenario was assessed as having a high confidence in success (Table 51).

# 4 FLAMBOROUGH AND FILEY COAST SPA – BREEDING GUILLEMOT

## 4.1 Conservation status of guillemot

Common guillemot has an IUCN Red List classification of "Least Concern" and the UK population was listed in BOCC 2, 3, and 4 as amber. It is listed by the Birds Directive as a migratory species. The biogeographic population (North Atlantic) comprises birds of subspecies *aalge* and of subspecies *albionis*, and was estimated at 2,850,000 pairs, of which 890,000 pairs breed in Great Britain and 160,000 pairs in all-Ireland (Mitchell et al. 2004). The guillemots at FFC SPA are part of the subspecies *Uria aalge albionis* which has an estimated population size of 800,000 individuals (AEWA 2012), equivalent to 266,667 pairs (NE 2018). However, there is little or no molecular genetic support for the designation of this subspecies, and there is clinal variation that suggests high gene flow between colonies within and between the *albionis* and *aalge* distributions. There are also ringing data showing natal dispersal from one subspecies to breed in the range of the other (Wernham et al. 2002), which further emphasises that these two subspecies probably have no biological validity.

National surveys found a 77% increase in common guillemot breeding numbers in the UK from 1969 to 1986, and a further 31% increase from 1986 to 2000 (JNCC 2020). JNCC SCM data (JNCC 2020) show little change in breeding numbers in the UK between 2000 and 2018. In Scotland, the JNCC SCM data show a decline in breeding numbers from 2001 to 2011, followed by a recovery so that the index in 2018 was back to the same as in 1986 (JNCC 2020). In contrast, breeding numbers in England increased almost continuously from 1990 to 2018, the index in 2018 reaching four times the 1986 value. In Wales, the pattern is much as in England, with an increase to an index of 350 in 2018 (JNCC 2020).

According to Stroud et al. (2016), the SPA suite with breeding common guillemot as a designated feature has 33 qualifying sites in Great Britain, 30 in Scotland (Ailsa Craig SPA; Buchan Ness to Collieston Coast SPA; Calf of Eday SPA; Canna and Sanday SPA; Cape Wrath SPA; Copinsay SPA; East Caithness Cliffs SPA; Fair Isle SPA; Forth Islands SPA; Flannan Isles SPA; Foula SPA; Fowlsheugh SPA; Handa SPA; Hermaness, Saxa Vord and Valla Field SPA; Hoy SPA; Marwick Head SPA; Mingulay and Berneray SPA; North Caithness Cliffs SPA; North Colonsay and Western Cliffs SPA; North Rona and Sula Sgeir SPA; Noss SPA; Rousay SPA; Rum SPA; St Abb's Head to Fast Castle SPA; St Kilda SPA; Sule Skerry and Sule Stack SPA; Sumburgh Head SPA; The Shiant Isles SPA; Troup, Pennan and Lion's Heads SPA; West Westray SPA), two in England (Farne Islands SPA; Flamborough and Filey Coast SPA) and one in Wales (Skokholm, Skomer and Middleholm SPA, now known as Skomer, Skokholm and seas off Pembrokeshire SPA). The SPAs in Great Britain were estimated to hold about 70% of the Great Britain breeding population of common guillemots present in 2000 (Stroud et al. 2016). One site in Northern Island also qualifies (Rathlin Island). Breeding guillemots have been classified as in Unfavourable Conservation Status at most SPA sites in north Scotland (especially in Orkney and Shetland). Numbers have declined at those sites considerably more than they have increased at sites in south Scotland, England, Wales and Northern Ireland. However, this feature is classified as Favourable Maintained at Fowlsheugh, Buchan Ness to Collieston, Forth Islands, St Abb's Head to Fast Castle and numbers at all those sites are higher in the most recent census than at designation. Numbers at Farne Islands are marginally lower now than at designation (64,042 individuals in 2019 versus 65,751 at designation), whereas at FFC SPA numbers have increased (84,647 individuals in 2017 versus 62,100 at designation).

Apart from the limited marine extensions to SPAs for loafing birds, no sites were listed in the 3rd UKSPA review as designated for guillemots at sea (Stroud et al. 2016). Since then, three marine sites designated on 3 December 2020 include guillemot as a feature. Guillemot is a breeding season feature at Outer Firth of Forth and St Andrews Bay Complex SPA, Seas off St Kilda SPA and Seas off Foula SPA, and is also a nonbreeding season feature at Seas off Foula SPA. In addition, guillemot is a breeding feature of the Northumberland Marine SPA.

Historical published counts of guillemots (numbers of individuals on land) at Flamborough Head and Bempton Cliffs SPA include 12,570 in 1969, 32,578 in 1987, 47,215 in 2000, 59,817 in 2008, and 84,647 in 2017 according to Lloyd et al. (2019). The counts from 1987, 2000, 2008 and 2017 are also listed in JNCC (2020). In the larger area of the whole FFC SPA there were 90,861 guillemots in 2017 (Lloyd et al. 2019), so the majority of this species are to be found within the original Flamborough Head and Bempton Cliffs SPA, with an additional 6,214 individuals (an additional 7%) in 2017 in the part of FFC SPA that is outwith Flamborough Head and Bempton Cliffs SPA boundaries. These data show a clear and strong increase in numbers of guillemots (Figure 21).

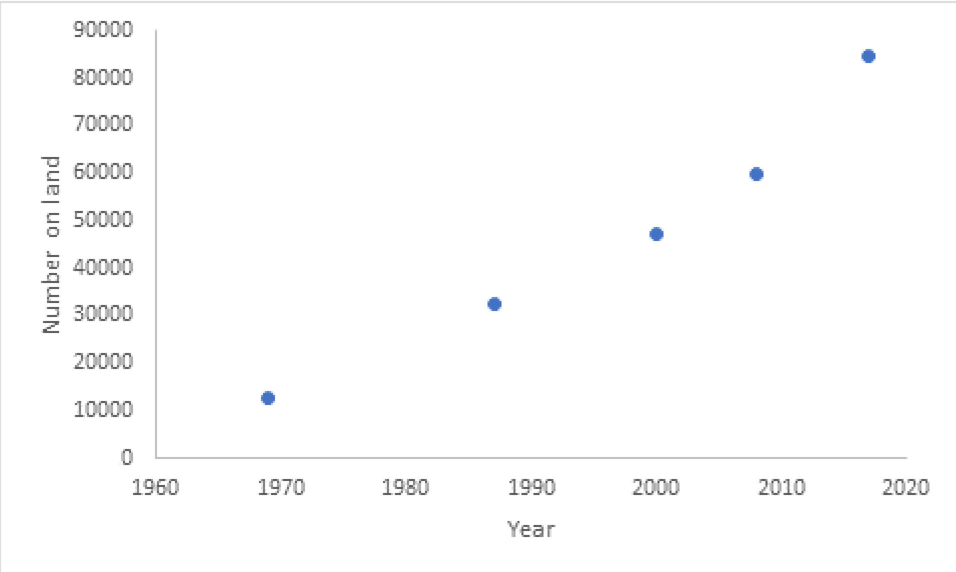



**Figure 21 Numbers of guillemots counted on land at Flamborough Head and Bempton Cliffs SPA (data from Lloyd et al. 2019).**

### 4.2 Citation population size

The FFC SPA citation (dated August 2018) states that the site qualifies under Article 4.2 of the Birds Directive by supporting over 1% of the biogeographical populations of four regularly occurring migratory species: kittiwake, gannet, guillemot, and razorbill. The site held 41,607 pairs of guillemots in 2008-2011, representing 15.6% of the subspecies *Uria aalge albionis* (but note reservations about the validity of this subspecies and evidence indicating that there is no meaningful subspecies boundary between *aalge* and *albionis* expressed in 2.3.1) This estimate of the number of pairs is derived from the mean count of individual guillemots on land in 2008-2011 (62,100 individuals) multiplied by a correction factor of 0.67 to translate to breeding pairs. The estimate of the population of *Uria aalge albionis* is from AEWA (2012): 800,000 individuals, translated to pairs by dividing this total by 3 (NE 2018), giving 266,667 pairs.

### 4.3 Conservation objectives

The site's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site.

More detailed conservation objectives have since been added online, last updated 13 March 2020 (Natural England 2020). For guillemot at FFC SPA these are:

- Maintain the size of the breeding population at a level which is above 41,607 breeding pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Maintain safe passage of birds moving between nesting and feeding areas;
- Restrict the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding);
- Maintain the distribution, abundance and availability of key food and prey items (e.g. Sandeel, herring, sprat) at preferred sizes;
- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;

- Maintain water quality and specifically mean winter dissolved inorganic nitrogen (DIN) at a concentration equating to High Ecological Status (specifically mean winter DIN is < 12 μM for coastal waters), avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for FFC pSPA was published in February 2015 (NE 2015). That identified public access/disturbance as a threat to guillemots and identified prevention of disturbance as a responsibility of East Riding of Yorkshire Council, Natural England, RSPB, Scarborough Borough Council, Yorkshire Wildlife Trust, and Flamborough Management Scheme. No other threats or pressures affecting guillemots at FFC SPA were specifically identified as requiring management in the SIP.

#### 4.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are two main sources of impact on guillemots from offshore wind farm development: displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of guillemots from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. That impact on survival may be a carry-over effect on reduced winter survival as birds are in poorer condition at the end of the breeding season than would have been the case in the absence of the wind farm. There is a week relationship between the condition (body mass) of guillemots at the end of the breeding season and their subsequent overwinter survival (Daunt et al. 2020).

The maintenance of the population of each of the qualifying features could be affected indirectly through impact to energy budgets from displacement and barrier effects.

### 4.4 Location of compensation

Cook et al. (2011) defined two Ecological Assessment Areas for guillemot (Figure 22). The FFC SPA occurs within EAA2. Consequently, the hierarchy of the locations of compensation are:

1. FFC SPA;
2. EAA 2; and

3. EAA 1.

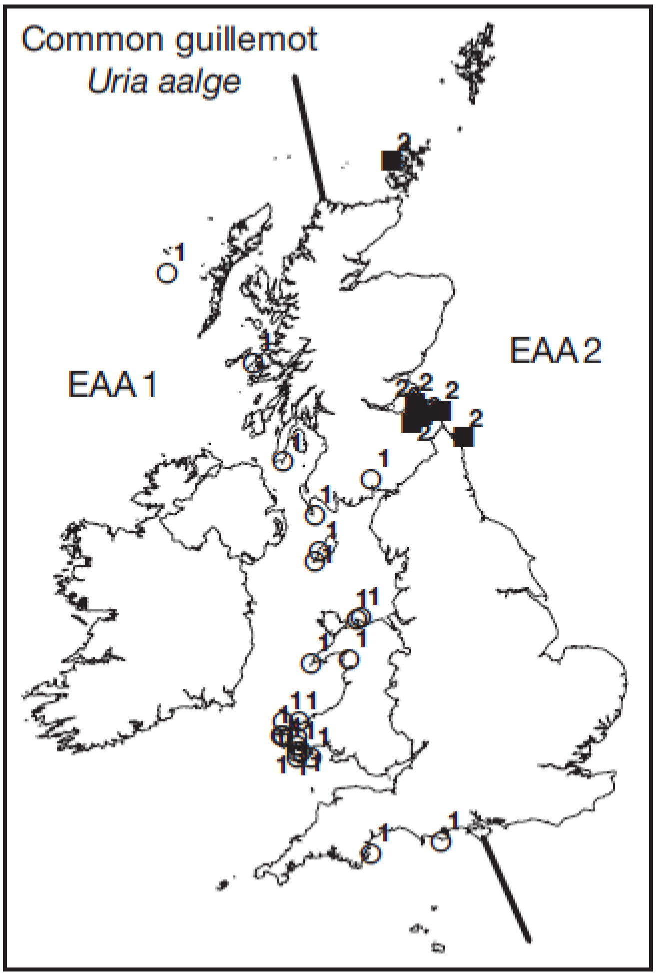



**Figure 22 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for guillemot by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

## 4.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding guillemots were developed based on the four potential compensation measures reviewed by Furness et al. (2013). The four potential measures listed were:

1. Closure of sandeel and sprat fisheries in all UK waters;
2. Closure of sandeel and sprat fisheries in guillemot wintering areas;
3. Rat eradication; and
4. Prevent oil spills.

Only the last of these potential measures was considered highly likely to be effective with high confidence in that assessment based on evidence. However, it was recognised that strong efforts are already made to prevent oil spills, so that this was unlikely to be a practical option. While there was strong evidence that closure of sandeel and sprat fisheries would benefit related seabird species, there was only limited evidence in this regard specifically for common guillemot. There was a lack of clear evidence that this species would benefit from eradication of rats, but that was considered a highly practical measure if new evidence indicated this to be an effective measure at some colonies. Guillemot is one of the most intensively studied of all seabirds, and so the evidence base on this species has increased considerably. The key biological questions for compensation measures for guillemots at FFC SPA are provided in Table 52.

**Table 52 Key Biological Questions in assessing the potential for compensation of breeding guillemot at FFC SPA.**

| No. | Key Biological question |
|---|---|
| **Closure of sandeel or sprat fisheries** | |
| 1 | Are guillemots sensitive to prey availability in the vicinity of their colony? |
| 2 | Can guillemots buffer against declines in fish stocks? |
| 3 | Can guillemots switch prey species when availability of one species declines? |
| 4 | Does sandeel stock biomass affect guillemot productivity? |
| 4.1 | At FFC SPA? |
| 4.2 | At EAA 2? |
| 4.3 | At EAA 1? |
| 5 | Does sandeel stock biomass affect adult survival of guillemots? |
| 5.1 | At FFC SPA? |
| 5.2 | At EAA 2 |
| 5.3 | At EAA 1 |
| 6 | Do adult guillemots forage within areas subject to a high level of sandeel mortality from fisheries? |
| 6.1 | At FFC SPA? |
| 6.2 | At EAA 2? |
| 6.3 | At EAA 1? |
| 7 | Would management, or closure, of sandeel fisheries within the foraging areas of adult guillemot result in greater availability of forage fish for adult guillemots? |
| 7.1 | At FFC SPA? |
| 7.2 | At EAA 2? |
| 7.3 | At EAA 1? |
| **Eradication of rats and other invasive mammal predators** | |
| 1 | Is there evidence that eradication of rats from guillemot colonies increases the population size? |
| 1.1 | At FFC SPA? |
| 1.2 | At EAA 2? |
| 1.3 | At EAA 1? |
| 2 | Is there evidence of rats on offshore islands that include breeding guillemot SPAs? |
| 2.1 | At FFC SPA? |
| 2.2 | At EAA 2? |

| No. | Key Biological question |
|---|---|
| 2.3 | At EAA 1? |

## 4.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 4.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

### 4.6.1 Closure of sandeel or sprat fisheries

In a study of common guillemots and razorbills, Chimienti et al. (2017) showed that razorbills made only pelagic dives whereas common guillemots made both benthic and pelagic dives. In another study of common guillemots and razorbills, Hentati-Sundberg et al. (2020) highlight the importance of maintaining sufficient prey densities in the vicinity of the colony, suggesting that fine-scale spatial fisheries management is necessary to maintain high seabird breeding success. They also emphasised that there can be differences in this regard between similar species at the same location. Despite foraging on the same prey, razorbills could breed successfully at lower prey densities than guillemots but needed higher densities for self-maintenance, emphasizing the importance of considering species-specific traits when determining sustainable forage fish densities for top predators. They concluded that in their study case, densities of forage fish corresponding to the current fisheries management target $B_{MSY}$ were sufficient for successful breeding, and that the fisheries management target for conserving seabirds proposed by Cury et al. (2011), 1/3 of historical maximum prey biomass (B1/3), was also sufficient. Montevecchi et al. (2019) agreed that forage fish availability is a key determinant of guillemot breeding success, survival and population change, but found that common guillemots at Newfoundland were able to buffer against declines in forage fish abundance (capelin in their case) over the range of fish abundances seen in that locality. Although guillemots worked harder as capelin stock declined, resulting in lower chick mass at fledging and lower body mass of adults, breeding numbers increased, and that was attributed by Montevecchi et al. (2019) to amelioration of anthropogenic risk factors resulting in increased survival of birds in winter (e.g. less hunting of guillemots for food and reduced bycatch in fishing nets). Nevertheless, Montevecchi et al. (2019) concluded that the reduction in capelin abundance had taken the common guillemots very close to their limit of buffering capacity. Buffering capacity was also demonstrated by Kadin et al. (2016) who found that guillemots adjusted their foraging effort to compensate, but only within limits, for reduced quality of prey brought to chicks. However, limits to buffering and a cost of such responses to reduced food abundance or quality can be seen at the physiological level. Storey et al. (2017) showed that guillemot body mass and chick-feeding rates were higher in good years than in poor years and heavier guillemots were more likely to fledge a chick than lighter birds. Stress hormone levels (corticosterone) were highest in adult guillemots in intermediate years (moderate forage fish availability) when foraging effort increased to rear surviving chicks but were lower in bad years (low forage fish availability) when extra foraging effort would have been unable to compensate for low prey. Wanless et al. (2018) showed that guillemots at the Isle of May were better able to switch from a diet of sandeels to a diet of sprats than were other seabird species and Smout et al. (2013) showed that chicks remained adequately provisioned despite prey switching.

Merkel et al. (2020) used geolocation tracking data from common guillemots to show that they use fixed and individual-specific migration strategies, i.e. individuals go to the same wintering areas in successive years, showing fidelity to geographical sites. They point out that while this behaviour allows individual guillemots to become familiar with their chosen winter home, it represents a constraint in the context of rapidly changing environments. Guillemots may not be able to adjust their migration strategy as conditions change, for example as a consequence of depletion of forage fish stocks in their chosen wintering area or impacts of climate change on forage fish distribution.

Using synoptic marine bird and hydroacoustic surveys during winter, Schaefer et al. (2020) showed that wintering common guillemots tended to distribute themselves above aggregations of forage fish; 40% of marine birds, including common guillemots, were within 150 m of a forage fish school, whereas only 20% of forage fish schools were associated with birds. The authors concluded that their data show the importance of forage fish aggregations as the main driver of guillemot spatial aggregations in winter.

There is evidence that guillemot mortality peaks during winter, and therefore that winter may represent a bottleneck of high energy demand and low availability of food, as well as a time of exposure to extreme weather (Louzao et al. 2019). Burke and Montevecchi (2018) converted data from dive-immersion geolocator tags deployed on common guillemots at Newfoundland into energy budgets in order to assess how they cope with cold exposure, short daylength and low prey availability in winter. Their study highlights late winter as an extremely challenging phase in the annual cycle of guillemots in Newfoundland and provides critical insights into the behavioural mechanisms underlying their winter survival. That study may be a very useful comparison for data from guillemots wintering in UK waters in order to assess whether guillemots in UK waters are close to their energy limits or have a relatively relaxed energy budget compared to guillemots in the much colder waters of Newfoundland. Using Time Depth Recorder (TDR) tags that record diving behaviour, Dunn et al. (2019) compared post-breeding and mid-winter diving activity budgets of guillemots, razorbills and puffins. Dunn et al. (2020) estimated the year-round activity budgets, energy expenditure, location, colony attendance and foraging behaviour for a sample of breeding adults from a population of common guillemots. They concluded that despite the potential constraints of reduced daylength and low sea surface temperatures in winter, guillemots managed their energy expenditure throughout the year, and were not showing a strong peak of energy expenditure in winter. Indeed, energy expenditure was highest immediately before and during the breeding season, driven by a combination of high thermoregulatory costs, diving activity, colony attendance and associated flight. Guillemots also exhibited partial colony attendance outside the breeding season, which they inferred must be supported by local resources (i.e. forage fish abundance), and which has been advancing to earlier dates as a consequence of warming climate (Merkel et al. 2019). Sinclair (2018) reported on the use of time-lapse cameras to monitor colony attendance by guillemots in Shetland outside the breeding season and finding a significant effect of wind speed on colony attendance in winter.

Piatt et al. (2020) reported on a ‘wreck’ of guillemots that resulted in the death of at least 60,000 birds in the North Pacific in 2015-2016. That particular wreck seems to have been caused by abnormally high water temperatures that resulted in breeding failure at 22 colonies in the region in 2015 followed by deaths of adults. The wreck was considered to be caused by high sea temperature leading to diminished forage fish stocks, so that guillemots starved. This abnormal case indicates the importance of sustained stocks of forage fish for guillemots, and a probable increase in vulnerability resulting from climate warming. It is reminiscent of the 85% decrease of adult common guillemot breeding numbers in the Barents Sea in the mid-1980s when the capelin stock there was reduced to very low abundance by fishing pressures and top-down impacts of predatory fish on capelin (Anker-Nilssen et al. 2000). Sadykova et al. (2020) modelled effects of warming sea temperatures on guillemots and other marine predators feeding on herring and sandeels in west European waters and concluded that northward shifts of prey fish caused by warming sea temperatures may reduce spatial overlap between breeding guillemots and their prey. However, they identified regions where overlap between guillemots and prey fish may increase, and there may be new prey fish species that move into southern areas that become less favourable for herring and sandeel, so the long-term consequence of warming sea temperatures for guillemots and their prey remain uncertain.

Although much emphasis of geolocator studies of auks has been on their wintering areas, the moulting locations are also important, and because auks become flightless during moult in the post-breeding period, these areas must contain high and stable supplies of forage fish to support the birds through moult. Glew et al. (2018) used a combination of stable isotopes and light-based geolocation data to identify and compare moulting areas used by guillemots, razorbills, and puffins from the Isle of May. Harris et al. (2015) reported on one individual guillemot from the Isle of May that moved 3,000 km northeast from the Isle of May to moult in the Barents Sea. More recently, several further examples of this behaviour have been seen in guillemots equipped with geolocators at Foula, Fair Isle and Canna (Buckingham et al. in review), so this behaviour is not unique, despite the particularly high energy cost of flight in this species.

[redacted]

[redacted]

**Figure 23** [redacted].

*4.6.1.1 Answers to the key biological questions (4.5).*

The answers to the key biological question in relation to compensation through closure of sandeel or sprat fisheries are shown in Table 53.

**Table 53 Answers to Key Biological Questions in assessing the potential for compensation through closure of sandeel or sprat fisheries.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Are guillemots sensitive to prey availability in the vicinity of their colony? | **Yes.** Hentati-Sundberg et al. (2020) highlighted the importance of maintaining sufficient prey densities in the vicinity of the colony, suggesting that fine-scale spatial fisheries management is necessary to maintain high seabird breeding success. Montevecchi et al. (2019) showed that forage fish availability is a key determinant of guillemot breeding success, survival and population change, but found that common guillemots at Newfoundland were able to buffer against declines in forage fish abundance (capelin in their case) over the range of fish abundances seen in that locality. |
| 2 | Can guillemots buffer against declines in fish stocks? | **Yes.** Montevecchi et al. (2019) concluded that the reduction in capelin abundance had taken the common guillemots very close to their limit of buffering capacity. Buffering capacity was also demonstrated by Kadin et al. (2016) who found that guillemots adjusted their foraging effort to compensate, but only within limits, for reduced quality of prey brought to chicks. However, limits to buffering and a cost of such responses to reduced food abundance or quality can be seen at the physiological level. Storey et al. (2017) showed that guillemot body mass and chick-feeding rates were higher in good years than in poor years and heavier guillemots were more likely to fledge a chick than lighter birds. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 3 | Can guillemots switch prey species when availability of one species declines? | **Yes.** Wanless et al. (2018) showed that guillemots at the Isle of May were better able to switch from a diet of sandeels to a diet of sprats than were other seabird species and Smout et al. (2013) showed that chicks remained adequately provisioned despite prey switching. |
| 4 | Does sandeel stock biomass affect guillemot productivity? | |
| 4.1 | At FFC SPA? | **Maybe.** Furness and Tasker (2000) reported that the influence of sandeel stock biomass on guillemot breeding success is relatively weak. This presumably applies to all guillemot colonies in the North Sea. Breeding success of guillemot is monitored every year by RSPB at the FFC SPA. The productivity of birds within monitored plots has been declining since 2009 (Figure 24). From 2009 to 2015 productivity varied between 0.7 and 0.8 chicks per pair. However, since 2016 productivity has mostly been between 0.6 and 0.7. The low breeding success of guillemots from 2016-2019 coincides with ICES Area 1r sandeel stock falling to the lowest stock biomass reported over the past 40 years and in several recent years below Blim (ICES 2020). This is evidence that sandeel abundance is likely to be limiting the productivity of guillemots at FFC SPA. |
| 4.2 | At EAA 2? | **Maybe.** Furness and Tasker (2000) reported that the influence of sandeel stock biomass on guillemot breeding success is relatively weak. This presumably applies to all guillemot colonies in the North Sea. |
| 4.3 | At EAA 1? | **Unlikely**. There is no sandeel fishery in the seas around EAA 1, so sandeel stock biomass is unlikely to be as variable as the North Sea (EAA 2). Sandeel are still an important part of guillemot diet from breeding colonies on the west coast of the UK (Anderson et al. 2013). |
| 5 | Does sandeel stock biomass affect adult survival of guillemots? | |
| 5.1 | At FFC SPA? | [redacted] |
| 5.2 | At EAA 2? | [redacted] |
| 5.3 | At EAA 1? | **Unlikely**. There is no sandeel fishery in the seas around EAA 1, so sandeel stock biomass is unlikely to be as variable as the North Sea (EAA 2). |
| 6 | Do adult guillemots forage within areas subject to a high level of sandeel mortality from fisheries? | |
| 6.1 | At FFC SPA? | **Yes.** Most of the Danish fishing effort on sandeels in UK waters is targeted at grounds on the western edge of Dogger Bank. The main sandeel fishing area in UK waters is around 100 km from FFC SPA. This is within the maximum foraging range of guillemots from FFC SPA (135 km excluding tracking from Fair Isle, Woodward et al. 2019) but beyond the mean of the maximum foraging range (55.5 km excluding tracking data from Fair Isle, Woodward et al. 2019). So, there is likely to be some connectivity between the FFC SPA guillemot population and the area where high levels of sandeel fishing occur, but the majority of birds likely forage much closer to the colony. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 6.2 | At EAA 2? | **Yes.** Most of the North Sea is either currently heavily fished for sandeels, or has been in the past. The sandeel box off the east coast of Scotland provides some protection for sandeel stocks for guillemots, but this stock is currently fished. |
| 6.3 | At EAA 1? | **No.** There is no sandeel fishery in the seas around EAA 1. |
| 7 | Would management, or closure, of sandeel fisheries within the foraging areas of adult guillemot result in greater availability of forage fish for adult guillemots? | |
| 7.1 | At FFC SPA? | **Yes.** Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. Evidence from the closure of the sandeel fishery off the east coast of Scotland indicates that populations of guillemots are likely to increase following closure of the fishery. |
| 7.2 | At EAA 2? | **Yes.** Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. Evidence from the closure of the sandeel fishery off the east coast of Scotland indicates that populations of guillemots are likely to increase following closure of the fishery. |
| 7.3 | At EAA 1? | **No.** There is no sandeel fishery in the seas around EAA 1. |

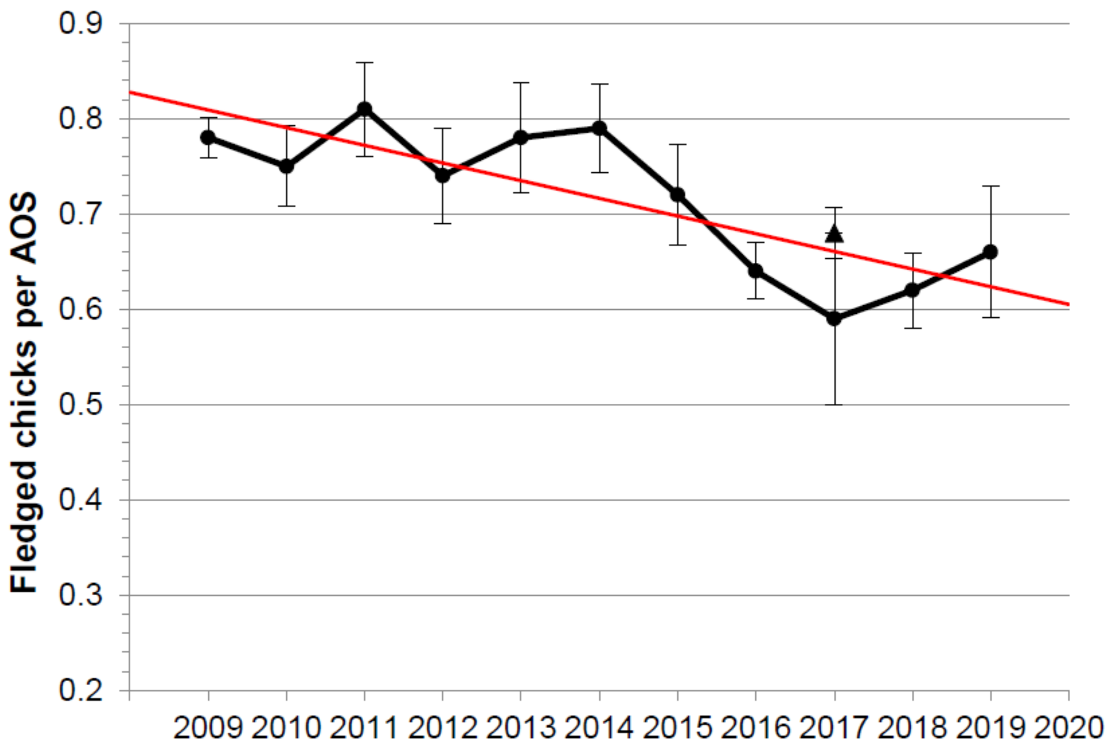



**Figure 24 RSPB data on breeding success of guillemot at Flamborough and Bempton from 2009 to 2019 (from Lloyd et al. 2019).**

### 4.6.2 Eradication of rats and other invasive mammal predators

Eradication of rats from Lundy resulted in guillemot breeding numbers increasing from 2,348 to 6,198 individuals and showing an increase in breeding distribution of this species on the island into areas that would have been accessible to rats, so the increase is attributed to the removal of the pressure of predation by rats (Booker et al. 2019). However, Luxmoore et al. (2019) found no evidence of any increase in guillemot breeding numbers at Canna as a consequence of eradication of rats from that island and suggested that guillemot breeding numbers there are probably constrained by some other factors. Clearly the Lundy case study provides strong evidence that eradication of rats can benefit guillemots in some colonies, but this may depend on the amount of boulder and cave nesting habitat (rather than cliff ledges) and whether or not guillemot numbers can increase into such habitat or are constrained by other factors such as food availability.

#### 4.6.2.1 *Answers to the key biological questions (4.5).*

The answers to the key biological question in relation to compensation through eradication of rats and other invasive mammal predators are shown in Table 54.

**Table 54 Answers to Key Biological Questions in assessing the potential for compensation through eradication of rats and other invasive mammal predators.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that eradication of rats from guillemot colonies increases the population size? | |
| 1.1 | At FFC SPA? | **No**. There is no evidence that rats are limiting the FFC SPA, so rat eradication would not result in increases in colony size. |
| 1.2 | At EAA 2? | **No**. There is no evidence from colonies in EAA 2. |
| 1.3 | At EAA 1? | **Yes**. Booker et al. (2019) provided evidence that rats constrained the size of the colony on Lundy. However, Luxmoore et al. (2019) found that the guillemot population on Canna did not change after rats were eradicated. It is important to note that rats may not be predating on guillemots on an island, but their presence can constrain the availability of nest sites to rat free locations on the island. |
| 2 | Is there evidence of rats on offshore islands that include breeding guillemot SPAs? | |
| 2.1 | At FFC SPA? | **No**. There is no evidence of rats occurring in the FFC SPA guillemot colony. |
| 2.2 | At EAA 2? | **Yes**. Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats were present. The islands with rats present and seabird colonies including guillemots in EAA 2 were:<br>Rousay, Orkney;<br>Unst, Shetland;<br>Inchkeith, Firth of Forth;<br>Hoy, Orkney;<br>Flotta, Orkney; and<br>Stronsay, Orkney. |
| 2.3 | At EAA 1? | **Yes**. Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| | | were present. The islands with rats present and seabird colonies including guillemots in EAA 1 were:<br>Garbh Eilean and Eilean an Taighe, Shiants;<br>Rathlin Island, Northern Ireland;<br>Colonsay and Oronsay, Argyll and Bute;<br>Tiree, Argyll and Bute; and<br>Herm, Channel Islands.<br>Black rats occurred on the Shiant Islands and Herm, while the remaining islands have brown rats. |

### 4.7 Population level assessment

#### 4.7.1 Flamborough and Filey Coast SPA

Population modelling required suitable input parameters to assess both the impacts on the population and the effects of proposed compensation scenarios. Closure of sandeel and sprat fisheries were predicted to have positive effects on adult survival, so this impact parameter was important. There were no data available for the survival of guillemots from the FFC SPA, so the rates used were from the Isle of May as the nearest colony in the North Sea basin with available survival data. The productivity rate (chicks per pair) was based on 19 years of data collected at FFC between 1991 and 2019 available from the Seabird Monitoring Programme database. There was relatively little variation in the productivity across this period (Figure 25), with very low productivity in 1997. The overall mean across the whole period was 0.7137 (1SD = 0.1094), which was used as the input parameter in the PVA.

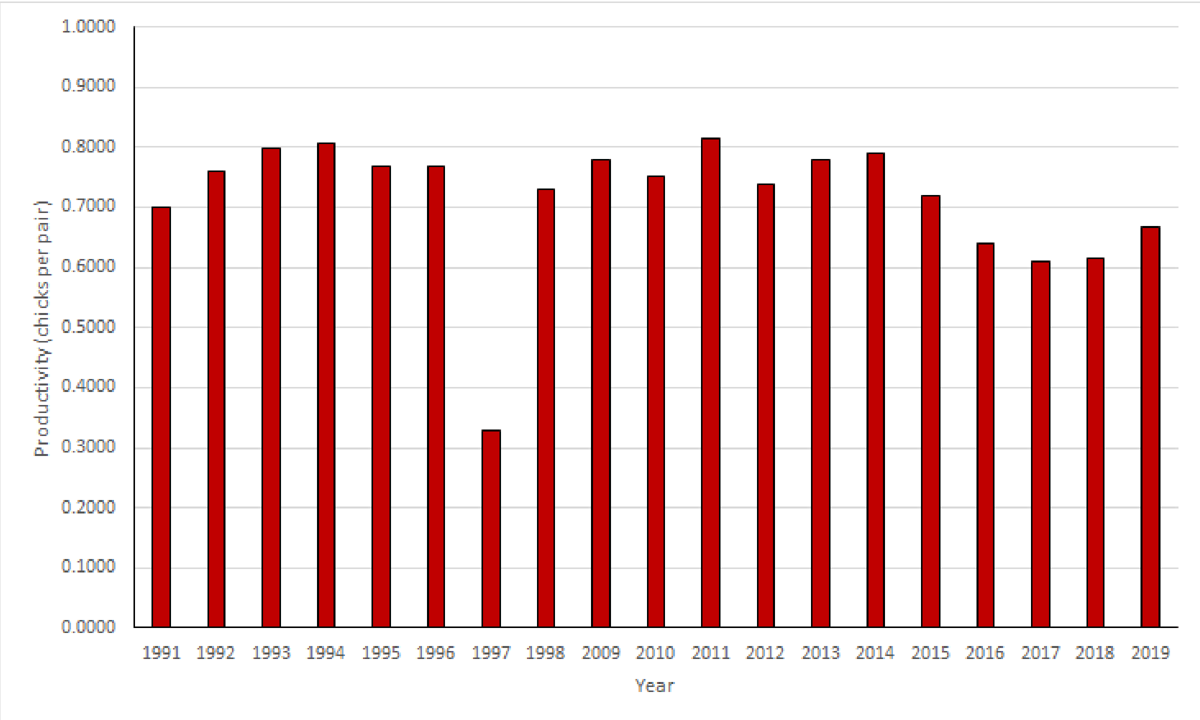



**Figure 25 Productivity of guillemots at FFC SPA between 1991 and 2019 (data from SMP database). Note that the x-axis is categorical.**

The population growth rate for the guillemot population at FFC SPA was then estimated based on the change in population size between the Seabird Colony Register count in 1988 (32,578 individuals), and the most recent count in 2017 (121,754 individuals; Aitken et al. 2017). This gave a population growth rate of 1.0465.

Using the productivity rate from the SMP database the annual adult survival rate was varied using the equation from Figure 23 ($y=0.0542\ln(x) + 0.2158$) across the sandeel stock at 200,000, 400,000, 600,000 and 800,000 tons and the median annual growth rate estimated using the PVA (Table 55). This showed the closest adult survival rate needed to result in a growth rate similar to that observed from the FFC SPA population. The adult survival rate of 0.9525 was therefore used as the input parameter for the baseline population. It is important to note that the population model assumed that the population was closed, while in reality there may be immigration of adults into the population in order to maintain the growth observed.

**Table 55 Predicted median annual growth rate of the FFC SPA guillemot population across a range of adult survival rates, assuming a closed population with demographic parameters typical for UK colonies.**

| Sandeel stock | Adult survival | Median Annual Growth Rate |
|---|---|---|
| 800,000 | 0.9525 | 1.0445 |
| 600,000 | 0.9369 | 1.0339 |
| 400,000 | 0.9149 | 1.0191 |
| 200,000 | 0.8774 | 0.9951 |

All the population level assessments for FFC SPA guillemot were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult mortality rate. For a population size of 83,214 breeding individuals and an adult survival rate of 0.9525 a 1% increase in baseline mortality would be 39.5 additional birds being killed per annum.

The medium impact scenario was based on a pro-rata impact for an additional 7GW of capacity for Round 4 offshore wind farm development. The high impact scenario was based on a pro-rata impact from an additional 74GW of capacity for a 2050 net zero target. This resulted in medium impact scenario of 817 birds and a high impact scenario of 8,635 birds (Table 56).

**Table 56 Values for low, medium and high impact scenarios for guillemots at FFC SPA.**

| Impact scenario | Low | Medium | High |
|---|---|---|---|
| Additional mortality (birds) | 39.5 | 817 | 8,635 |
| Additional mortality (rate) | 1% | 20.7% | 218.5% |

The PVA was parameterised using the values in Table 57.

**Table 57 PVA input parameters baseline vs impact scenarios.**

| Model parameter | Parameter values | | | Source |
|---|---|---|---|---|
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 6 | | | PVA app default |
| upper constraint on productivity | 1 chick per pair | | | PVA app default |
| Initial population size | 83,214 individuals in 2008 | | | SACO |
| Productivity rate per pair | mean: 0.7137, sd: 0.1094 | | | PVA app "National" default value |
| Adult survival rate | mean: 0.9525, sd: 0.058 | | | Value needed for observed population growth rate |
| Age class 0 to 1 | mean: 0.56, sd: 0.058 | | | PVA app "Isle of May" default value |
| Age class 1 to 2 | mean: 0.792, sd: 0.152 | | | PVA app "Isle of May" default value |
| Age class 2 to 3 | mean: 0.917, sd: 0.098 | | | PVA app "Isle of May" default value |
| Age class 3 to 4 | mean: 0.938, sd: 0.107 | | | PVA app "Isle of May" default value |
| Age class 4 to 5 | mean: 0.924, sd: 0.058 | | | PVA app "Isle of May" default value |
| Age class 5 to 6 | mean: 0.924, sd: 0.058 | | | PVA app "Isle of May" default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Impact on adult survival rate | Low | Medium | High | Calculated as above |
| | 0.000474937 | 0.009818059 | 0.1037686 | |
| **Sandeel fishery closure scenarios** | | | | |

| Model parameter | Parameter values | | | Source |
|---|---|---|---|---|
| Impact on adult survival rate | Low | Medium | High | Calculation described in 4.7.2 below |
| | 0.0826 | 0.0613 | 0.0189 | |

The baseline population projection was compared with the three impact scenarios (Table 56). The model projected that the baseline, low impact and medium impact scenario populations would all increase, while the high impact scenario population would decrease (Figure 26).

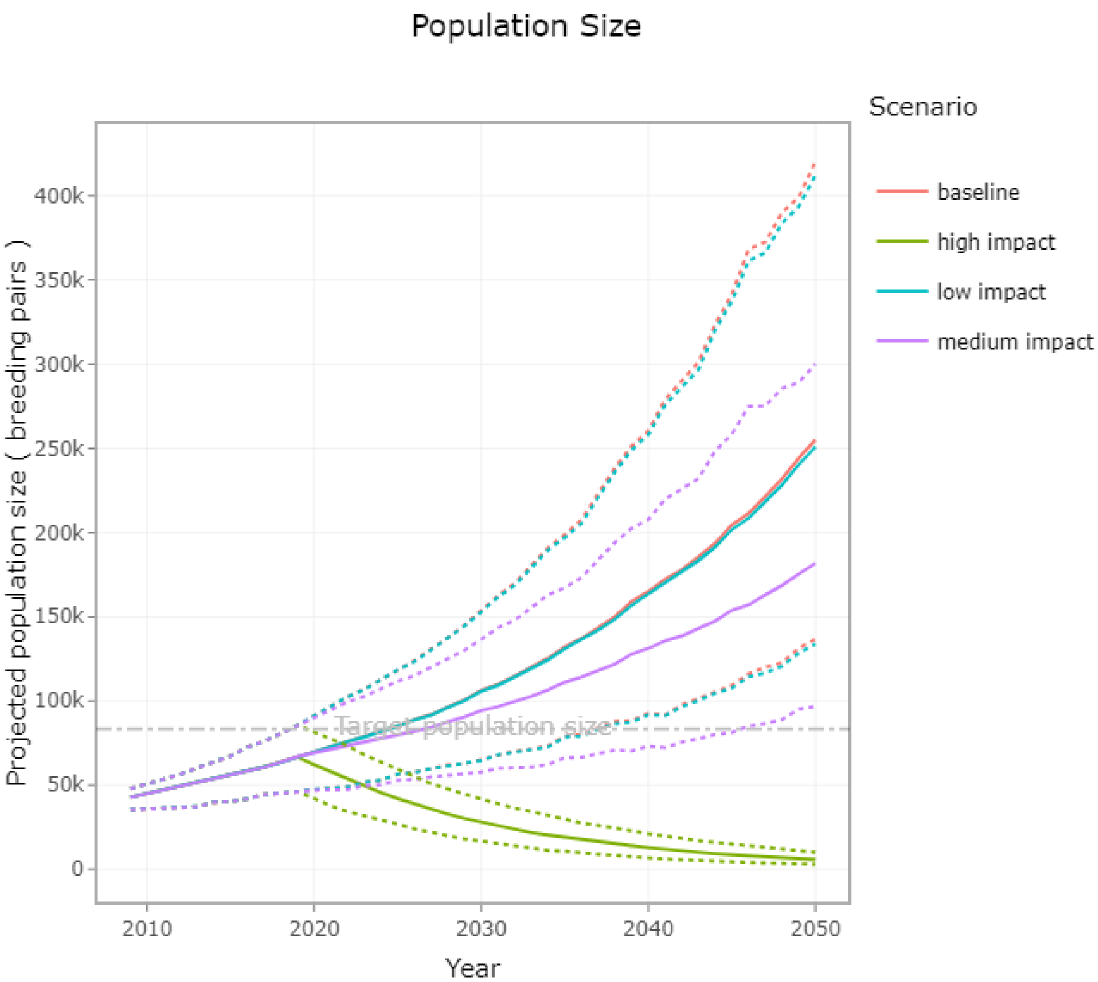



**Figure 26 Population projections for the baseline and low, medium and high impact scenarios.**

The counterfactuals of population size showed potentially important impacts on the population at medium impacts, and clearly important impacts at high impacts. At low impacts the CPS was close to one (no impact). The counterfactuals of growth rate show less important impacts for both the low and medium impact scenarios than the high impact scenario. The high impact scenario had a CGR that was clearly problematic.

**Table 58 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9838 | 0.9839 (0.9748 - 0.9932) | 0.9995 | 0.9995 (0.9992 - 0.9997) |
| Medium | 0.7119 | 0.7116 (0.7031 - 0.7194) | 0.9891 | 0.9891 (0.9887 - 0.9894) |
| High | 0.0225 | 0.0224 (0.0205 - 0.0242) | 0.8847 | 0.8846 (0.8821 - 0.8868) |

### 4.7.2 Closure of sandeel or sprat fisheries

[redacted]

**Table 59 Annual adult survival rate for each impact scenario compensated for by each compensation scenario (green > baseline survival, red < baseline adult survival).**

| | low compensation | medium compensation | high compensation |
|---|---|---|---|
| low impact | [redacted] | [redacted] | [redacted] |
| medium impact | [redacted] | [redacted] | [redacted] |
| high impact | [redacted] | [redacted] | [redacted] |

Projected population change of the baseline population (no impact) was compared with the high impact scenarios combined with the three potential levels of compensation on adult survival (Figure 27).

[redacted]

**Figure 27 Projected population size of breeding guillemots (individuals) at FFC SPA comparing the baseline with the high impact scenario combined with low, medium and high compensation scenarios.**

[redacted].

**Table 60 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | [redacted] |
| High | Low | [redacted] |
| High | Medium | [redacted] |
| High | High | [redacted] |

Examination of the counterfactuals of population size showed little difference in the CPS between the low and medium compensation scenarios. The high compensation scenario showed a larger CPS value, but this was still absolutely a small value and considerably below one (Figure 28).

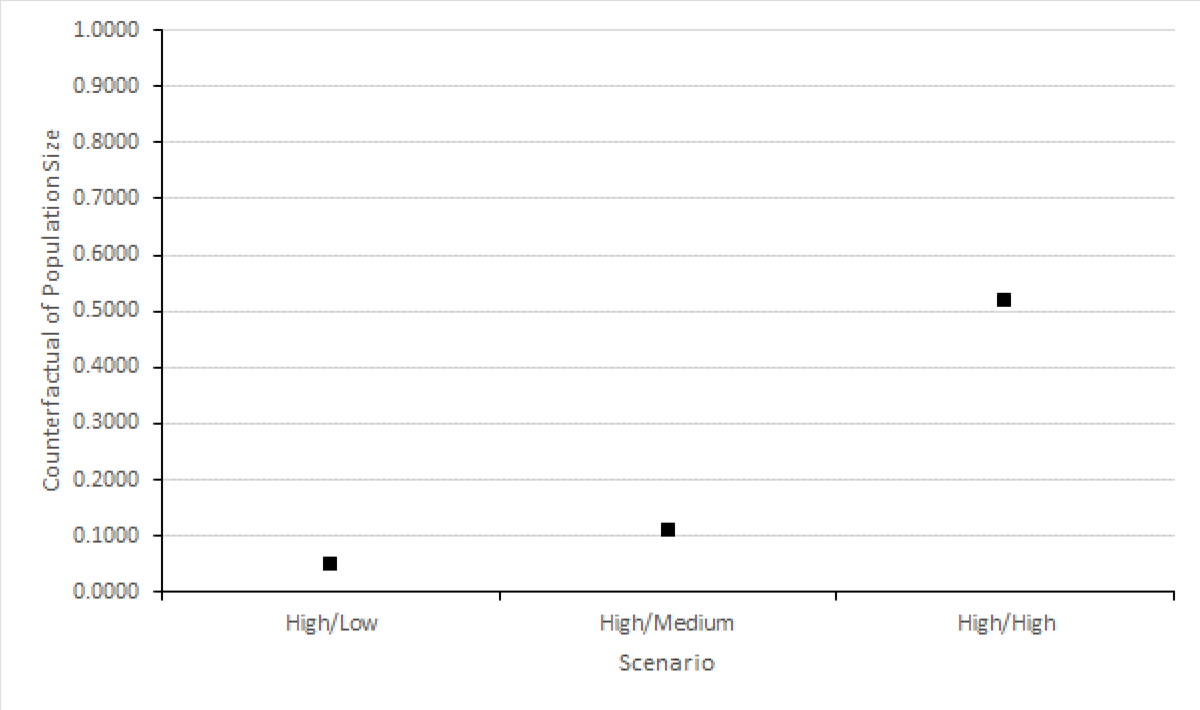



**Figure 28 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR showed a relatively little difference in the CGR between the low, medium, and high compensation scenarios (Figure 29). None of the values exceeded one and only the high compensation CGR value was close to one.

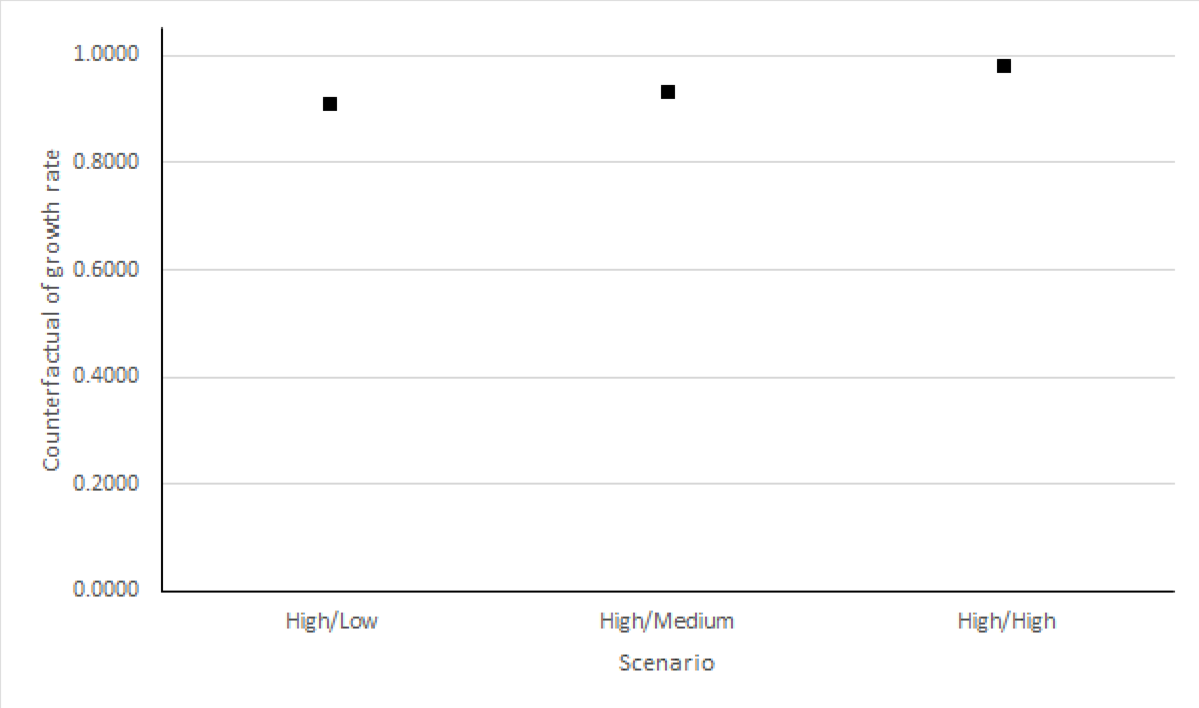



**Figure 29 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Overall, this assessment suggests that the compensation available from closing the UK sandeel and sprat fishery would likely be sufficient to overcome the low and medium impact scenarios entirely. The high impact scenario was too large for even a high level of compensation to fully address, but the model did suggest that some level of population growth would be possible with the resultant level of adult survival under that combination of high impact and compensation. The high impact scenario for guillemot is a very high level of impact on the population, and is perhaps unlikely to ever be realised, as future offshore wind farms sites beyond Round 4 are more likely to be increasingly far from shore. However, it does provide a useful indication of the limits of compensation through sandeel and sprat fisheries closures for the guillemot population at FFC SPA.

#### *4.7.2.1 Compensation ratios*

The levels of impact that compensation measures would need to overcome were calculated for 1:3 and 1:6 ratios (Table 61). The 1:1 ratio impacts were tested above and the high impact scenario was considered too large to be compensated for at any level. With the higher ratios, the high impact scenario at 1:3 and 1:6 was much larger exceeded the high impact ratio at 1:1 and so could not be compensated. The low impact scenarios at 1:3 and 1:6 were below the medium impact scenario tested at 1:1, so these could be compensated from closure of the sandeel and sprat fisheries. The medium impact scenario at 1:6 was greater than 100% additional mortality, so could not be compensated. The medium impact scenario at 1:3 was not more than 100% additional mortality, so the ability to compensate for this level of impact was also tested using a PVA. The low impact scenarios at both 1:3 and 1:6 was small enough that the testing above based on 1:1 ratios encompassed these levels, so PVAs were not run for these.

**Table 61 Low, medium and high impact scenarios at 1:3 and 1:6 ratios.**

| Impact scenario | Ratio | Low | Medium | High |
|---|---|---|---|---|
| Additional mortality (birds) | 1:3 | 118.5 | 2,451 | 25,903 |
| Additional mortality (rate) | | 3% | 62.1% | 655.5% |
| Additional mortality (birds) | 1:6 | 237 | 4,902 | 51,806 |
| Additional mortality (rate) | | 6% | 124.2% | 1,311% |

Projected population change of the baseline population (no impact) was compared with the medium impact scenario at a 1:3 ratio combined with the three potential levels of compensation on adult survival (Figure 30).

**Figure 30 Projected population size of breeding guillemots (individuals) at FFC SPA comparing the baseline with the medium impact scenario with a 1:3 ratio combined with low, medium and high compensation scenarios.**

The medium 1:3 impact scenario with low, medium, and high compensation scenarios resulted in the projected population size increasing in all cases (Table 62). Only the low compensation scenario was predicted to have a lower median annual growth rate than the baseline population projection. The population size increases shown are likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition.

**Table 62 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | [redacted] |
| Medium (1:3) | Low | [redacted] |
| Medium (1:3) | Medium | [redacted] |
| Medium (1:3) | High | [redacted] |

Examination of the CPS showed large differences in the values between the low, medium, and high compensation scenarios. The low compensation scenario resulted in a CPS value less than one, but for the medium and high scenarios the CPS value was greater than one (Figure 31).

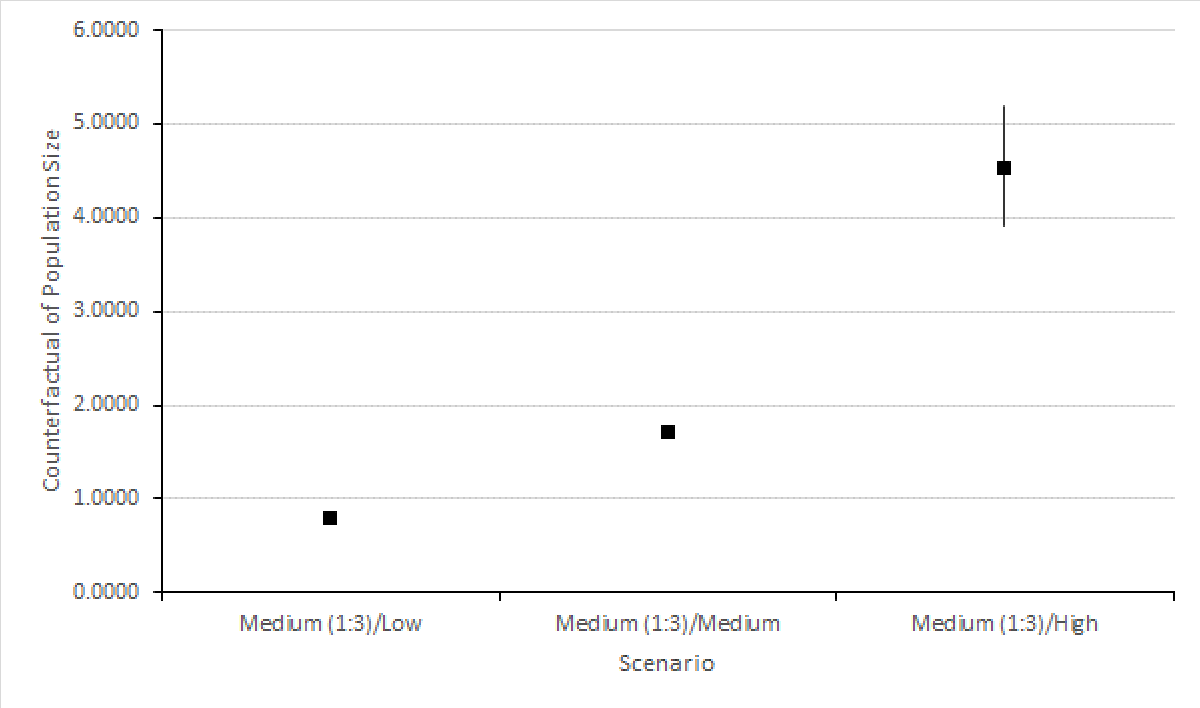



**Figure 31 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Examination of the CGR showed large differences in the values between the low, medium and high compensation scenarios. The low compensation scenario resulted in a CGR value less than one, but for the medium and high scenarios the CGR value was greater than one (Figure 32).

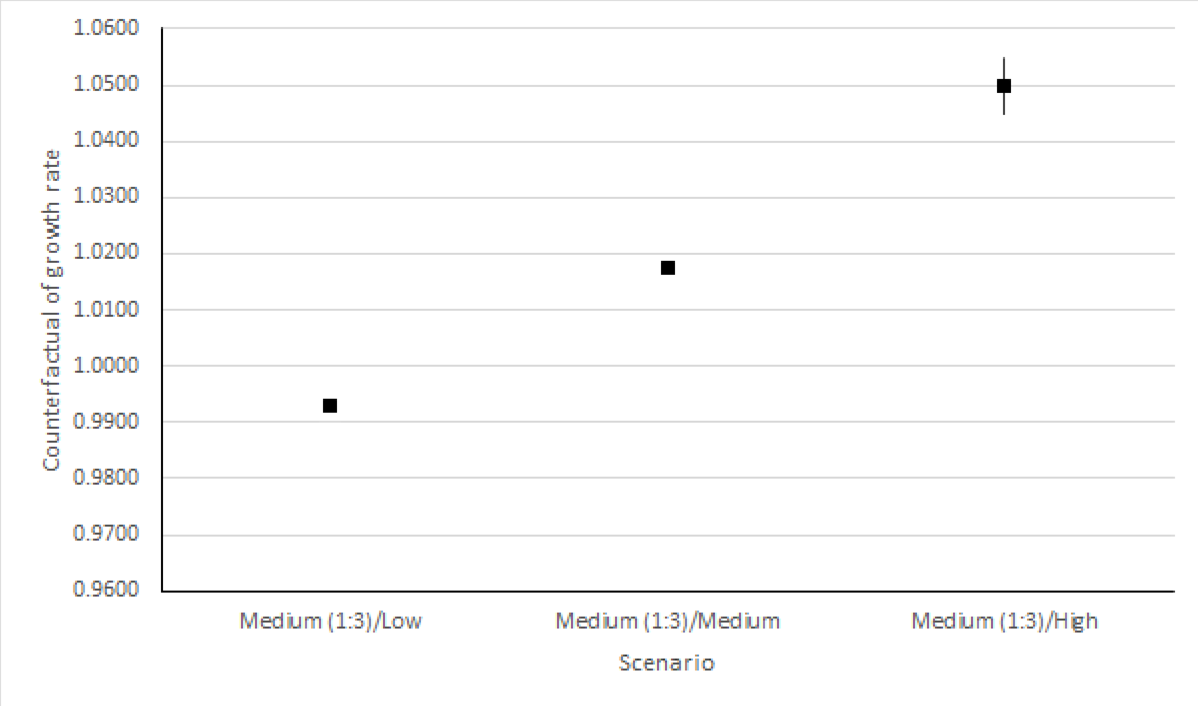



**Figure 32 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

This assessment suggests that the compensation available from closing the UK sandeel and sprat fishery would likely be sufficient to overcome the medium impact scenario with a 1:3 ratio if a medium or high compensation scenario is achieved. The low compensation scenario was still enough to result in some level of population growth, but this was a lower growth than the baseline scenario, so the impact was not fully compensated for.

It is therefore possible to conclude that 1:3 ratio could be compensated with the low and medium impact scenarios, but not the high impact scenario. The 1:6 ratio could only be compensated for with the low impact scenario, but not the medium or high impact scenarios.

#### 4.7.3 Eradication of rats and other invasive mammal predators

Population level assessment of the effects of rat and other invasive mammal predators on offshore islands on the FFC SPA guillemot population is not necessary. In the absence of any scope to deliver benefits to guillemots at FFC through predator eradication there, looking to enhance the population of guillemots at another site may be considered as an acceptable form of compensation with the aim of maintaining the coherence of network of SPAs for breeding guillemots in the UK.

Stanbury et al. (2017) identified islands in UK by the benefit of eradicating rats and other invasive mammal predators to breeding seabirds. For each for the 25 islands identified by Stanbury et al. (2017) as high priority those with an SPA where guillemot was a feature (either in its own right, or as a named feature of the breeding seabird assemblage) was identified (Table 63). For each island the citation population was recorded, and the most recent population size was determined from the SMP database. The numerical change in population size and percentage change since designation was calculated and the current Site Condition Monitoring status recorded. The presence of feral cats, brown and black rats, and American mink were determined using the information in Stanbury et al. (2017).

Some of the islands identified by Stanbury et al. (2017) either did not have areas on them designated as SPAs, or the seabird assemblage did not include guillemot as a named feature of the SPA breeding seabird assemblage. The absence of a SPA or guillemot appearing as a named feature of the assemblage did not necessarily mean the absence of breeding guillemots on that island. So for those islands where there was not a SPA the SMP database was used to determine whether breeding guillemots were present, what their population size was around 2000, what the most recent population count was and whether this had changed (Table 64).

**Table 63 Top 25 islands prioritised for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) and the SPAs designated for their guillemot populations, their citation population size (individuals), current population size (individuals), change in population size since designation, percentage change in population size since designation, current Site Condition Monitoring (SCM) status, and the presence of key invasive mammal predators. SCM status is highlighted as green if the population is Favourable and red if Unfavourable.**

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Foula | Foula SPA | 37,500 | 24,799 in 2007 | -12,701 | -34 | UD | Y | N | N | N |
| Fair Isle | Fair Isle SPA | 32,300 | 20,924 in 2015 | -11,376 | -35 | UD | Y | N | N | N |
| Westray | West Westray SPA | 42,150 | 28,697 in 2017 | -13,453 | -32 | UD | Y | N | N | N |
| Garbh Eilean and Eilean an Taighe, Shiants | Shiant Isles SPA | 18,380 | 9,054 in 2015 | -9,326 | -51 | UNc | N | N | Y | N |
| Rousay | Rousay SPA | 10,600 | 6,500 in 2016 | -4,100 | -39 | UD | Y | Y | N | N |
| Rathlin Island | Rathlin Island | 41,887 | 130,445 in 2011 | 88,558 | 211 | | Y | Y | N | N |
| Colonsay and Oronsay | North Colonsay & Western Cliffs SPA | 6,656 | 18,724 in 2018 | 12,068 | 181 | FM | Y | Y | N | N |
| Unst | Hermaness, Saxa Vord & Valla Field SPA | 25,000 | 6,109 in 2016 | -18,891 | -76 | UD | Y | Y | N | N |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Yell | None | | | | | | Y | N | N | N |
| Rum | Rum SPA | 4,000 | 2,454 in 2000 | -1,546 | -39 | UNc | Y | N | N | N |
| Papa Westray | None | | | | | | Y | N | N | N |
| Fetlar | None | | | | | | Y | N | N | N |
| Inchkeith | None | | | | | | N | Y | N | N |
| Hoy | Hoy SPA | 13,400 | 12,198 in 2017 | -1,202 | -9 | UNc | Y | Y | N | N |
| Flotta | None | | | | | | Y | Y | N | N |
| Tiree | None | | | | | | Y | Y | N | N |
| Inchmarnock | None | | | | | | N | Y | N | Y |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Stronsay | None | | | | | | Y | Y | N | N |
| Eilean Mhuire, Shiant Islands | Shiant Isles SPA | As above | | | | | N | N | Y | N |
| Gairsay | None | | | | | | Y | Y | N | N |
| North Ronaldsay | None | | | | | | Y | N | N | N |
| Muck | None | | | | | | N | Y | N | N |
| Housay, Out Skerries | None | | | | | | Y | Y | N | N |
| South Havra, Shetland | None | | | | | | Y | N | N | N |
| Herm, Channel Islands | None | | | | | | Y | Y | Y | N |

**Table 64 Islands prioritised for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) where the site was either not designated for their guillemot populations, or guillemot was not a named feature in the assemblage of more than 20,000 breeding individuals. Change and Percent change is highlighted as green if the population is increasing and red if decreasing.**

| Island | SPA | Seabird assemblage ? | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Yell | None | N | 208 | 1999 | no count | no count | | | Y | N | N | N |
| Papa Westray | North Hill and Holm SPA | N | 1712 | 1999 | 898 | 2019 | -814 | -47.5% | Y | N | N | N |
| Fetlar | Fetlar | Y | 136 | 2000 | no count | no count | | | Y | N | N | N |
| Inchkeith | None | N | 73 | 2000 | 187 | 2019 | 114 | 156.2% | N | Y | N | N |
| Flotta | None | N | 30 | 2002 | 62 | 2019 | 32 | 106.7% | Y | Y | N | N |
| Tiree | None | N | 1974 | 1999 | 3610 | 2008 | 1,636 | 82.9% | Y | Y | N | N |
| Inchmarnock | None | N | no count | no count | no count | no count | | | N | Y | N | Y |
| Stronsay | None | N | no count | no count | 761 | 2018 | | | Y | Y | N | N |
| Gairsay | None | N | no count | no count | no count | no count | | | Y | Y | N | N |
| North Ronaldsay | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Muck | None | N | no count | no count | no count | no count | | | N | Y | N | N |
| Housay, Out Skerries | None | N | no count | no count | no count | no count | | | Y | Y | N | N |

| Island | SPA | Seabird assemblage ? | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| South Havra, Shetland | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Herm, Channel Islands | None | N | 105 | 1999 | 135 | 2015 | 30 | 28.6% | Y | Y | Y | N |

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The breeding guillemot population sizes in SPAs on the islands identified by Stanbury et al. (2017) had mostly declined. Exceptions were the colonies in the North Colonsay & Western Cliffs SPA and Rathlin Island SPA, which had both increased. While the declines in the colonies shown in Table 63 and Table 64 are likely to be for a variety of reasons, it is possible that the presence of rats and other invasive mammal predators are a contributing factor. It is therefore possible that eradication of rats and other invasive mammal predators from these islands could contribute to the compensation for losses of breeding adult guillemots predicted for the FFC SPA population.

Comparing the low, medium and high impact scenarios (Table 56) with the declines shown in Table 63 and Table 64 indicates that based on the untested assumption that those recorded declines were driven by mammalian predation and could be wholly reversed if those predators were eradicated there is potential to compensate at a 1:1 ratio for even the high impact scenario through eradication of rats and other invasive mammal predators at a variety of islands, mostly in Scotland. Islands with a change in their population size since designation smaller than the impact scenario could have lower potential to compensate. This is summarised in Table 65, but note that rat eradication has already been completed on the Shiant Islands. It is important to note, that these islands would likely have been designated with the same rat and other invasive mammal predator population as they have now, so in the absence of other constraints to population size it is possible that population would grow in excess of the original citation population if predation pressure was reduced or removed, including those islands where, based simply on recorded declines since citation, the potential for successful compensation through this measure would appear to be lower (i.e. Amber shaded in Table 65).

**Table 65 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios at 1:1 ratio (Green = high potential, Amber = lower potential, Red = no potential).**

| Islands suitable for compensation | Compensation | | |
|---|---|---|---|
| | Low | Medium | High |
| | 39.5 | 817 | 8,635 |
| Foula | Green | Green | Green |
| Fair Isle | Green | Green | Green |
| Westray | Green | Green | Green |
| Garbh Eilean and Eilean an Taighe, Shiants | Red | Red | Red |
| Rousay | Green | Green | Amber |
| Unst | Green | Green | Green |
| Rum | Green | Green | Amber |
| Hoy | Amber | Amber | Amber |
| Papa Westray | Green | Amber | Amber |
| Inchkeith | Green | Red | Red |
| Flotta | Red | Red | Red |
| Tiree | Green | Green | Red |
| Herm, Channel Islands | Amber | Red | Red |

#### 4.7.3.1 *Compensation ratios*

Increasing the compensation ratios to 1:3 and 1:6 resulted, unsurprisingly, in fewer potentially suitable islands being available for compensation (Table 66). No islands were suitable for High impacts at either 1:3 or 1:6 ratios. At the medium impact level six islands were probably or possibly suitable and at the low impact level there were nine possible islands.

**Table 66 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios at 1:3 and 1:6 ratios (Green = high probability, Amber = lower probability, Red = no probability).**

| Islands suitable for compensation | Change | 1:3 | | | 1:6 | | |
|---|---|---|---|---|---|---|---|
| | | Low | Medium | High | Low | Medium | High |
| | | 118.5 | 2,451 | 25,905 | 237 | 4,902 | 51,810 |
| Foula | -12,701 | Green | Green | Red | Green | Green | Red |
| Fair Isle | -11,376 | Green | Green | Red | Green | Green | Red |
| Westray | -13,453 | Green | Green | Red | Green | Green | Red |
| Garbh Eilean and Eilean an Taighe, Shiants* | -9,326 | Red | Red | Red | Red | Red | Red |
| Rousay | -4,100 | Green | Green | Red | Green | Amber | Red |
| Unst | -18,891 | Green | Green | Red | Green | Green | Red |
| Rum | -1,546 | Green | Red | Red | Green | Red | Red |
| Hoy | -1,202 | Green | Red | Red | Green | Red | Red |
| Papa Westray | -814 | Green | Red | Red | Green | Red | Red |
| Inchkeith | 114 | Red | Red | Red | Red | Red | Red |
| Flotta | 32 | Red | Red | Red | Red | Red | Red |
| Tiree | 1,636 | Red | Red | Red | Red | Red | Red |
| Herm, Channel Islands | 30 | Red | Red | Red | Red | Red | Red |

* Invasive terrestrial mammals have already been eradicated from the Shiants.

### 4.7.4 EAA 2

#### 4.7.4.1 *Sandeel closure*

Population level assessment was not completed for any other SPAs in the EAA 2 area. Twenty-two SPAs designated for their breeding guillemot populations also occurred within EAA 2.

All of these colonies would be expected to benefit from a closure of UK water to sandeels, as all of these colonies occur within foraging range of sandeel stocks that are currently or previously depleted through fishing. The sandeel box off the east coast of Scotland has likely positively affected the guillemot populations in SPAs on the east coast of Scotland. It is important to note that the sandeel box is a fisheries management tool, not a seabird conservation tool, and that current management of the stock that includes the sandeel box allows a take of the stock that could deplete the stock despite the presence of the box. This stock has not experienced the same level of take since the box was put in place, though fishing occurred within the stock in 2021.

#### 4.7.4.2 *Eradication of rats and other invasive mammal predators*

The majority of islands identified in Table 65 occurred in EAA 2 (Table 67). Since eradication of mammals from FFC SPA is not a suitable compensation measure, the next level of preferred location would be the islands occurring in EAA 2 shown in Table 67.

**Table 67 EAA for islands identified as potentially suitable for compensation through eradication of rats and other invasive mammal predators.**

| Islands suitable for compensation | EAA |
|---|---|
| Foula | 2 |
| Fair Isle | 2 |
| Westray | 2 |
| Garbh Eilean and Eilean an Taighe, Shiants | 1 |
| Rousay | 2 |
| Unst | 2 |
| Rum | 1 |
| Hoy | 2 |
| Papa Westray | 2 |
| Inchkeith | 2 |
| Flotta | 2 |
| Tiree | 1 |
| Herm, Channel Islands | 1 |

### 4.7.5 EAA 1

#### 4.7.5.1 *Sandeel closure*

The closure of UK waters to sandeel fisheries would be expected to mainly benefit colonies in areas that are currently, or have previously been, subject to sandeel fisheries. There are 12 SPAs designated for their guillemot populations in EAA 1, but much of the areas of sea within the foraging range of birds in these SPAs has not experienced the same take of sandeels from stocks as those in the North Sea. It is therefore considered unlikely that closure of sandeel fisheries would have the same level of benefit to guillemot populations in EAA 1.

#### 4.7.5.2 *Eradication of rats and other invasive mammal predators*

The islands with potential for compensation in EAA 1 are shown in Table 67. These islands would be in the final tier of location hierarchy, with sites in EAA 2 preferred. Only four islands were identified and one of these (Shiant Islands) has already had invasive mammals eradicated.

## 4.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the two recommended compensation measures was estimated for the closure of sandeel or sprat fisheries compensation approach and the eradication of rats and other invasive mammals approach. The summary table for the confidence in the closure of sandeel or sprat fisheries compensation method is shown in Table 68, and the confidence in the closure of sandeel or sprat fisheries compensation assessment is shown in Table 69. The summary table for confidence in the eradication of rats and other invasive mammals compensation method is shown in Table 70 and the summary table for confidence in the eradication of rats and other invasive mammals compensation method is shown in Table 71. The narrative describing and justifying the values given to the evidence and applicability metrics are described in and Table 72 and Table 73 (closure of sandeel or sprat fisheries) and in Table 74 and Table 75 (eradication of rats and other invasive mammals).

**Table 68 Assessment of confidence in the closure of sandeel or sprat fisheries method to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Importance of maintaining sufficient prey densities in the vicinity of the colony | n/a | Hentati-Sundberg et al. (2020) | Robust | Robust | n/a | Medium | ROBUST | MEDIUM | HIGH |
| Forage fish availability as determinant of breeding success, survival and population change | n/a | Montevecchi et al. (2019) | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Fisheries management target for conserving seabirds as 1/3 of historical maximum prey biomass | B1/3 | Cury et al. (2011) | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Limited buffering capacity of guillemots to prey reduction | n/a | Kadin et al. (2016) | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Effect of prey availability on guillemot body mass and chick-feeding rates | n/a | Storey et al. (2017) | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Fixed and individual-specific migration strategies | n/a | Merkel et al. (2020) | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Fish aggregations as the main driver of guillemot spatial aggregations in winter | n/a | Schaefer et al. (2020) | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Guillemot mortality peaks during winter | n/a | Louzao et al. 2019 | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Guillemots in UK waters are NOT close to their energy limits in winter | n/a | Dunn et al. (2020) | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| [redacted] | n/a | In prep. | [redacted] | | | | | | |
| | | | | | | | | OVERALL CONFIDENCE SCORE | MEDIUM |

**Table 69 Assessment of confidence in the closure of sandeel or sprat fisheries assessment of compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 6 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 1 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 83,214 individuals in 2008 | SACO | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | mean: 0.7137, sd: 0.1094 | SMP database | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| FFC SPA population growth rate | mean 1.0465 from 1988 to 2017 | Seabird Colony Register, Aitken et al. 2017 | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Relationship between sandeel biomass and guillemot annual return rates | y=0.0542ln(x) + 0.2158 | Newell et al. (2016), JNCC (2020), NatureScot | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Adult survival rate | mean: 0.9525, s.d. 0.058 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Age class 0 to 1 | mean: 0.56, sd: 0.058 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.792, sd: 0.152 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 2 to 3 | mean: 0.917, sd: 0.098 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.938, sd: 0.107 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 4 to 5 | mean: 0.924, sd: 0.058 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 5 to 6 | mean: 0.924, sd: 0.058 | PVA app "Isle of May" default value | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Target population size | 83,214 individuals | NE FFC SPA SACO | Robust | Medium | Medium | Robust | MEDIUM | HIGH | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 70 Assessment of confidence in the eradication of rats and other invasive mammals method to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Eradication of rats increases breeding population size | n/a | Booker et al. 2019, Luxmoore et al. (2019) | Robust | Limited | Low | Robust | MEDIUM | HIGH | HIGH |
| Presence of invasive mammals on islands with guillemots | n/a | Stanbury et al. 2017 | Robust | Robust | Robust | Medium | MEDIUM | HIGH | HIGH |
| Rats reduce adult survival | n/a | n/a | Limited | Limited | n/a | Limited | LIMITED | LOW | VERY LOW |
| Rats reduce productivity | n/a | n/a | Limited | Limited | n/a | Limited | LIMITED | LOW | VERY LOW |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | MEDIUM |

**Table 71 Assessment of confidence in the eradication of rats and other invasive mammals approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Citation population size | n/a | SiteLink | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |
| Current population size | n/a | SMP database | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Site Condition | n/a | SiteLink | Robust | n/a | n/a | Medium | MEDIUM | HIGH | MEDIUM |
| Presence of invasive mammals | n/a | Stanbury et al. 2017 | Robust | n/a | n/a | Medium | MEDIUM | HIGH | MEDIUM |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 72 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures method from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Importance of maintaining sufficient prey densities in the vicinity of the colony | Hentati-Sundberg et al. (2020) created a model to explore the effect of prey densities near the colony. As this result was based on a model output the type of evidence was assessed as medium. The study was based on a guillemot colony in the Baltic Sea, so the applicability was assessed as low. |
| Forage fish availability as determinant of breeding success, survival and population change | Montevecchi et al. (2019) undertook an analysis of a very large data set. All of the evidence metrics were robust. However, since the study was in Canada and the key prey species was capelin, the applicability was assessed as low. As such the overall confidence was medium. |
| Fisheries management target for conserving seabirds as 1/3 of historical maximum prey biomass | The study by Cury et al. (2011) was very large and examined the effects of fisheries management on a wide variety of seabirds and their prey. Guillemot was included in their study from three locations in different parts of the world, including in the North Sea with sandeel prey. Thus, evidence was all robust and applicability high giving a very high overall confidence. |
| Limited buffering capacity of guillemots to prey reduction | Kadin et al. (2016) undertook a robust study of guillemot response to quantity and quality, so evidence scores were all robust. Since the study was in the Baltic Sea the applicability was considered low. However, an overall evidence score of high was still given and seems reasonable. |
| Effect of prey availability on guillemot body mass and chick-feeding rates | Storey et al. (2017) was also a very robust study, but since it was undertaken in Newfoundland, Canada an applicability score of low was given. This resulted in an overall confidence score of medium. |
| Fixed and individual-specific migration strategies | Merkel et al. (2019) was also a very robust study but undertaken in the north-east Atlantic Ocean. Most of the colonies studied were from Iceland east to Novaya Zemlya, but since the Isle of May was included in the study the applicability as assessed as medium. This resulted in a high overall confidence. |
| Fish aggregations as the main driver of guillemot spatial aggregations in winter | Schaefer et al. (2020) was another very robust study but was carried out in Alaska. So, the applicability was assessed as low. Giving an overall medium confidence score. |
| Guillemot mortality peaks during winter | Louzao et al. (2019) was a further robust study but had a low applicability score as the study was from the north coast of Spain. The overall confidence score was medium. |
| Guillemots in UK waters are NOT close to their energy limits in winter | Dunn et al. (2020) was a robust study from the Isle of May giving a medium applicability score and a high overall confidence score. |
| [redacted] | [redacted] |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was mostly low or medium. Confidence scores were medium or high, with one very high. So, an overall score of medium was given. However, the spread of studies across the range of guillemots helped to build a picture of the importance of |

| Metric | Narrative |
|---|---|
| | foraging fish to guillemot populations and strongly supported the approach for sandeel fisheries closures as an effective compensation approach. **Therefore, the overall confidence score was increased to high.** |

**Table 73 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures assessment from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of guillemots is well established and not in question. It is not variable between populations and is directly applicable to the FFC SPA population. |
| Upper constraint on productivity | Guillemots lay a single egg and so productivity cannot be above this. |
| Initial population size | These data are based on recent counts using accepted and standardised methods from the colony being investigated. |
| Productivity rate per pair | The productivity rate was based on the mean of the available data in the SMP database across a relatively long time series at FFC SPA. |
| FFC SPA population growth rate | This value was calculated from robust count data from two dates across a relatively long period. |
| Relationship between sandeel biomass and guillemot annual return rates | This relationship was derived from published or reliable data sources and was considered robust. |
| Adult survival rate | The adult survival rate was derived from the PVA model output to achieve the observed growth rate at the FFC SPA. This was a more robust approach than using generic data on adult survival from one or more other guillemot colonies. Only an adult survival rate modelled from ringing data from the FFC SPA colony itself would be considered a more robust data source. |
| Age specific survival rates from 0 to 6 years | These survival rates were based on the default "Isle of May" survival rates in the Seabird PVA Tool. These values were from reliable published sources. They were from another North Sea guillemot colony, but one relatively close to the FFC SPA and so their applicability was considered medium. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was high or medium, so an overall score of high was given. |

**Table 74 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation method from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Eradication of rats increases breeding population size | Two sources of the effects of rat eradication from offshore islands were found. One found a strong response of guillemot populations to rat eradication (Booker et al. 2019) and another found no response (Luxmoore et al. 2019). Both studies were considered to provide robust quality of evidence, and both were robust types of evidence (numeric). However, with only two studies available the amount of evidence was low. Since both studies had opposing results the consistency of evidence was also low. This resulted in an overall medium evidence score. Since this evidence would be directly applicable to colonies in the UK suggested for compensation, the applicability was scored high. **While this resulted in an overall evidence score of high, the opposing evidence from Lundy and Canna was considered to be sufficiently important to reduce the confidence to medium**. |
| Presence of invasive mammals on islands with guillemots | Stanbury et al. (2017) was only used to identify potential islands for compensation. The quality, consistency and amount of evidence were all scored as robust, but the type of evidence was scored as medium as the study only determined the presence/absence of predators and not their absolute or relative abundance. This was considered sufficiently important to result in an overall evidence score of medium. Applicability was high as this evidence was for the islands that could be used for compensation. The overall confidence score was therefore high. |
| Rats reduce adult survival | There was no direct empirical evidence found that rats reduce the adult survival of guillemots, so this metric was assessed as having low evidence and low applicability giving a very low confidence score. |
| Rats reduce productivity | There was no direct empirical evidence found that rats reduce the productivity of guillemots, so this metric was assessed as having low evidence and low applicability giving a very low confidence score. |
| **OVERALL CONFIDENCE SCORE** | With medium confidence in the evidence that rat eradication can increase guillemot breeding populations and high confidence in the evidence of rats occurring on islands with guillemot colonies there was a medium confidence in this measure. The lack of evidence that rats predate adult guillemots, or their egg/chicks was considered less important than the evidence that rat eradication may result in increases in guillemot populations. This is because the mechanism for population increase may not be due to direct predation but on the restriction of nesting location on offshore islands. Removal of rats may result in expansion of the colony into areas that were previously not used because of the presence of rats. |

**Table 75 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Citation population size | The citation population size was obtained from the Citation document from NatureScot SiteLink. This is a legal document, so the value is not in question. |
| Current population size | The current population size was the most recent whole SPA count available from the SMP database. The most recent count was used for each site, so these may be different years between different SPAs. However, these are the best available evidence, and the count method and source were considered robust. |
| Site Condition | Site Condition of each guillemot SPA population was obtained from the NatureScot SiteLink website. Site Condition Monitoring is a count-based methodology, but the assignment of condition scores is not based on site- or species-specific methodologies, so the type of evidence was scored as medium. |
| **OVERALL CONFIDENCE SCORE** | There were two very high and one medium confidence score. Thus, an overall confidence score of high was considered appropriate. However, the assumption that changes in population size between citation counts and the most recent count were caused by invasive terrestrial predators is untested. Indeed, for many sites this is unlikely to be the only, or even main, cause of population decline. **Therefore, confidence in this metric needs to be reduced to low.** |

With an overall assessment of high confidence in the method for the closure of the sandeel and sprat fishery and high confidence in the assessment method, the assessment of confidence in the proposed compensation methods against the three impact scenarios needs to be carefully considered. Under each of the three different increases to adult survival rate assumed to follow from closure of sandeel and/or sprat fisheries (+2.5%, +5% or +10%), the increases to adult survival were sufficient to raise the impacted adult survival back to the baseline value. As each of these assumed increases to adult survival were [redacted] which evidence suggests might arise from a closure that markedly increases forage fish (sandeel) stock sizes (Figure 21) confidence was high that this measure could effectively compensate for these levels of impact. It was apparent from the population level assessment that the high impact scenario could not be compensated through closure of the UK sandeel or sprat fisheries (see 4.7.2), with population growth below one for the low and medium compensation scenarios but greater than one for the high compensation scenarios (Table 60). For the low and medium impact scenarios all of the compensation approaches were assumed to bring the adult survival rate to the same as the baseline (see Table 59). For the high impact scenario, the CPS values were all low (less than 0.6; Figure 28). The CGR values did not exceed one for any of the high impact scenarios (Figure 29). Thus, with medium confidence in the compensation method and high confidence in the assessment there was low confidence that the high impact scenarios could be compensated for, but high confidence for all the other scenarios (Table 76). It is important to note that these assessments used only an increase in adult survival as there was insufficient empirical evidence to parameterise an increase in productivity. However, it is likely that closure of sandeel and sprat fisheries would also increase the productivity of guillemots. This would likely occur at the FFC SPA and all other guillemot colonies dependent on these fish stocks.

**Table 76 Assessment of confidence in the impact/compensation scenarios for guillemots from sandeel or sprat fisheries closure.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | HIGH | LOW |
| Medium compensation | HIGH | HIGH | LOW |
| High compensation | HIGH | HIGH | LOW |

The rat and other invasive mammal eradication measures also had a medium confidence in the compensation method and a high confidence in the assessment. The confidence in the ability to compensate for the three impact scenarios varied between islands (Table 65). The identification of islands with invasive terrestrial predators and guillemot populations that have declined (Table 65) provided a suitable assessment in the confidence that invasive mammal eradication was a suitable compensation method. Since it is apparent that rat eradication from some islands (i.e. Lundy) has a positive effect on the guillemot population but from others it has no effect (i.e. Canna) it is clear that further evidence is needed to determine what the confidence is in this compensation method. It seems highly likely that rat eradication on some of the islands identified by Stanbury et al. (2017) would result in positive effects on the guillemot population, and that there would be high confidence **if these could be identified**. Since these were not identified by Stanbury et al. (2017) and it was beyond the scope of this study to identify these islands, then the current confidence assessed as medium, should be assessed as low at present, as it would not be possible to assume that rat eradication on one of these islands would result in a population increase.

**Table 77 Assessment of confidence in the impact/compensation scenarios for guillemots from eradication of rats.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | LOW | LOW | LOW |
| Medium compensation | LOW | LOW | LOW |
| High compensation | LOW | LOW | LOW |

## 4.9 Future monitoring and adaptive management

### 4.9.1 Closure of sandeel or sprat fisheries

If compensation through closure of the UK sandeel fishery was applied, it would be important that suitable monitoring is put in place to demonstrate that this has been effective at:

- Increasing the sandeel stock available to the guillemot population at FFC SPA; and
- Increasing the survival of adult guillemots at FFC SPA.

Methods for suitable monitoring of sandeel stocks would need to be established with experts in this field and is beyond the scope of this study. However, fishery-independent assessment methods have been developed by Norway (using acoustic sampling) and by Marine Scotland (using grab samples) as described in the latest sandeel benchmark stock assessment (ICES 2017). Monitoring would need to determine overall abundance of the stock and perhaps also the stock within the foraging range of guillemots from the FFC SPA to ensure that the compensation measure is having the desired effect on the prey resource for the population at FFC SPA.

In addition to assessing the stock at both the relevant ICES stock area and within the foraging range of the FFC SPA guillemot population, monitoring of the annual return rates of marked individuals, as a proxy for survival, or modelling of survival rate based on recaptured/resightings over a period of years would be important.

These monitoring measures need to be connected to adaptive management decision making. The proposed monitoring needs to be considered together when adapting the management to the results of the monitoring. The aim of the proposed compensation is to increase the adult survival of guillemots at the FFC SPA colony, with an overall objective of maintaining or increasing the population size. Monitoring of sandeel stocks is needed to determine whether recovery of the stock was as expected, below the level expected or above the level expected. Similarly adaptive management will need to consider whether action is necessary if the change in adult survival is above or below the expected value. Ultimately the need to adapt management actions will need to be based on whether the population size at the SPA changes as a result of the proposed compensation method. Adaptive management actions will need to consider the pattern of change in all the monitored elements before deciding whether, and what type of, corrective action is needed. Even if the population size does not increase in the FFC SPA, no further action may be appropriate if the other elements have been shown to have positively changed and the guillemot population in the UK network of SPAs can be shown to have increased. This could occur where the FFC SPA becomes a source population, exporting more birds than needed for intrinsic growth, or it could be as a result of competition from other species that also benefit from the closure of the sandeel fishery. Where closure of the fishery can be shown not to have had a positive effect on the FFC SPA guillemot population it will be important to understand why this is before deciding on subsequent management actions. It may be necessary to move to other compensation mechanisms should the closure of the sandeel fishery ultimately prove to be unsuccessful.

#### 4.9.2 Eradication of rats and other invasive mammal predators

After a suitable location for rat eradication is selected it will be important to collect data on the current population size of the guillemot population on that island. At least one breeding season count using the SMP methodology should be completed to provide an up-to-date baseline condition for future comparison. In addition, evidence should be collected from the island, preferably around the colony, to demonstrate that rats or other invasive mammals are present. This would likely need to be carefully designed for the specific island and colony being considered for eradication. Suitable methods for collecting these data would be chew sticks, camera traps, live traps and kill traps. Following application of an eradication method suitable for the invasive mammals present (there may need to be more than one) it will be necessary to continue monitoring to ensure eradication has been successful. Should either the targeted species, or other invasive mammal species, be detected further eradication measures would likely be necessary. Monitoring for the presence/absence of mammals should continue throughout the period of compensation. Details on the intensity and frequency of this monitoring would need to be site specific. Several important characteristics will need to be considered when determining this frequency and intensity. These include distance from nearest other mammal population (more remote island will need less monitoring), type and frequency of boat visits (islands only visited occasionally by small passenger only vessels are less exposed to re-introduction of invasive mammalian predators than more frequent visits by larger cargo vessels), presence and type of working agriculture on the island (islands with working farms are more likely to accidentally introduce invasive mammal predators than those with none), and presence and size of the human population living on the island (reintroduction risk of invasive mammals increases with the presence of human populations on the island).

Monitoring of the breeding adult guillemot population would be necessary to demonstrate that the colony increases in size following the removal of invasive mammal predators. Initially annual counts of the colony size are advised, with this frequency being reduced as recovery continues. The level of reduction in monitoring frequency should be related to the level of recovery observed. Monitoring should continue throughout the period compensation is required, at least periodically. If invasive mammal predators appear to have re-established then monitoring frequency should increase following additional eradication, particularly if the population was suppressed by the re-establishment.

#### 4.9.3 Future research

Despite a wealth of high-quality published studies on the effects of forage fish population on breeding guillemots there are some important gaps in knowledge. A key gap is the effect of sandeel fisheries stocks on productivity of guillemots. While this assessment has been limited to considering the potential effects on populations through changes to adult survival, it seems likely that increases in sandeel stocks would also positively affect the productivity of guillemots. In addition, it is likely that the quality of fish available to chicks is important, and improved stocks may also increase the quality of available fish and therefore have a benefit to productivity.

The two studies available on the effects of rat eradication on guillemots yielded contradictory results. It appears that the causes for this are down to the structure of the habitat affected, but further research would be valuable. Understanding how rats, and other predators, limit guillemot colonies would allow appropriate islands for eradication to be identified with greater certainty.

### 4.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for guillemot impacts at FFC SPA: closure of sandeel and sprat fisheries in all UK waters, rat eradication, and prevention of oil spills. Oil spills have become rare in the UK through both strong legislation and strong application of best practice, so it was concluded that this would no longer be a suitable compensation measure. Since Furness et al. (2013) was published there has been a considerable amount of new evidence published on the benefits of fisheries closure and invasive mammal eradication. While existing evidence on the effects of sandeel biomass on guillemot breeding success was weak, new evidence was available showing a strong effect on adult survival of guillemots. While bycatch has been noted as a potential impact that if reduced could compensate for impacts to FFC SPA guillemots, conservation measures are already underway for this, so there is no scope for compensation to provide additional benefits.

PVA suggested that the FFC SPA population should be increasing and that this increase would remain likely with low and medium impact levels. However, the high impact scenario resulted in a rapid projected decline in the FFC SPA population. Three levels of compensation measure were then assessed against the three levels of impact. The increase of adult survival of guillemots from sandeel fisheries closure was tested in the PVA model. However, there is an upper limit to adult survival and it was found that for both the low and medium impact scenarios this upper limit was exceeded for all levels of compensation. Thus, there was high confidence that for these impact scenarios even the low compensation scenario (a 2.5% increase in adult survival) was sufficient to compensate. This was not the case for the high impact scenario, so the PVA focussed on the ability of measures to compensate high impact only. The PVA results suggested that none of the compensation scenarios were sufficient to overcome the High impact scenario. It was thought that this was due to the very high level of impact being tested in the High scenario and that it seems unlikely that such a high impact would be realised through future offshore wind development.

Assessment of compensation through eradication of invasive mammal predators from offshore islands was able to identify several islands where the recorded decline in the population size of guillemots was larger than the impact scenarios being considered. On the assumption that these declines may have been driven largely or in part by the effects of mammalian predation, and that removal of that pressure would see those declines reversed, there can be at best a medium degree of confidence that implementing this measure at islands where the decline exceeds the impact level at FFC SPA could compensate for the impact level being considered albeit only at the wider SPA network level. However, the likelihood that all of the declines shown were due to invasive terrestrial predators is low and the contribution of predators to this is unknown, so confidence was modified to low. It is important to note that not all of the islands identified as potential sites for compensation will be suitable and further research, including site visits, are likely necessary to identify locations where rat eradication could result in positive improvements for guillemot colonies.

Confidence in the sandeel and sprat fishery closure approach was assessed as medium and in the PVA assessment process was high. The narrative suggested that the confidence in the approach should be increased to high, due to the multiple sources of high-quality information to build a picture of the effects of forage fish abundance on guillemot demographics. However, with very high impacts being tested in the high impact scenario it was concluded there was low confidence that these could be compensated for through sandeel fisheries closures. However, confidence was high for compensating the low and medium impact scenarios because the simulated increases in survival were all lower than empirical data indicate may occur following sandeel stock recovery. There was low confidence in the ability of rat eradication from offshore islands to compensate for low, medium or high impact scenarios across multiple islands among those identified by Stanbury et al. (2017).

# 5 FLAMBOROUGH AND FILEY COAST SPA – BREEDING RAZORBILL

The FFC SPA is on the east Yorkshire coast of the North Sea. The site is in two sections, with a gap in the middle of Filey Bay. The Flamborough section is south of this gap and the Filey section is north. The habitats within the site include clifftop, sea cliff and intertidal rock habitats, and the coastal sea out to two km from the coast.

The site was designated due to its nationally and internationally important breeding seabird colony, currently the largest mainland seabird colony in England. The SPA includes the only mainland breeding gannet colony in England, the largest kittiwake colony in the UK and the largest guillemot and razorbill colonies in England. The whole site supports more than 200,000 seabirds during the breeding season. The SPA includes all of a slightly smaller area that had been designated as Flamborough Coast and Bempton Cliffs SPA for breeding kittiwake. The change in area covered between these two designations makes some historical comparisons of breeding numbers of seabirds more difficult because of changing boundaries to areas counted.

The sea adjacent to the SPA is used by the breeding seabirds for a range of activities, including bathing, preening, displaying, loafing and local foraging. Offshore of the SPA the oceanographic frontal system known as the 'Flamborough Front' results in nutrient-rich waters and contributes to sustaining many of the qualifying features of the site.

## 5.1 Conservation status of razorbill

Razorbill has an IUCN Red List classification of "Near Threatened" and the UK population was listed in BOCC 2, 3, and 4 as amber. It is listed by the Birds Directive as a migratory species. The biogeographic population (subspecies *islandica*, in NW Europe) was estimated at 530,000 pairs, of which 110,000 pairs breed in Great Britain and 35,000 pairs in all-Ireland (Mitchell et al. 2004). AEWA (2012) give an equivalent estimate of 1,380,000 individuals, translated to pairs by dividing this total by 3 (NE 2018), giving 460,000 pairs. National surveys found a 16% increase in breeding numbers in the UK from 1969 to 1986, and a 21% increase from 1986 to 2000 (JNCC 2020). JNCC SCM data (JNCC 2020) show that breeding numbers in Scotland increased from an index of 100 in 1986 to about 180 in 2001, but then decreased back to an index of 100 in 2013 before increasing again, to 182 in 2018 (JNCC 2020). Stronger increases are evident in England and Wales (JNCC 2020).

Stroud et al. (2016) identified that the SPA suite with breeding razorbill as a designated feature has 18 qualifying sites in Great Britain, 16 in Scotland (Cape Wrath SPA; East Caithness Cliffs SPA; Fair Isle SPA; Forth Islands SPA; Flannan Isles SPA; Foula SPA; Fowlsheugh SPA; Handa SPA; Mingulay and Berneray SPA; North Caithness Cliffs SPA; North Rona and Sula Sgeir SPA; St Abb's Head to Fast Castle SPA; St Kilda SPA; The Shiant Isles SPA; Troup, Pennan and Lion's Heads SPA; West Westray SPA), one in England (FFC SPA) and one in Wales (Skokholm, Skomer and Middleholm SPA, now known as Skomer, Skokholm and seas off Pembrokeshire SPA). The SPAs in Great Britain were estimated to hold 62% of the Great Britain breeding population of razorbills present in 2000 (Stroud et al. 2016). One site in Northern Island also qualifies (Rathlin Island).

Apart from the marine extensions at some SPAs for loafing seabirds close to colonies, no sites were listed in the 3rd UKSPA review as designated as marine areas for razorbills (Stroud et al. 2016). Since then, Outer Firth of Forth and St Andrews Bay Complex SPA, designated on 3 December 2020 includes razorbill as a nonbreeding season feature.

Historical published counts of razorbills (numbers of individuals on land) at Flamborough Head and Bempton Cliffs SPA include 1,724 in 1969, 7,688 in 1987, 8,463 in 2000, 14,956 in 2008, and 27,967 in 2017 according to Lloyd et al. (2019). The counts from 1987, 2000, 2008 and 2017 are also listed in JNCC (2020). In the larger area of the whole FFC SPA there were 30,228 razorbills in 2017 (Lloyd et al. 2019), so the majority of this species are to be found within the original Flamborough Head and Bempton Cliffs SPA, with an additional 2,261 individuals (an additional 7%) in 2017 in the part of FFC SPA that is outwith Flamborough Head and Bempton Cliffs SPA boundaries. These data show a clear and strong increase in numbers of razorbills (Figure 33).

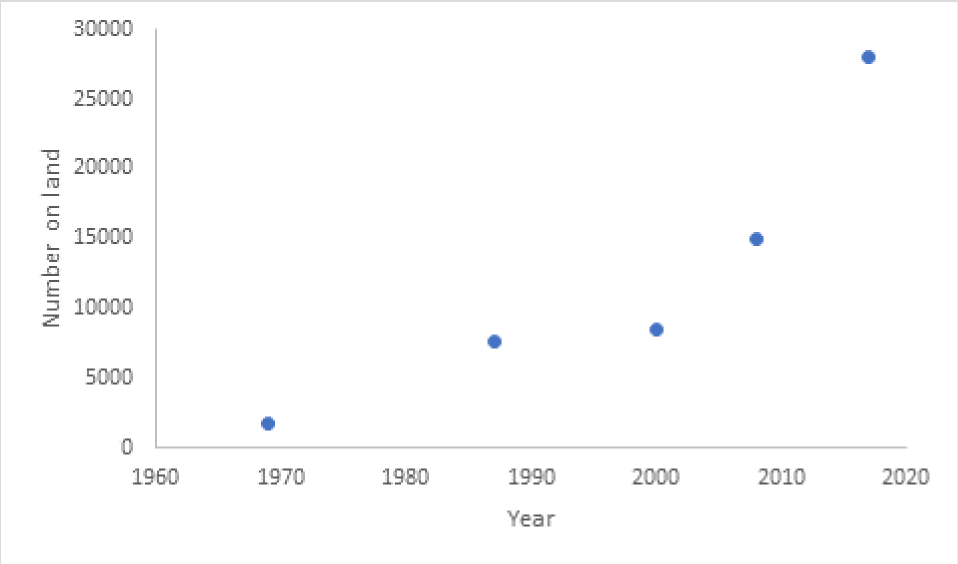



**Figure 33 Numbers of razorbills counted on land at Flamborough Head and Bempton Cliffs SPA (data from Lloyd et al. 2019).**

Like common guillemot, razorbill is an auk that uses its wings to propel itself underwater in pursuit of small fish. However, razorbill differs from common guillemot in many ways. Razorbill tends to make shallower dives, tends to feed more on sandeel and less on sprat compared with common guillemots at the same colony (evidence from Isle of May and Flamborough & Filey Coast), tends to feed on much smaller fish than fed to chicks by common guillemots, and carries multiple fish across the bill rather than a single fish inside the bill as carried by common guillemot. Razorbills also tend to nest in crevices and cavities under boulders rather than on cliff ledges, and often nest as individual pairs rather than in high density aggregations as seen in common guillemot. Woodward et al. (2019) list the foraging range of breeding razorbills as mean 61.3 km, mean maximum 88.7 km, maximum 313 km.

Like common guillemot, razorbill chicks fledge when only partly grown. However, razorbills tend to move further from their colonies than common guillemots, with some razorbills from UK colonies wintering off Iberia or Denmark, rather than in UK waters. These differences in ecology are important as they lead to the two species facing somewhat different threats and pressures. Wrecks of razorbills can occur in autumn and winter. These are thought mainly to involve juvenile birds, and to relate to local or regional scarcity of forage fish. However, wrecks often affect either razorbill or common guillemot but not necessarily both species together. Severe weather may be involved but that is certainly not always the case. Toxic chemicals may also affect survival, especially when birds are starving so that mobilisation of lipids increases contaminant concentration in the blood. However, legacy contaminant levels have been decreasing in razorbills and other UK seabirds so probably do not influence population trends. Razorbills are very vulnerable to oil pollution, but razorbills may winter in quite different areas from common guillemots from the same breeding site, so impacts may be quite different between the two species. Furthermore, there is no clear connection between oil spills and changes in breeding numbers. Oil risk has also decreased over recent decades so is unlikely to be having a strong influence on population trends. Reduction in forage fish abundance caused by fisheries for sandeels and sprats may affect survival of razorbills (Mitchell et al. 2004), and differences in prey preference may also result in these impacts differing between the two species, with razorbill perhaps being less dependent on fish and able to take more zooplankton. Climate change is considered to be one of the main threats, with increased stormy weather likely to affect breeding success and survival (Mitchell et al. 2004). Predation of eggs by ravens, crows, gulls and skuas is widespread, while gulls, especially great black-backed gulls and great skuas take fledging razorbill chicks and some chicks from nest sites. However, the generally more hidden nest sites of razorbills than common guillemots make razorbills less vulnerable to egg predation by birds. Predation by rats has been recorded and is likely to be more of a pressure on razorbills because most nest in sites accessible to rats rather than on less accessible cliff ledges. Razorbill flight heights suggest low risk of collision with offshore wind farm turbines, but there is strong evidence of some avoidance of offshore wind farms by razorbills.

### 5.2 Citation population size

The FFC SPA citation (dated August 2018) states that the site qualifies under Article 4.2 of the Birds Directive by supporting over 1% of the biogeographical populations of four regularly occurring migratory species: kittiwake, gannet, guillemot and razorbill. The site held 10,570 pairs of razorbills in 2008-2011, representing 2.3% of the subspecies *Alca torda islandica*. This estimate of the number of pairs is derived from the mean count of individual razorbills on land in 2008-2011 (15,776 individuals) multiplied by a correction factor of 0.67 to translate to breeding pairs. The estimate of the population of *Alca torda islandica* is from AEWA (2012): 1,380,000 individuals, translated to pairs by dividing this total by 3 (NE 2018), giving 460,000 pairs.

### 5.3 Conservation objectives

The site’s conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site.

More detailed conservation objectives have since been added online, last updated 13 March 2020 (Natural England 2020). For razorbill at FFC SPA these are:

- Maintain the size of the breeding population at a level which is above 10,570 breeding pairs whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Maintain safe passage of birds moving between nesting and feeding areas;
- Restrict the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding);
- Maintain the distribution, abundance and availability of key food and prey items (eg. Sandeel, sprat, krill) at preferred sizes;
- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;
- Maintain water quality and specifically mean winter dissolved inorganic nitrogen (DIN) at a concentration equating to High Ecological Status (specifically mean winter DIN is < 12 μM for coastal waters), avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for FFC pSPA was published in February 2015 (NE 2015). That identified public access/disturbance as a threat to razorbills and identified prevention of disturbance as a responsibility of East Riding of Yorkshire Council, Natural England, RSPB, Scarborough Borough Council, Yorkshire Wildlife Trust, and Flamborough Management Scheme. No other threats or pressures affecting razorbills at FFC SPA were specifically identified as requiring management in the SIP.

### 5.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are two main sources of impact on razorbills from offshore wind farm development: displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of razorbills from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. That impact on survival may be a carry-over effect on reduced winter survival as birds are in poorer condition at the end of the breeding season than would have been the case in the absence of the wind farm. There is a week relationship between the condition (body mass) of razorbills at the end of the breeding season and their subsequent overwinter survival (Daunt et al. 2020).

The maintenance of the population of each of the qualifying features could be affected indirectly through impact to energy budgets from displacement and barrier effects.

### 5.4 Location of compensation

Cook et al. (2011) defined four Ecological Assessment Areas for razorbill (Figure 34). The FFC SPA occurs within EAA2. Consequently, the hierarchy of the locations of compensation are:

1. FFC SPA;
2. EAA 2; and
3. All other EAAs in the UK.

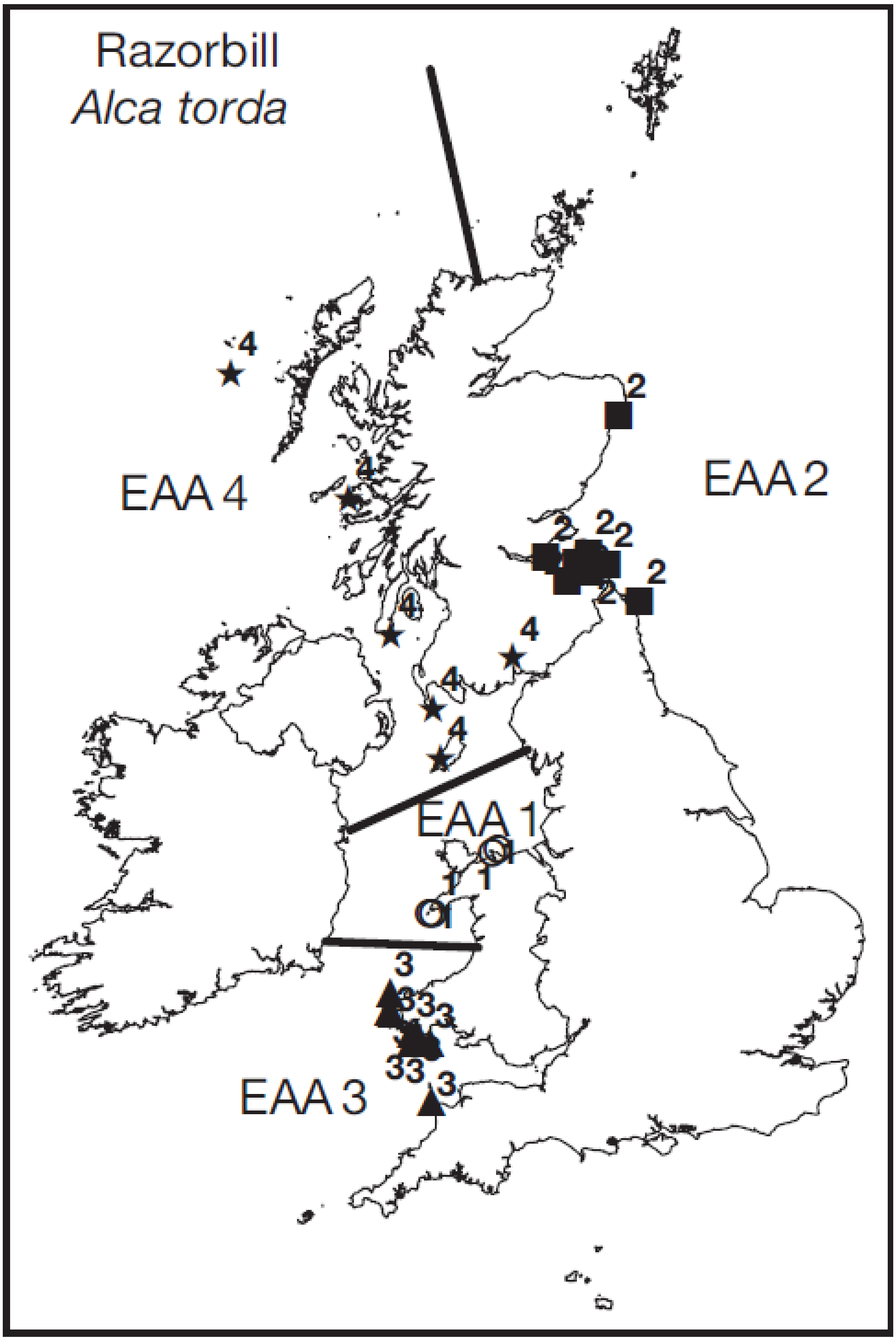



**Figure 34 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for razorbill by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

## 5.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding razorbills were developed based on the four potential compensation measures reviewed by Furness et al. (2013). The four potential measures listed were:

1. Closure of sandeel and sprat fisheries in all UK waters;
2. Closure of sandeel and sprat fisheries in razorbill wintering areas;
3. Rat eradication; and
4. Prevent oil spills.

None of these potential measures was considered highly likely to be effective with high confidence in that assessment based on evidence. It was recognised that strong efforts are already made to prevent oil spills, so that this was unlikely to be a practical option, but also that the wide distribution of razorbills beyond UK waters in the nonbreeding season also made this less effective for this species. While there was strong evidence that closure of sandeel and sprat fisheries would benefit related seabird species, there was a lack of evidence in this regard specifically for razorbill. There was a lack of clear evidence that this species would benefit from eradication of rats, but that was considered a highly practical measure if new evidence indicated this to be an effective measure at some colonies. The key biological questions for compensation measures for razorbills at FFC SPA are provided in Table 78.

**Table 78 Key Biological Questions in assessing the potential for compensation of breeding razorbill at FFC SPA.**

| No. | Key Biological question |
|---|---|
| **Closure of sandeel or sprat fisheries** | |
| 1 | Are razorbills sensitive to prey availability in the vicinity of their colony? |
| 2 | Does sandeel stock biomass affect razorbill survival? |
| 2.1 | At FFC SPA? |
| 2.2 | At EAA 2? |
| 2.3 | All other EAAs in the UK? |
| 3 | Do adult razorbills forage within areas subject to a high level of sandeel mortality from fisheries? |
| 3.1 | At FFC SPA? |
| 3.2 | At EAA 2? |
| 3.3 | All other EAAs in the UK? |
| 4 | Would management, or closure, of sandeel fisheries within the foraging areas of adult razorbills result in greater availability of forage fish for adult razorbills? |
| 4.1 | At FFC SPA? |
| 4.2 | At EAA 2? |
| 4.3 | All other EAAs in the UK? |
| **Eradication of rats and other invasive mammal predators** | |
| 1 | Is there evidence that eradication of rats from razorbill colonies increases the population size? |
| 1.1 | At FFC SPA? |
| 1.2 | At EAA 2? |

| No. | Key Biological question |
|---|---|
| 1.3 | All other EAAs in the UK? |
| 2 | Is there evidence of rats on offshore islands that include breeding razorbill SPAs? |
| 2.1 | At FFC SPA? |
| 2.2 | At EAA 2? |
| 2.3 | All other EAAs in the UK? |

### 5.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 5.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

#### 5.6.1 Closure of sandeel or sprat fisheries:

There is evidence that razorbill return rates (a suitable proxy for adult survival) to the Isle of May was higher when sandeel stock biomass in ICES Area 4 was higher. However, that relationship is not fully analysed or published at the time of writing. In a study of common guillemots and razorbills, Hentati-Sundberg et al. (2020) highlighted the importance of maintaining sufficient prey densities in the vicinity of the colony, suggesting that fine-scale spatial fisheries management is necessary to maintain high seabird breeding success. They also emphasised that there can be differences in this regard between similar species at the same location. Despite foraging on the same prey, razorbills could breed successfully at lower prey densities than guillemots but needed higher densities for self-maintenance, emphasizing the importance of considering species-specific traits when determining sustainable forage fish densities for top predators. They concluded that in their study case, densities of forage fish corresponding to the current fisheries management target $B_{MSY}$ were sufficient for successful breeding, and that the fisheries management target for conserving seabirds proposed by Cury et al. (2011), 1/3 of historical maximum prey biomass (B1/3), was also sufficient.

Glew et al. (2019) found that behavioural responses of puffins and razorbills differed in response to low forage fish availability in winter in the North Sea between two contrasting winter conditions. Razorbills' trophic position increased in the winter characterised by low survival and poor condition and the population foraged in more distant southerly waters of the North Sea compared with the winter characterised by higher survival and more favourable conditions.

##### *5.6.1.1 Answers to the key biological questions (5.5).*

The answers to the key biological question in relation to compensation through closure of sandeel or sprat fisheries are shown in Table 79.

**Table 79 Answers to Key Biological Questions in assessing the potential for compensation through closure of sandeel or sprat fisheries.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Are razorbills sensitive to prey availability in the vicinity of their colony? | **Yes.** Hentati-Sundberg et al. (2020) highlighted the importance of maintaining sufficient prey densities in the vicinity of the colony, suggesting that fine-scale spatial fisheries management is necessary to maintain high seabird breeding success. |
| 2 | Does sandeel stock biomass affect razorbill survival? | |
| 2.1 | At FFC SPA? | **Likely**. There is no evidence from FFC SPA that razorbill survival is affected by sandeel stock biomass, but there is from the Isle of May. |
| 2.2 | At EAA 2? | **Yes.** There is evidence that razorbill return rates (a suitable proxy for adult survival) to the Isle of May was higher when sandeel stock biomass in ICES Area 4 was higher. However, that relationship is not fully analysed or published at the time of writing. |
| 2.3 | All other EAAs in the UK? | **Likely**. There is no evidence from any other razorbills colonies in the rest of the EAAs in the UK that razorbill survival is affected by sandeel stock biomass, but there is from the Isle of May. |
| 3 | Do adult razorbills forage within areas subject to a high level of sandeel mortality from fisheries? | |
| 3.1 | At FFC SPA? | **Yes.** Most of the Danish fishing effort on sandeels in UK waters is targeted at grounds on the western edge of Dogger Bank. The main sandeel fishing area in UK waters is around 100 km from FFC SPA. This is within the maximum foraging range of razorbills from FFC SPA (191 km excluding tracking from Fair Isle, Woodward et al. 2019) but beyond the mean of the maximum foraging range (73.8 km excluding tracking data from Fair Isle, Woodward et al. 2019). So, there is likely to be some connectivity between the FFC SPA razorbill population and the area where high levels of sandeel fishing occur, but the majority of birds likely forage much closer to the colony. |
| 3.2 | At EAA 2? | **Yes.** Most of the North Sea is either currently heavily fished for sandeels or has been in the past. The sandeel box off the east coast of Scotland provides some protection for sandeel stocks for razorbills, but this stock is currently fished. |
| 3.3 | All other EAAs in the UK? | **No**. There is no sandeel fishery in the seas around the other EAAs in the UK. |
| 4 | Would management, or closure, of sandeel fisheries within the foraging areas of adult razorbills result in greater availability of forage fish for adult razorbills? | |
| 4.1 | At FFC SPA? | **Likely**. Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. Hentati-Sundberg et al. (2020) concluded that the fisheries management target for conserving seabirds proposed by Cury et al. (2011), 1/3 of historical maximum prey biomass (B1/3), would be sufficient, but it is apparent the North Sea sandeel fishery is well below this level. |
| 4.2 | At EAA 2? | **Yes.** Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. Evidence from the closure of the sandeel fishery off the east coast of Scotland indicates that populations of razorbills are likely to increase following closure of the fishery. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 4.3 | All other EAAs in the UK? | **No.** There is no sandeel fishery in the seas around the other EAA's in the UK. |

### 5.6.2 Eradication of rats and other invasive mammal predators

Eradication of rats from Lundy resulted in razorbill breeding numbers increasing from 950 to 1,735 individuals and showing an increase in breeding distribution of this species on the island into areas that would have been accessible to rats, so the increase is attributed to the removal of the pressure of predation by rats (Booker et al. 2019). Luxmore et al (2019) attributed the sudden increase in razorbill abundance on Canna in 2006 to rat eradication on the island but noted that the lack of an increase since then was likely due to a shortage of food.

#### 5.6.2.1 *Answers to the key biological questions (4.5).*

The answers to the key biological question in relation to compensation through eradication of rats and other invasive mammal predators are shown in Table 80.

**Table 80 Answers to Key Biological Questions in assessing the potential for compensation through eradication of rats and other invasive mammal predators.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that eradication of rats from razorbill colonies increases the population size? | |
| 1.1 | At FFC SPA? | **No.** There is no evidence that rats are limiting the FFC SPA, so rat eradication would not result in increases in colony size. |
| 1.2 | At EAA 2? | **No.** There is no evidence from colonies in EAA 2. |
| 1.3 | All other EAAs in the UK? | **Yes.** Eradication of rats from Lundy resulted in razorbill breeding numbers increasing from 950 to 1,735 individuals and showing an increase in breeding distribution of this species on the island into areas that would have been accessible to rats, so the increase is attributed to the removal of the pressure of predation by rats (Booker et al. 2019). Luxmore et al (2019) attributed the sudden increase in razorbill abundance on Canna in 2006 to rat eradication on the island but noted that the lack of an increase since then was likely due to a shortage of food. |
| 2 | Is there evidence of rats on offshore islands that include breeding razorbill SPAs? | |
| 2.1 | At FFC SPA? | **No.** There is no evidence of rats occurring in the FFC SPA razorbill colony. |
| 2.2 | At EAA 2? | **Yes.** Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats were present. The islands with rats present and seabird colonies including razorbills in EAA 2 were:<br>Rousay, Orkney;<br>Unst, Shetland;<br>Inchkeith, Firth of Forth; |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| | | Hoy, Orkney;<br>Flotta, Orkney; and<br>Stronsay, Orkney. |
| 2.3 | All other EAAs in the UK? | **Yes**. Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats were present. The islands with rats present and seabird colonies including razorbill in all other EAAs in the UK were:<br>• Garbh Eilean and Eilean an Taighe, Shiants;<br>• Rathlin Island, Northern Ireland;<br>• Colonsay and Oronsay, Argyll and Bute;<br>• Tiree, Argyll and Bute; and<br>• Herm, Channel Islands.<br>Black rats occurred on the Shiant Islands and Herm, while the remain islands have brown rats. |

## 5.7 Population level assessment

### 5.7.1 Flamborough and Filey Coast SPA

All the population level assessments for FFC SPA razorbills were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult survival rate. For a population size of 21,140 individuals and an adult survival rate of 0.895 a 1% increase in baseline mortality would be 22.2 additional birds being killed per annum.

The medium impact scenario was based on a pro-rata impact for an additional 7GW of capacity for Round 4 offshore wind farm development. The high impact scenario was based on a pro-rata impact from an additional 74GW of capacity for a 2050 net zero target. This resulted in medium impact scenario of 134 birds and a high impact scenario of 1,412 birds (Table 81).

**Table 81 Values for low, medium and high impact scenarios for razorbills at FFC SPA.**

| Impact scenario | Low | Medium | High |
|---|---|---|---|
| Additional mortality (birds) | 22.2 | 134 | 1412 |
| Additional mortality (rate) | 1% | 6.0% | 63.6% |

The PVA was parameterised using the values in Table 82.

**Table 82 PVA input parameters baseline vs impact scenarios.**

| Model parameter | Parameter values | | | Source |
|---|---|---|---|---|
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 5 | | | PVA app default |
| Upper constraint on productivity | 1 chick per pair | | | PVA app default |
| Initial population size | 21,140 individuals in 2011 | | | SACO |
| Productivity rate per pair | mean: 0.72, sd: 0.1050 | | | Flamborough plots from Aitken et al. (2017) |
| Adult survival rate | mean: 0.894, sd: 0.07 | | | PVA app “Isle of May” default value |
| Age class 0 to 1 | mean: 0.63, sd: 0.07 | | | PVA app “Isle of May” default value |
| Age class 1 to 2 | mean: 0.63, sd: 0.07 | | | PVA app “Isle of May” default value |
| Age class 2 to 3 | mean: 0.894, sd: 0.07 | | | PVA app “Isle of May” default value |
| Age class 3 to 4 | mean: 0.894, sd: 0.07 | | | PVA app “Isle of May” default value |
| Age class 4 to 5 | mean: 0.894, sd: 0.07 | | | PVA app “Isle of May” default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Impact on adult survival rate | Low | Medium | High | Calculated as above |
| | 0.000525071 | 0.003169347 | 0.0333964 | |

The baseline population projection was compared with the three impact scenarios (Table 81). The model projected that the baseline, low, medium, and high impact scenario populations would all decrease (Figure 35).

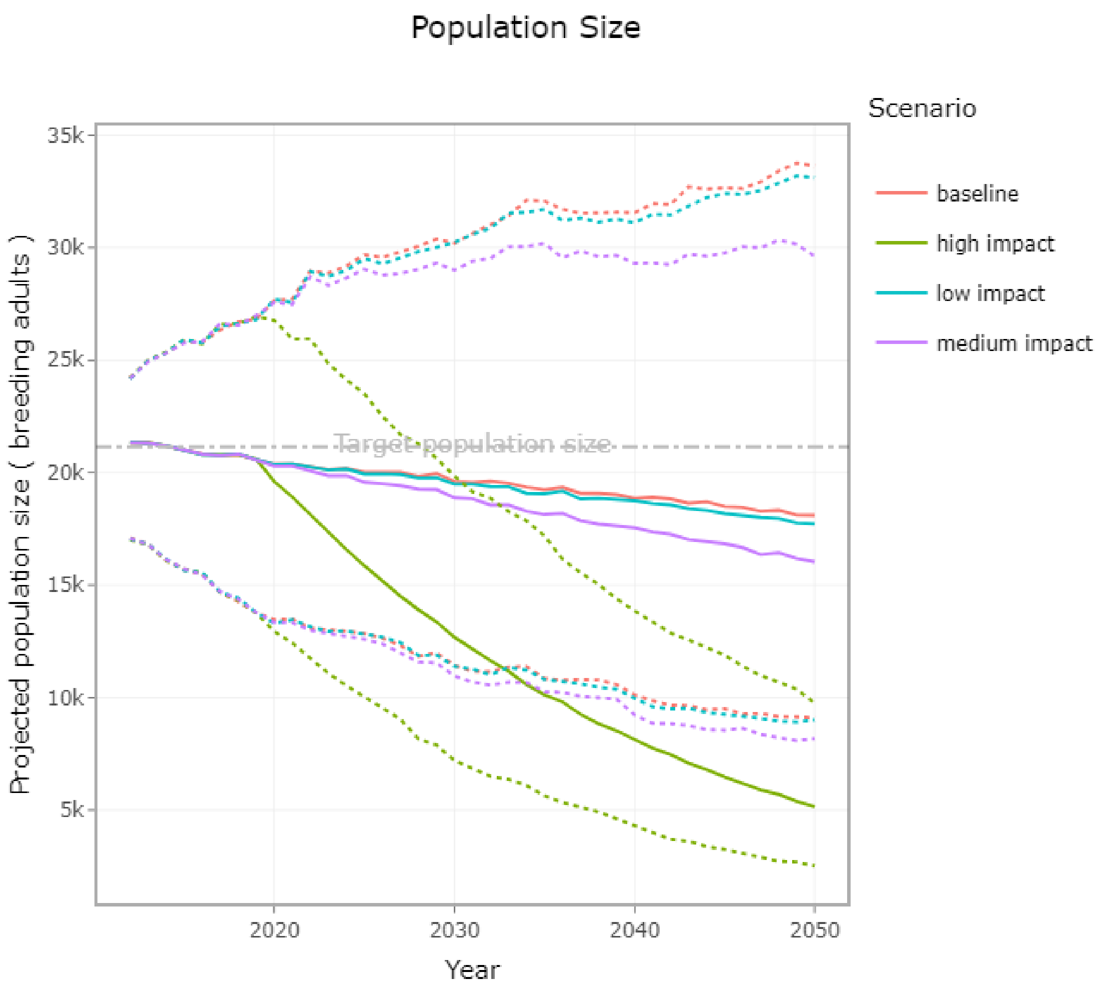



**Figure 35 Population projections for the baseline and low, medium and high impact scenarios.**

The counterfactuals of population size showed potentially important impacts on the population at the medium impact level and clearly important impacts at the high impact level. At low impacts the CPS was close to one (no impact). The counterfactuals of growth rate show less important impacts for both the low and medium impact scenarios. The high impact scenario had a CGR that was clearly problematic.

**Table 83 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9815 | 0.9816 (0.9441 - 1.0182) | 0.9994 | 0.9994 (0.9983 - 1.0005) |
| Medium | 0.8903 | 0.8905 (0.8568 - 0.9253) | 0.9963 | 0.9962 (0.9950 - 0.9974) |
| High | 0.2864 | 0.2861 (0.2702 - 0.3014) | 0.9605 | 0.9604 (0.9587 - 0.9620) |

### 5.7.2 Closure of sandeel or sprat fisheries

There was no empirical evidence to derive potential compensation scenarios for razorbills. While closure of the sandeel or sprat fishery in UK waters would potentially result in important benefits to the breeding razorbill population at FFC SPA, these cannot be tested using a population modelling approach. In addition, the baseline population model projects a population decline using the best available input parameter values (Figure 35). However, the razorbill population at FFC SPA has been increasing strongly for some time (Figure 36). This difference may be due to the assumption of a closed population being inappropriate, sampling of productivity from the FFC SPA colony providing an unrepresentative sample for the whole colony, or the relatively low survival to breeding age (0.2836) derived from the default, age-specific annual survival rates used in the model. All of these elements may be having an effect, but the low survival to breeding age seems at odds with the life history of the species.

While it may be possible to model some hypothetical scenarios on the effects of sandeel or sprat fisheries closures on razorbills there is so little empirical evidence that confidence in the assessment of the efficacy of these measures would be very low.

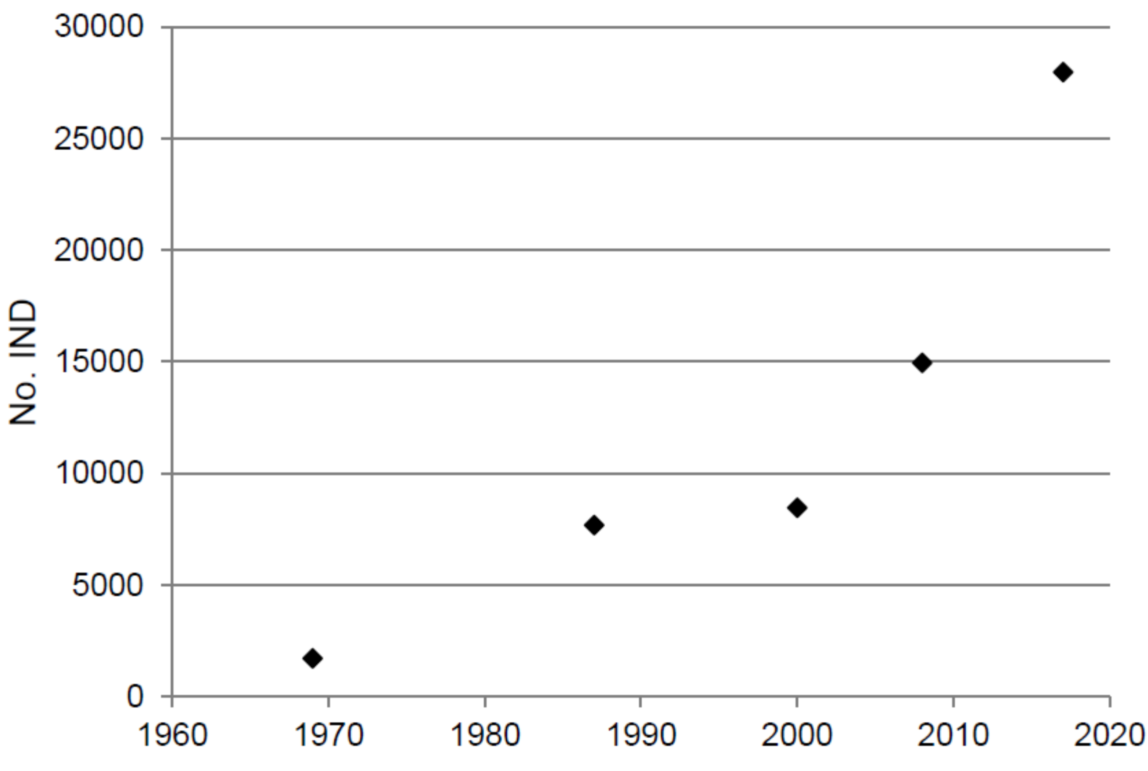



**Figure 36 Population counts of individual razorbills at FFC SPA (from Aitken et al. 2017).**

### 5.7.3 Eradication of rats and other invasive mammal predators

Population level assessment of the effects of rat and other invasive mammal predators on offshore islands on the FFC SPA razorbill population is not necessary. In the absence of any scope to deliver benefits to razorbills at FFC through predator eradication there, looking to enhance the population of razorbills at another site may be considered as an acceptable form of compensation with aim of maintaining the integrity of network of SPAs for breeding razorbills in the UK.

Stanbury et al. (2017) identified islands in UK where eradicating rats and invasive mammal predators may benefit breeding seabirds. For each of the 25 islands identified by Stanbury et al. (2017) those with an SPA where razorbill was a feature (either in its own right, or as a named feature of the breeding seabird assemblage) was identified (Table 84). For each island the citation population was recorded, and the most recent population size was determined from the SMP database. The numerical change in population size and percentage change since designation was calculated and the current Site Condition Monitoring status recorded. The presence of feral cats, brown and black rats, and American mink were determined using the information in Stanbury et al. (2017).

Some of the islands identified by Stanbury et al. (2017) either did not have areas on them designated as SPAs, or the seabird assemblage did not include razorbill as a named feature of the SPA breeding seabird assemblage. The absence of a SPA or razorbill appearing as a named feature of the assemblage did not necessarily mean the absence of breeding razorbills on that island. So, for those islands where there was not a SPA the SMP database was used to determine whether breeding razorbills were present, what their population size was around the Seabird 2000 count, what the most recent population count was and whether this had changed (Table 85).

**Table 84 Top 25 islands identified for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) and the SPAs designated for their razorbill populations, their citation population size, current population size, change in population size since designation, percentage change in population size since designation, current Site Condition Monitoring (SCM) status, and the presence of key invasive mammal predators. SCM status is highlighted as green if the population is Favourable and red if Unfavourable. Percent change is highlighted as green if the population has grown since designation and red if it has declined. SCM status is highlighted as green if the population is Favourable and red if Unfavourable.**

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Foula | Foula SPA | 6,200 | 559 in 2007 | -5,641 | -91 | UD | Y | N | N | N |
| Fair Isle | Fair Isle SPA | 3,400 | 1,930 in 2015 | -1,470 | -43 | UD | Y | N | N | N |
| Westray | West Westray SPA | 1,946 | 2,159 in 2017 | 213 | 11 | FM | Y | N | N | N |
| Garbh Eilean and Eilean an Taighe, Shiants | Shiant Isles SPA | 10,950 | 8,029 in 2015 | -2,921 | -27 | FR | N | N | Y | N |
| Rousay | Rousay SPA | Not a qualifying feature | | | | | Y | Y | N | N |
| Rathlin Island | Rathlin Island | 8,922 | 22,975 in 2011 | 14,053 | 158 | | Y | Y | N | N |
| Colonsay and Oronsay | North Colonsay & Western Cliffs SPA | Not a qualifying feature | | | | | Y | Y | N | N |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Unst | Hermaness, Saxa Vord & Valla Field SPA | Not a qualifying feature | | | | | Y | Y | N | N |
| Yell | None | | | | | | Y | N | N | N |
| Rum | Rum SPA | Not a qualifying feature | | | | | Y | N | N | N |
| Papa Westray | North Hill and Holm SPA | Not a qualifying feature | | | | | Y | N | N | N |
| Fetlar | Fetlar | Not a qualifying feature | | | | | Y | N | N | N |
| Inchkeith | None | | | | | | N | Y | N | N |
| Hoy | Hoy SPA | Not a qualifying feature | | | | | Y | Y | N | N |
| Flotta | None | | | | | | Y | Y | N | N |
| Tiree | None | | | | | | Y | Y | N | N |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Inchmarnock | None | | | | | | N | Y | N | Y |
| Stronsay | None | | | | | | Y | Y | N | N |
| Eilean Mhuire, Shiant Islands | Shiant Isles SPA | As above | | | | | N | N | Y | N |
| Gairsay | None | | | | | | Y | Y | N | N |
| North Ronaldsay | None | | | | | | Y | N | N | N |
| Muck | None | | | | | | N | Y | N | N |
| Housay, Out Skerries | None | | | | | | Y | Y | N | N |
| South Havra, Shetland | None | | | | | | Y | N | N | N |
| Herm, Channel Islands | None | | | | | | Y | Y | Y | N |

**Table 85 Islands identified for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) where the site was either not designated for their razorbill populations, or razorbill was not a named feature in the assemblage of more than 20,000 breeding individuals. Change and percent change is highlighted as green if the population is increasing, amber if decreasing slightly and red if decreasing severely.**

| Island | SPA | Seabird assemblage | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Rousay | Rousay | Y | 510 | 1999 | 469 | 2016 | -41 | -8.0% | Y | Y | N | N |
| Colonsay and Oronsay | North Colonsay & Western Cliffs SPA | Y | 2393 | 2000 | 2166 | 2018 | -227 | -9.5% | Y | Y | N | N |
| Unst | Hermaness, Saxa Vord & Valla Field SPA | Y | 617 | 2000 | 139 | 2016 | -478 | -77.5% | Y | Y | N | N |
| Yell | None | N | 12 | 1999 | 3 | 2018 | -9 | -75.0% | Y | N | N | N |
| Rum | Rum SPA | Y | 94 | 2000 | no count | no count | | | Y | N | N | N |
| Papa Westray | North Hill and Holm SPA | N | 195 | 1999 | 220 | 1999 | 25 | 12.8% | Y | N | N | N |
| Fetlar | Fetlar | Y | 47 | 1999 | no count | no count | | | Y | N | N | N |
| Inchkeith | None | N | 64 | 2000 | 131 | 2019 | 67 | 104.7% | N | Y | N | N |
| Hoy | Hoy SPA | Y | | | | | | | Y | Y | N | N |
| Flotta | None | N | 112 | 2002 | 267 | 2019 | 25 | 22.3% | Y | Y | N | N |
| Tiree | None | N | 384 | 1999 | 372 | 2018 | 25 | 6.5% | Y | Y | N | N |

| Island | SPA | Seabird assemblage | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Inchmarnock | None | N | no count | no count | no count | no count | | | N | Y | N | Y |
| Stronsay | None | N | no count | no count | no count | no count | | | Y | Y | N | N |
| Gairsay | None | N | no count | no count | no count | no count | | | Y | Y | N | N |
| North Ronaldsay | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Muck | None | N | no count | no count | no count | no count | | | N | Y | N | N |
| Housay, Out Skerries | None | N | 3 | 2001 | no count | no count | | | Y | Y | N | N |
| South Havra, Shetland | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Herm, Channel Islands | None | N | 2 | 1999 | 35 | 2015 | 25 | 1250.0% | Y | Y | Y | N |

There were many fewer breeding razorbill populations in SPAs on the islands identified by Stanbury et al. (2017) than for guillemot. Foula and Fair Isle SPA population had declined and were in unfavourable condition. The Shiants SPA had declined but was in favourable condition. Recently, rat eradication has been completed on all of the islands in the Shiants, so it is no longer available as a compensation location. Both the Westray and Rathlin Island populations had increased. Among the islands where there were no SPAs, or razorbill was not a named feature of the SPA, declines were noted on Rousay, Colonsay and Oronsay, and Unst. Small increases in the razorbill population were noted from Papa Westray, Inchkeith, Hoy, Flotta, Tiree, and Herm in the Channel Islands. While the declines in the colonies shown in Table 84and Table 85 are likely to be for a variety of reasons, it is possible that the presence of rats and other invasive mammal predators are a contributing factor. It is therefore possible that eradication of rats and other invasive mammal predators from these islands could contribute to the compensation for losses of breeding adult razorbills predicted for the FFC SPA population.

Comparing the low, medium and high impact scenarios (Table 86) with the declines shown in Table 84and Table 85 indicates that there is only potential to compensate for the high impact scenario through eradication of rats and other invasive mammal predators from Foula (or from more than one of the other sites). Islands with a change in their population size since designation smaller than the impact scenario could have lower potential to compensate. This is summarised in Table 86, but note that rat eradication has already been completed on the Shiant Islands. It is important to note, that these islands would likely have been designated with the same rat and other invasive mammal predator population as they have now, so it is likely that population would grow if predation pressure was reduced or removed, including those islands where the probability may be lower (i.e. Amber shaded in Table 86). Other islands not listed here would likely be unsuitable.

**Table 86 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios (Green = high probability, Amber = lower probability, Red = no probability).**

| Islands suitable for compensation | Compensation | | |
|---|---|---|---|
| | Low | Medium | High |
| | 22.2 | 134 | 1412 |
| Foula | | | |
| Fair Isle | | | |
| Westray | | | |
| Garbh Eilean and Eilean an Taighe, Shiants* | | | |
| Rathlin Island | | | |
| Rousay | | | |
| Colonsay and Oronsay | | | |
| Unst | | | |
| Yell | | | |
| Papa Westray | | | |
| Inchkeith | | | |
| Flotta | | | |

| Islands suitable for compensation | Compensation | | |
|---|---|---|---|
| | Low | Medium | High |
| | 22.2 | 134 | 1412 |
| Tiree | Amber | Amber | Amber |
| Herm, Channel Islands | Amber | Amber | Amber |
| * Invasive mammal predators already eliminated from the Shiants | | | |

#### 5.7.3.1 *Compensation ratios*

Increasing the compensation ratios to 1:3 and 1:6 resulted, unsurprisingly, in fewer potentially suitable islands being available for compensation (Table 87). The only island potentially suitable for high impacts was Foula at a 1:3 ratio. At the medium impact level four islands were probably or possibly suitable and at the low impact level there were five possible islands.

**Table 87 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios at 1:3 and 1:6 ratios (Green = high probability, Amber = lower probability, Red = no probability).**

| Islands suitable for compensation | Change | 1:3 | | | 1:6 | | |
|---|---|---|---|---|---|---|---|
| | | Low | Medium | High | Low | Medium | High |
| | | 66.6 | 402 | 4,236 | 133.2 | 804 | 8,472 |
| Foula | -5,641 | Green | Green | Amber | Green | Green | Red |
| Fair Isle | -1,470 | Green | Green | Red | Green | Green | Red |
| Westray | 213 | Red | Red | Red | Red | Red | Red |
| Garbh Eilean and Eilean an Taighe, Shiants* | -2,921 | Red | Red | Red | Red | Red | Red |
| Rathlin Island | 158 | Red | Red | Red | Red | Red | Red |
| Rousay | -41 | Red | Red | Red | Red | Red | Red |
| Colonsay and Oronsay | -227 | Green | Red | Red | Green | Red | Red |
| Unst | -478 | Green | Amber | Red | Green | Red | Red |
| Yell | -9 | Red | Red | Red | Red | Red | Red |
| Papa Westray | 25 | Red | Red | Red | Red | Red | Red |
| Inchkeith | 67 | Red | Red | Red | Red | Red | Red |
| Flotta | 25 | Red | Red | Red | Red | Red | Red |
| Tiree | 25 | Red | Red | Red | Red | Red | Red |
| Herm, Channel Islands | 25 | Red | Red | Red | Red | Red | Red |

* Invasive terrestrial mammals have already been eradicated from the Shiants.

### 5.7.4 EAA 2

#### 5.7.4.1 *Closure of sandeel or sprat fisheries*

Population level assessment was not completed for any other SPAs in the EAA 2 area. Nine SPAs designated for their breeding razorbill populations also occurred within EAA 2.

All of these colonies would be expected to benefit from a closure of UK water to sandeels, as all of these colonies occur within foraging range of sandeel stocks that are currently or previously depleted through fishing. The sandeel box off the east coast of Scotland has likely positively affected the razorbill populations in SPAs on the east coast of Scotland. It is important to note that the sandeel box is a fisheries management tool, not a seabird conservation tool, and that current management of the stock that includes the sandeel box allows a take of the stock that could deplete the stock despite the presence of the box. This stock has not experienced the same level of take since the box was put in place, though fishing occurred within the stock in 2021.

#### 5.7.4.2 *Eradication of rats and other invasive mammal predators*

The majority of islands identified in Table 87 occurred in EAA 2 (Table 67). Since eradication of mammals from FFC SPA is not a suitable compensation measure, the next level of preferred location would be the islands occurring in EAA 2 shown in Table 88.

**Table 88 EAA for islands identified as potentially suitable for compensation for razorbill through eradication of rats and other invasive mammal predators.**

| Islands suitable for compensation | EAA |
|---|---|
| Foula | 2 |
| Fair Isle | 2 |
| Westray | 2 |
| Garbh Eilean and Eilean an Taighe, Shiants* | 4 |
| Rathlin Island | 4 |
| Rousay | 2 |
| Colonsay and Oronsay | 4 |
| Unst | 2 |
| Yell | 2 |
| Papa Westray | 2 |
| Inchkeith | 2 |
| Flotta | 2 |
| Tiree | 4 |
| Herm, Channel Islands | 3 |

MacArthur Green

#### 5.7.5 All other EAAs in the UK

##### *5.7.5.1 Closure of sandeel or sprat fisheries*

The closure of UK waters to sandeel fisheries would be expected to mainly benefit colonies in areas that are currently, or have previously been, subject to sandeel fisheries. There are nine SPAs designated for their razorbill populations in all the other EAAs in the UK, but much of the areas of sea within the foraging range of birds in these SPAs has not experienced the same take of sandeels from stocks as those in the North Sea. It is therefore considered unlikely that closure of sandeel fisheries would have the same level of benefit to razorbill populations in EAA 2.

##### *5.7.5.2 Eradication of rats and other invasive mammal predators*

The islands with potential for compensation in all the other EAAs in the UK are shown in Table 88. These islands would be in the final tier of location hierarchy, with sites in EAA 2 preferred. Only five islands were identified and one of these (Shiant Islands) has already had invasive mammals eradicated.

### 5.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the two recommended compensation measures were undertaken for the closure of sandeel or sprat fisheries compensation approach and the eradication of rats and other invasive mammals approach. The summary table for the closure of sandeel or sprat fisheries compensation assessment is shown in Table 89, and the eradication of rats and other invasive mammals compensation assessment is shown in Table 90. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 92 (closure of sandeel or sprat fisheries) and in Table 94 (eradication of rats and other invasive mammals).

Table 89 Assessment of confidence in the closure of sandeel or sprat fisheries method to compensation.

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Importance of maintaining sufficient prey densities in the vicinity of the colony | n/a | Hentati-Sundberg et al. (2020) | Robust | Robust | n/a | Medium | ROBUST | LOW | HIGH |
| Fisheries management target for conserving seabirds as 1/3 of historical maximum prey biomass | B1/3 | Cury et al. (2011) | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Low winter survival when higher trophic levels unavailable | n/a | Glew et al. (2019) | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

Table 90 Assessment of confidence in the eradication of rats and other invasive mammals method to compensation.

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Eradication of rats increases breeding population size | n/a | Booker et al. 2019, Luxmoore et al. (2019) | Robust | Robust | Low | Robust | MEDIUM | HIGH | HIGH |
| Presence of invasive mammals on islands with razorbills | n/a | Stanbury et al. 2017 | Robust | Robust | Robust | Medium | MEDIUM | HIGH | HIGH |
| Rats reduce adult survival | n/a | n/a | Limited | Limited | n/a | Limited | LIMITED | LOW | VERY LOW |
| Rats reduce productivity | n/a | n/a | Limited | Limited | n/a | Limited | LIMITED | LOW | VERY LOW |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | OVERALL CONFIDENCE SCORE | MEDIUM |

**Table 91 Assessment of confidence in the eradication of rats and other invasive mammals approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Citation population size | n/a | SiteLink | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |
| Current population size | n/a | SMP database | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |
| Site Condition | n/a | SiteLink | Robust | n/a | n/a | Medium | MEDIUM | HIGH | MEDIUM |
| Presence of invasive mammals | n/a | Stanbury et al. 2017 | Robust | n/a | n/a | Medium | MEDIUM | HIGH | MEDIUM |
| | | | | | | | | OVERALL CONFIDENCE SCORE | HIGH |

**Table 92 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures method from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Importance of maintaining sufficient prey densities in the vicinity of the colony | Hentati-Sundberg et al. (2020) created a model to explore the effect of prey densities near the colony. As this result was based on a model output the type of evidence was assessed as medium. The study was based on a razorbill colony in the Baltic Sea, so the applicability was assessed as low. |
| Fisheries management target for conserving seabirds as 1/3 of historical maximum prey biomass | The study by Cury et al. (2011) was very large and examined the effects of fisheries management on a wide variety of seabirds and their prey. This metric was found to be true for a wide variety of seabirds, including auks. Thus, evidence was all robust and applicability high giving a very high overall confidence. |
| Low winter survival when higher trophic levels unavailable | The study by Glew et al (2019) provided a robust study, which was from the Isle of May, so assessed as medium applicability. Thus, the confidence score was high. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was low, medium or high. Confidence scores were high or very high. So, an overall score of high was given. However, the evidence available for razorbill was much more limited than for guillemot, with many fewer published studies. There was also insufficient information to parameterise the PVA assessment for razorbill. **Therefore, the overall confidence score was reduced to low.** However, it should be noted that unpublished data and the life history of the species suggest that sandeel and sprat fisheries closure would be highly likely to improve the adult survival and productivity of colonies constrained by low prey abundance. |

**Table 93 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation method from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Eradication of rats increases breeding population size | Two sources of the effects of rat eradication from offshore islands were found. Both found a response of razorbill populations to rat eradication. Booker et al. (2019) found a strong response on Lundy and Luxmoore et al. (2019) found a less strong response on Canna but attributed this to food supply and notes that rats were breeding on parts of the island they had previously deserted. Both studies were considered to provide robust quality of evidence, and both were robust types of evidence (numeric). However, with only two studies available the amount of evidence was low. Since both studies had similar results the consistency of evidence was robust. This resulted in an overall medium evidence score. Since this evidence would be directly applicable to colonies in the UK suggested for compensation, the applicability was scored high. This resulted in an overall evidence score of high. |

| Metric | Narrative |
|---|---|
| Presence of invasive mammals on islands with razorbills | Stanbury et al. (2017) was only used to identify potential islands for compensation. The quality, consistency and amount of evidence were all scored as robust, but the type of evidence was scored as medium as the study only determined the presence/absence of predators and not their absolute or relative abundance. This was considered sufficiently important to result in an overall evidence score of medium. Applicability was high as this evidence was for the islands that could be used for compensation. The overall confidence score was therefore high. |
| Rats reduce adult survival | There was no direct empirical evidence found that rats reduce the adult survival of razorbills, so this metric was assessed as having low evidence and low applicability giving a very low confidence score. |
| Rats reduce productivity | There was no direct empirical evidence found that rats reduce the productivity of razorbills, so this metric was assessed as having low evidence and low applicability giving a very low confidence score. |
| **OVERALL CONFIDENCE SCORE** | With high confidence in the evidence that rat eradication can increase razorbill breeding populations and high confidence in the evidence of rats occurring on islands with razorbill colonies there was a high confidence in this measure. The lack of evidence that rats predate adult razorbills, or their egg/chicks, was considered less important than the evidence that rat eradication may result in increases in razorbill populations. This is because the mechanism for population increase may not be due to direct predation but on the restriction of nesting location on offshore islands. Removal of rats may result in expansion of the colony into areas that were previously not used because of the presence of rats. |

**Table 94 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Citation population size | The citation population size was obtained from the Citation document from NatureScot SiteLink. This is a legal document, so the value is not in question. |
| Current population size | The current population size was the most recent whole SPA count available from the SMP database. The most recent count was used for each site, so these may be different years between different SPAs. However, these are the best available evidence, and the count method and source were considered robust. |
| Site Condition | Site Condition of each razorbill SPA population was obtained from the NatureScot SiteLink website. Site Condition Monitoring is a count-based methodology, but the assignment of condition scores is not based on site or species specific methodologies, so the type of evidence was scored as medium. |

| Metric | Narrative |
|---|---|
| Presence of invasive mammals | The presence of each key invasive mammal species on each island were obtained from Stanbury et al. (2017). This is a published source that has been peer reviewed so was considered to be robust with high applicability to the islands being addressed. |
| **OVERALL CONFIDENCE SCORE** | The confidence scores were very high with two medium scores. The applicability scores were high. Thus, an overall confidence score of high was determined. However, the assumption that changes in population size between citation counts and the most recent count were caused by invasive terrestrial predators is untested. Indeed, for many sites this is unlikely to be the only, or even main, cause of population decline. **Therefore, confidence in this metric needs to be reduced to medium.** |

In the absence of empirical data to support a response in productivity or adult survival of razorbills to changes in prey fish abundance no meaningful quantitative assessment of compensation could be undertaken. Due to that lack of evidence, confidence in the likely efficacy of this measure to deliver compensation for any impact level on razorbill at FFC was low.

The rat and other invasive mammal eradication measures had a medium confidence in the assessment process. The confidence in the ability to compensate for the three impact scenarios varied between islands (Table 86). The high and low probability values provided in Table 86 provide a suitable assessment in the confidence that invasive mammal eradication would be sufficient to compensate for the three impact scenarios tested. It seems highly likely that rat eradication on some of the islands identified by Stanbury et al. (2017) would result in positive effects on the razorbill population, and that there would be high confidence if these could be identified. Since these were not identified by Stanbury et al. (2017) and it was beyond the scope of this study to identify these islands, then the current confidence assessed as high, should be assessed as medium at present, as it would not be possible to assume that rat eradication on one of these islands would result in a population increase. Overall, confidence that rat eradication would benefit razorbill colonies is higher than for guillemot, so an overall confidence of medium is justified.

## 5.9 Future monitoring and adaptive management

### 5.9.1 Closure of sandeel or sprat fisheries

If compensation through closure of the UK sandeel fishery was applied, it would be important that suitable monitoring is put in place to demonstrate that this has been effective. This would follow the same approach for sandeel and sprat fisheries closure for other species described here.

With much less evidence to support the effect on razorbills directly there would likely need to be greater monitoring of the effects of fisheries closure on razorbill demographics, as well as some primary research (see below).

### 5.9.2 Eradication of rats and other invasive mammal predators

After a suitable location for rat eradication is selected it will be important to collect data on the current population size of the razorbill population on that island. At least one breeding season count using the SMP methodology should be completed to provide an up-to-date baseline condition for future comparison. In addition, evidence should be collected from the island, preferably around the colony, to demonstrate that rats or other invasive mammals are present. This would likely need to be carefully designed for the specific island and colony being considered for eradication. Suitable methods for collecting these data would be chew sticks, camera traps, live traps and kill traps. Following application of an eradication method suitable for the invasive mammals present (there may need to be more than one) it will be necessary to continue monitoring to ensure eradications method have been successful. Should either the targeted species, or other invasive mammal species, be detected further eradication measures would likely be necessary. Monitoring for the presence/absence of mammals should continue throughout the period of compensation. Details on the intensity and frequency of this monitoring would need to be site specific. Several important characteristics will need to be considered when determining this frequency and intensity. These include distance from nearest other mammal population (more remote island will need less monitoring), type and frequency of boat visits (islands only visited occasionally by small passenger only vessels are less exposed to re-introduction of invasive mammalian predators than more frequent visits by larger cargo vessels), presence and type of working agriculture on the island (islands with working farms are more likely to accidentally introduce invasive mammal predators than those with none), and presence and size of the human population living on the island (reintroduction risk of invasive mammals increases with the presence of human populations on the island).

Monitoring of the breeding adult population would be necessary to demonstrate that the colony increases in size following the removal of invasive mammal predators. Initially annual counts of the colony size are advised, with this frequency being reduced as recovery continues. The level of reduction in monitoring frequency should be related to the level of recovery observed. Monitoring should continue throughout the period compensation is required, at least periodically. If invasive mammal predators appear to have re-established then monitoring frequency should increase following additional eradication, particularly if the population was suppressed by the re-establishment.

#### 5.9.3 Future research

In comparison to guillemot, there was much less primary research available to build a picture of the effects of sandeel fisheries on razorbills, both individually and at a population scale. Analysis of existing data from the Isle of May may prove very valuable in determining the relationship between adult survival (using return rate) and sandeel stock levels. The same analysis could be applied to data on razorbill productivity.

While the current evidence on the effects of invasive mammals on offshore islands on razorbills is limited, the evidence is strong. Further research on the effects of invasive mammals on razorbill demographics would be valuable. Understanding the effects of rats, in particular, on the distribution of razorbills within offshore islands would also be valuable. Razorbills nest on both cliff ledges and in boulders and suitable scree, where they may be more vulnerable to rats and other invasive mammals. This may limit nesting opportunities and therefore population responses may be stronger where more of this type of nesting habitat is available on an island. Razorbill productivity also tends to be larger from nests in boulders than on cliff ledges (e.g. Lloyd, 1979), so the population response from islands with a greater proportion of this type of habitat could be stronger (in the absence of other population constraints).

## 5.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for razorbills at FFC SPA: closure of sandeel and sprat fisheries in all UK waters, closure of sandeel and sprat fisheries in razorbill wintering areas, rat eradication, and prevention of oil spills. Oil spills have become rare in the UK through both strong legislation and strong application of best practice, so it was concluded that this would no longer be a suitable compensation measure. Since Furness et al. (2013) was published some new evidence has been published on the benefits of fisheries closure and invasive mammal eradication. However, evidence was much more limited than was found for guillemot. Given the life history of the species and similarity with guillemot, it seems likely that fisheries closures would have a positive effect on populations, but this is more difficult to prove at present.

There was insufficient evidence to meaningfully parameterise a PVA assessment of the potential for closure of sandeel and sprat fisheries to either increase productivity or increase adult survival. So, while closure of these fisheries may be beneficial to the FFC SPA razorbill population confidence in any assessment of these would be very low.

Assessment of compensation through eradication of invasive mammal predators from offshore islands was able to identify several islands where the recorded decline in the population size of razorbills was larger than the impact scenarios being considered. On the assumption that these declines may have been driven largely or in part by the effects of mammalian predation, and that removal of that pressure would see those declines reversed, there can be a medium degree of confidence that implementing this measure at islands where the decline exceeds the impact level at FFC could compensate for the impact level being considered, albeit only at the wider SPA network level. There were fewer islands identified than for guillemot and the ability to compensate for the low, medium and high impact scenarios was also less than for guillemot. This was mostly because most razorbill colonies were much smaller in size than guillemot colonies at the same locations and more colonies were in favourable conservation status. It is important to note that not all of the islands identified as potential sites for compensation will be suitable and further research, including site visits, are likely necessary to identify locations where rat eradication could result in positive improvements for razorbill colonies.

Confidence was so low in the ability of fisheries closures to provide compensation that a PVA assessment was not attempted. This was due to a lack of empirical evidence on the responses to razorbills to increases in sandeel abundance. There was medium confidence in the ability of rat eradication from offshore islands to compensate for low, medium, or high impact scenarios across multiple islands, including those ranked most highly by Stanbury et al. (2017).

# 6 FLAMBOROUGH AND FILEY COAST SPA – BREEDING ATLANTIC PUFFIN

## 6.1 Conservation status Atlantic puffin

Atlantic puffin has an IUCN Red List classification of "Vulnerable" and the UK population was listed in BOCC 2 and 3 amber which was changed to red in BoCC 4. It is listed by the Birds Directive as a migratory species. The biogeographic population (subspecies *arctica*, in NW Europe) was estimated at 5,676,000 pairs, of which 580,000 pairs breed in Great Britain and 21,000 pairs in all-Ireland (Harris & Wanless 2004). National surveys found a 15% increase in breeding numbers in the UK from 1969 to 1986, and a 19% increase from 1986 to 2000 (JNCC 2020). JNCC SCM data (JNCC 2020) show that breeding numbers in England increased by a factor of four between the Operation Seafarer and Seabird Colony Register counts (1969 – 1986) and doubled again by 2000. However, counts at two of the largest colonies in England (Coquet and Farne Islands) have shown a mixture of declines and recoveries of differing amounts.

Stroud et al. (2016) identified that the SPA suite with breeding puffin as a designated feature has 20 qualifying sites in Great Britain, 16 in Scotland (Canna and Sanday SPA, Cape Wrath SPA; East Caithness Cliffs SPA; Fair Isle SPA; Forth Islands SPA; Flannan Isles SPA; Foula SPA; Hermaness, Saxa Vord and Valla Field SPA; Hoy SPA; Mingulay and Berneray SPA; North Caithness Cliffs SPA; North Rona and Sula Sgeir SPA; Noss SPA; St Kilda SPA; The Shiant Isles SPA; Sule Skerry and Sule Stack SPA), three in England (Coquet Island SPA; Farne Islands SPA; FFC SPA) and one in Wales (Skokholm, Skomer and Middleholm SPA, now known as Skomer, Skokholm and seas off Pembrokeshire SPA). The SPAs in Great Britain were estimated to hold 83% of the Great Britain breeding population of puffins present in 2000 (Stroud et al. 2016). One site in Northern Island also qualifies (Rathlin Island).

Apart from the marine extensions at some SPAs for loafing seabirds close to colonies, no sites were listed in the 3rd UKSPA review as designated as marine areas for puffins (Stroud et al. 2016). Since then, Outer Firth of Forth and St Andrews Bay Complex SPA, designated on 3 December 2020, and Northumberland Marine SPA include puffin as a breeding season feature.

Puffin colonies are very hard to count unless there is good access to the burrows. At FFC SPA puffins are nesting in gaps and cracks in the cliff, so accurate counts are extremely difficult. An estimate of 7,000 individuals was made in 1987, which was only 2,615 when repeated in 2000. Since then, counts of birds on the sea in the early morning at the start of the breeding season (March & April) in 2016 (2,267 – not including Filey), 2017 (2,879) and 2018 (4,279) highlight the variability in puffin numbers at the colony (Lloyd et al. 2019).

Puffin is a burrow-nesting auk which breeds in colonies that are often rather large, mostly on steep grassy slopes at the top or part way down sea cliffs. Unlike common guillemots and razorbills, puffin chicks grow slowly, put on large amounts of fat, and when they fledge, they normally leave the nest at night, without adults present, and disperse over the sea on their own. Puffins use their wings to propel themselves underwater in pursuit of small fish but tend to forage in the upper layers of the sea rather than diving deep. When breeding they feed a lot on sandeels but can take a variety of small fish and some marine invertebrates. After breeding, they move offshore, some crossing the Atlantic to Canada and Greenland. Woodward et al. (2019) listed the foraging range of breeding puffins as: mean 62 km, mean maximum 137 km, maximum 383 km.

### 6.2 Citation population size

The FFC SPA citation (dated August 2018) states that the site qualifies under Article 4.2 of the Birds Directive by supporting over 20,000 seabirds in any season. Puffin is a component of that seabird assemblage. SMP data suggested there were 980 breeding pairs in 2015 and Stroud et al. (2016) stated there were 958 pairs in 2008, but these estimates are probably very unreliable.

### 6.3 Conservation objectives

The site's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site.

More detailed conservation objectives have since been added online, last updated 13 March 2020 (Natural England 2020). For the seabird assemblage at FFC SPA (of which puffin is a part) these are:

- Maintain the overall abundance of the assemblage at a level which is above 216,730 individuals whilst avoiding deterioration from its current level as indicated by the latest peak mean count or equivalent;
- Maintain the species diversity of the bird assemblage (The total number of species (nine) comprising the seabird assemblage should not reduce over time);
- Restrict the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding);
- Maintain the distribution, abundance and availability of key food and prey items (eg. Sandeel, sprat, krill) at preferred sizes;

- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;
- Maintain water quality and specifically mean winter dissolved inorganic nitrogen (DIN) at a concentration equating to High Ecological Status (specifically mean winter DIN is < 12 μM for coastal waters), avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for FFC pSPA was published in February 2015 (NE 2015). That identified public access/disturbance as a threat to the seabird assemblage (including puffin) and identified prevention of disturbance as a responsibility of East Riding of Yorkshire Council, Natural England, RSPB, Scarborough Borough Council, Yorkshire Wildlife Trust, and Flamborough Management Scheme. No other threats or pressures affecting the seabird assemblage at FFC SPA were specifically identified as requiring management in the SIP.

#### 6.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are two main sources of impact on puffins from offshore wind farm development: displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of puffins from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. That impact on survival may be a carry-over effect on reduced winter survival as birds are in poorer condition at the end of the breeding season than would have been the case in the absence of the wind farm. There is a strong relationship between the condition (body mass) of puffins at the end of the breeding season and their subsequent overwinter survival (Daunt et al. 2020).

The maintenance of the population of each of the qualifying features could be affected indirectly through impact to energy budgets from displacement and barrier effects.

## 6.4 Location of compensation

Cook et al. (2011) did not define Ecological Assessment Areas for puffin. Therefore, the EEAs for razorbill were used as the closest similar species available assessed by Cook et al. (2011). The FFC SPA occurs within EAA2. Consequently, the hierarchy of the locations of compensation were:

1. FFC SPA;
2. EAA 2; and
3. All other EAAs in the UK.

## 6.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding puffins were developed based on the three potential compensation measures reviewed by Furness et al. (2013). The three potential measures listed were:

1. Closure of sandeel and sprat fisheries in all UK waters;
2. Rat eradication; and
3. Prevent oil spills.

None of these potential measures was considered highly likely to be effective with high confidence in that assessment based on evidence. Oil spill prevention is already a heavily regulated subject with good success in Europe in recent decades, so it was unlikely to be a useful source of compensation to the FFC SPA puffin population. In addition, Atlantic puffins are widely distributed across the North Atlantic during the nonbreeding season reducing the potential for oil spill impacts, and therefore compensation, to be important. While there was strong evidence that closure of sandeel and sprat fisheries would benefit similar species, there was a lack of evidence for puffin specifically. The key biological questions for compensation measures for puffins at FFC SPA are provided in Table 95.

**Table 95 Key Biological Questions in assessing the potential for compensation of breeding puffins at FFC SPA.**

| No. | Key Biological question |
|---|---|
| **Closure of sandeel or sprat fisheries** | |
| 1 | Is there evidence that sandeel stock biomass affects puffin populations? |
| 1.1 | At FFC SPA? |
| 1.2 | At EAA 2? |
| 1.3 | All other EAAs in the UK? |
| 3 | Is puffin adult survival negatively affected by declines in sandeel biomass? |
| 3.1 | At FFC SPA? |
| 3.2 | At EAA 2? |
| 3.3 | All other EAAs in the UK? |
| 4 | Is puffin productivity negatively affected by declines in sandeel biomass? |

| No. | Key Biological question |
|---|---|
| 4.1 | At FFC SPA? |
| 4.2 | At EAA 2? |
| 4.3 | All other EAAs in the UK? |
| 5 | Do adult puffins forage within areas subject to a high level of sandeel mortality from fisheries? |
| 5.1 | At FFC SPA? |
| 5.2 | At EAA 2? |
| 5.3 | All other EAAs in the UK? |
| 4 | Would management, or closure, of sandeel fisheries within the foraging areas of adult puffins result in greater availability of forage fish for adult puffins? |
| 4.1 | At FFC SPA? |
| 4.2 | At EAA 2? |
| 4.3 | All other EAAs in the UK? |
| **Eradication of rats and other invasive mammal predators** | |
| 1 | Is there evidence that eradication of rats from puffin colonies increases the population size? |
| 1.1 | At FFC SPA? |
| 1.2 | At EAA 2? |
| 1.3 | All other EAAs in the UK? |
| 2 | Is there evidence of rats on offshore islands that include breeding puffin SPAs? |
| 2.1 | At FFC SPA? |
| 2.2 | At EAA 2? |
| 2.3 | All other EAAs in the UK? |

## 6.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 6.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

### 6.6.1 Closure of sandeel or sprat fisheries

When the sandeel stock collapsed in Shetland, puffins were strongly affected. Like most other seabirds breeding in Shetland, puffins used to feed their chicks primarily on sandeels (Tasker & Furness 1996). There has been a general decline in numbers of breeding puffins at SPAs in Shetland. This decline includes numbers of breeding puffins at Fair Isle where reduced breeding success was shown to be caused by collapse of the Shetland sandeel stock, so that recruitment into the population was reduced (Miles et al. 2015). Breeding success of puffins in colonies in Shetland has been reduced much more than many other seabirds (JNCC 2020). This indicates that, like Arctic tern, kittiwake, and shag, puffins are sandeel specialist, at least while breeding in Shetland. It is apparent that puffins find switching to other prey species difficult and that they are unable to increase foraging effort to sustain breeding when sandeel abundance is low (see the annual Shetland Bird Reports and Fair Isle Bird Observatory Reports). This is supported by experimental evidence; at a colony with low forage fish abundance puffin chicks grew better when given supplementary food (Fitzsimmons et al. 2017). This shows that the conservation of puffin populations generally would benefit from increases in forage fish stocks. It is important to note that supplementary feeding would not be a practical compensation measure because populations are generally large, most puffin burrows at most colonies are inaccessible and where burrows are accessible disturbance in providing supplementary food to chicks would likely cause significant harm. Lindegren et al. (2018) indicated that improving forage fish stocks is best achieved by limiting fishing effort on those stocks. However, recovery can be slow and incomplete due to other ecosystem pressures on depleted stocks, including the higher natural mortality imposed on depleted stocks by top predators (Saraux et al. 2020).

Glew et al. (2019) found that behavioural responses of puffins and razorbills differed in response to low forage fish availability in winter in the North Sea. Puffin diet was significantly different between good and bad years of food availability, with a lower average trophic position in the winter characterised by lower survival rates. This implies that while razorbills move to other areas in search of their preferred forage fish prey, puffins are more likely to switch to invertebrate prey when forage fish are scarce. The lower survival of puffins in years with low sandeel abundance (Harris et al. 2005) suggests that moderate forage fish stock biomass need to be maintained in order to ensure sufficiently high survival of puffins. Thus, reducing the depletion of forage fish stocks in UK waters would improve puffin overwinter survival. Overwinter movements of 270 puffins were tracked from numerous colonies to compare movement patterns (Fayet et al. 2017). It was found that puffins from larger colonies or with relatively poorer local winter conditions undertook longer migratory movements and visited less-productive waters. This resulted in differences in flight activity and energy expenditure. Competition and local winter resource availability were important drivers of migratory movements and were most likely major drivers of adult survival. This further emphasises the importance of healthy stocks of forage fish within foraging range of puffin populations in both the breeding season and non-breeding season.

The diet of puffins in the non-breeding period has been assessed by examining the stomach contents of 176 puffins shot legally around the Faroe Islands in October-January (Harris et al. 2015b). The most frequent winter prey item was small sandeels, present in 82% of stomachs. Large sandeels were present in 32% of birds sampled. This further supports the evidence that auks feed on sandeels even in winter, even though sandeels spend much of the winter buried in sandy substrates.

#### *6.6.1.1 Answers to the key biological questions (6.5).*

The answers to the key biological question in relation to compensation through closure of sandeel or sprat fisheries are shown in Table 96.

**Table 96 Answers to Key Biological Questions in assessing the potential for compensation through closure of sandeel or sprat fisheries.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that sandeel stock biomass affects puffin populations? | |
| 1.1 | At FFC SPA? | No. There is no direct evidence from FFC SPA. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1.2 | At EAA 2? | Yes. When the sandeel stock collapsed in Shetland, puffins were strongly affected. There has been a general decline in numbers of breeding puffins at SPAs in Shetland. |
| 1.3 | All other EAAs in the UK? | No. There is no direct evidence from puffin colonies on all other EAAs in the UK. |
| 3 | Is puffin adult survival negatively affected by declines in sandeel biomass? | |
| 3.1 | At FFC SPA? | No. There is no direct evidence from FFC SPA or any other colony. |
| 3.2 | At EAA 2? | No. There is no direct evidence from EAA 2 or any other colony. |
| 3.3 | All other EAAs in the UK? | No. There is no direct evidence from all other EAAs in the UK or any other colony. |
| 4 | Is puffin productivity negatively affected by declines in sandeel biomass? | |
| 4.1 | At FFC SPA? | No. There is no direct evidence from FFC SPA. |
| 4.2 | At EAA 2? | Yes. Reduced breeding success was shown to be caused by collapse of the Shetland sandeel stock, so that recruitment into the population was reduced (Miles et al. 2015). |
| 4.3 | All other EAAs in the UK? | No. There is no direct evidence from puffin colonies on all other EAAs in the UK. |
| 5 | Do adult puffins forage within areas subject to a high level of sandeel mortality from fisheries? | |
| 5.1 | At FFC SPA? | Yes. Most of the Danish fishing effort on sandeels in UK waters is targeted at grounds on the western edge of Dogger Bank. The main sandeel fishing area in UK waters is around 100 km from FFC SPA. This is within the maximum foraging range of puffins from FFC SPA (383 km Woodward et al. 2019) and the mean of the maximum foraging range (119 km excluding tracking data from Fair Isle, Woodward et al. 2019). So, there is likely to be connectivity between the FFC SPA puffin population and the area where high levels of sandeel fishing occur. |
| 5.2 | At EAA 2? | Yes. Most of the North Sea is either currently heavily fished for sandeels or has been in the past. The sandeel box off the east coast of Scotland provides some protection for sandeel stocks for puffins, but this stock is currently fished. |
| 5.3 | All other EAAs in the UK? | No. There is no sandeel fishery in the seas around the other EAAs in the UK. |
| 4 | Would management, or closure, of sandeel fisheries within the foraging areas of adult puffins result in greater availability of forage fish for adult puffins? | |
| 4.1 | At FFC SPA? | Likely. Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. With longer maximum and mean maximum foraging ranges than the other auks it is more likely that closure of UK waters to sandeel fishers would increase the availability of forage fish to puffins from FFC SPA. |
| 4.2 | At EAA 2? | Likely. Closure of the sandeel fishery in the UK would likely result in increases in the sandeel stock in the North Sea, even if fisheries were displaced. With longer maximum and mean maximum foraging ranges than the other auks it is more likely that closure of UK waters to sandeel fishers would increase the availability of forage fish to puffins from colonies in EAA 2. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 4.3 | All other EAAs in the UK? | No. There is no sandeel fishery in the seas around the other EAAs in the UK. |

### 6.6.2 Eradication of rats and other invasive mammal predators

There are several new studies of rat eradication from offshore islands benefiting breeding puffin colonies. Puffin breeding numbers on Lundy increased from 5 individuals in 2004, the year rat eradication was completed, to 375 individuals in 2017 which has been attributed to the removal of predation pressure from rats (Booker et al. 2019). Rats were eradicated from Canna in 2005-2006. Before this, puffins had been confined to offshore stacks. Since rat eradication, puffins have recolonised sites on the mainland of Canna with more than 2,000 birds recorded in 2016, a 500% increase (Luxmoore et al. 2019). Puffins recolonised Ailsa Craig following eradication of rats (Zonfrillo 2002, 2007). There is, therefore, very clear evidence rat eradication from offshore islands can be highly beneficial for puffin populations.

#### *6.6.2.1 Answers to the key biological questions (6.5).*

The answers to the key biological question in relation to compensation through eradication of rats and other invasive mammal predators are shown in Table 97.

**Table 97 Answers to Key Biological Questions in assessing the potential for compensation through eradication of rats and other invasive mammal predators.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that eradication of rats from puffin colonies increases the population size? | |
| 1.1 | At FFC SPA? | No. There is no evidence that rats are limiting the FFC SPA, so rat eradication would not result in increases in colony size. |
| 1.2 | At EAA 2? | No. There is no evidence from colonies in EAA 2. |
| 1.3 | All other EAAs in the UK? | Yes. Puffin breeding numbers on Lundy increased from 5 individuals in 2004, the year rat eradication was completed, to 375 individuals in 2017 which has been attributed to the removal of predation pressure from rats (Booker et al. 2019). Rats were eradicated from Canna in 2005-2006. Before this, puffins had been confined to offshore stacks. Since rat eradication puffins have recolonised sites on the mainland of Canna with more than 2,000 birds recorded in 2016, a 500% increase (Luxmoore et al. 2019). Puffins recolonised Ailsa Craig following eradication of rats (Zonfrillo 2002, 2007). |
| 2 | Is there evidence of rats on offshore islands that include breeding puffin SPAs? | |
| 2.1 | At FFC SPA? | No. There is no evidence of rats occurring in the FFC SPA puffin colony. |
| 2.2 | At EAA 2? | Yes. Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats were present. The islands with rats present and seabird colonies including puffins in EAA 2 were:<br>• Foula, Shetland;<br>• Fair Isle, Shetland;<br>• Unst, Shetland; and |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| | | • Hoy, Orkney. |
| 2.3 | All other EAAs in the UK? | Yes. Stanbury et al. (2017) reviewed the available information on the presence of rats on offshore islands with seabird colonies. This showed that there were several islands where brown or black rats were present. The islands with rats present and seabird colonies including puffins in all other EAAs in the UK were:<br>• Garbh Eilean and Eilean an Taighe, Shiants;<br>• Rathlin Island, Northern Ireland; and<br>• Colonsay and Oronsay, Argyll and Bute. |

## 6.7 Population level assessment

### 6.7.1 Flamborough and Filey Coast SPA

All the population level assessments for FFC SPA puffins were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult mortality. For a population size of 980 pairs (from the Supplementary Advice on Conservation Objectives) and an adult survival rate of 0.907 a 1% increase in baseline mortality would be 1.8 additional birds being killed per annum.

The medium and high impact scenarios were based on the ratio of low to medium and high impact scenarios for razorbill (5.7). This resulted in medium impact scenario of 10.9 birds and a high impact scenario of 114.5 birds (Table 98)

**Table 98 Values for low, medium and high impact scenarios for puffins at FFC SPA.**

| Impact scenario | "Low" | "Medium" | "High" |
|---|---|---|---|
| Additional mortality (birds) | 1.8 | 10.9 | 114.5 |
| Additional mortality (rate) | 1% | 6.0% | 62.8% |

The PVA was parameterised using the values in Table 99. The age specific survival rates for puffins in the Seabird PVA Tool provide "National" default values that result in a very low survival rate from fledging to first breeding (0.21). This low survival rate always caused the population to decline when modelled and appears unrealistic based on the life history of the species (i.e. high adult survival and low productivity). Harris (1983) reported that the survival rate from fledging to first breeding on the Isle of May was 0.39. This was used to provide identical age specific survival rates of 0.8283, which resulted in an increasing baseline population. Tuning the model in this way appears to provide a reasonable population growth rate for the baseline population (1.034), and therefore provides a more reasonable comparison with the three impact scenarios.

**Table 99 PVA input parameters baseline vs impact scenarios.**

| Model parameter | Parameter values | | | Source |
|---|---|---|---|---|
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 5 | | | PVA app default |
| upper constraint on productivity | 1 chick per pair | | | PVA app default |
| Initial population size | 980 pairs in 2015 | | | SACO |
| Productivity rate per pair | mean: 0.7522, sd: 0.1289 | | | PVA app "National" default value |
| Adult survival rate | mean: 0.907, sd: 0.108 | | | PVA app "National" default value |
| Age class 0 to 1 | mean: 0.8283, sd: 0.077 | | | Calculated from survival to breeding age of 0.39 (from Harris 1983), s.d. from PVA app "National" default value. |
| Age class 1 to 2 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Age class 2 to 3 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Age class 3 to 4 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| | Low | Medium | High | |

| Model parameter | Parameter values | | | Source |
|---|---|---|---|---|
| Impact on adult survival rate | 0.000918367 | 0.005543298 | 0.05841147 | Calculated as above |
| **Sandeel fishery closure scenarios** | | | | |
| Impact on productivity rate | -0.06195076 | -0.1239015 | -0.2478 | Calculated as above |

The baseline population projection was compared with the three impact scenarios (Table 98). The model projected that the baseline, low impact and medium impact scenario populations would all increase, while the high impact scenario population would decrease (Figure 37).

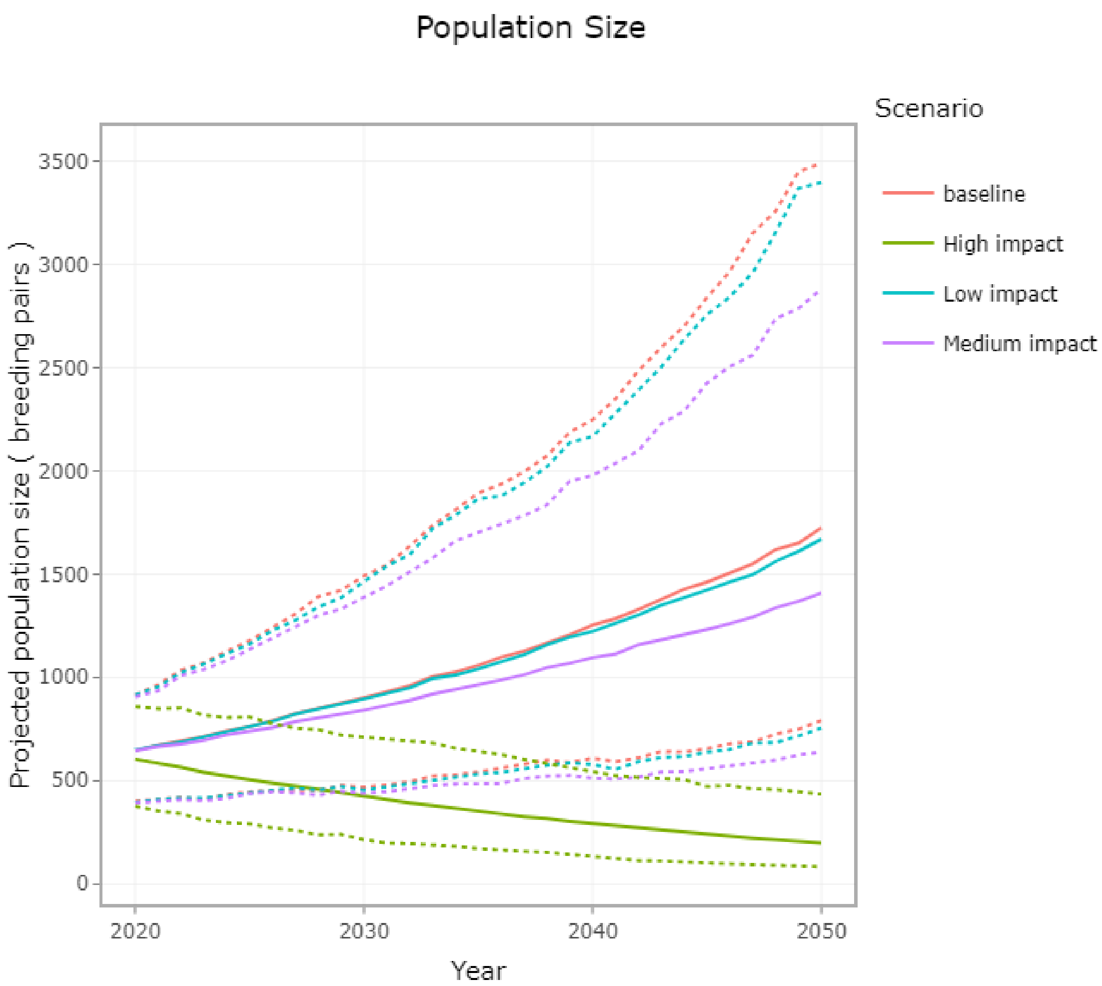



**Figure 37 Population projections for the baseline and low, medium and high impact scenarios.**

The counterfactuals of population size showed potentially important impacts on the population at medium impacts, and clearly important impacts at high impacts. At low impacts the CPS was close to one (no impact). The counterfactuals of growth rate show less important impacts for both the low and medium impact scenarios. The high impact scenario had a CGR that was clearly problematic.

**Table 100 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium, and high impact scenarios.**

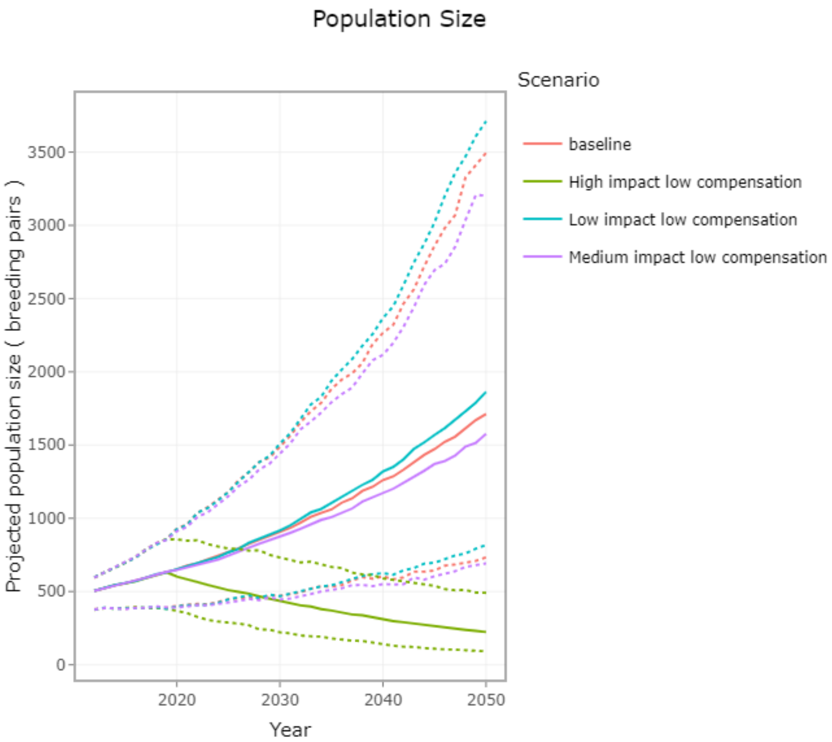
| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9662 | 0.9704 (0.8597 - 1.0940) | 0.9989 | 0.9989 (0.9958 - 1.0024) |
| Medium | 0.8216 | 0.8234 (0.7260 - 0.9264 | 0.9937 | 0.9937 (0.9904 - 0.9969) |
| High | 0.1165 | 0.1163 (0.0963 - 0.1366) | 0.9330 | 0.9328 (0.9274 - 0.9378) |

#### 6.7.1.1 *Closure of sandeel or sprat fisheries*

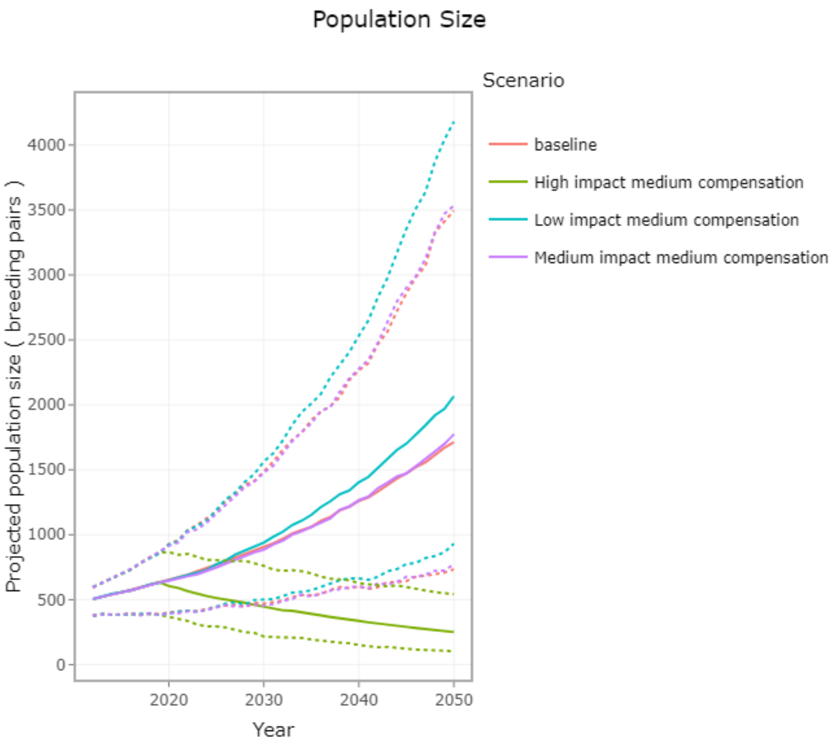
The effect of the decline of sandeels on puffin productivity in Shetland was shown by Miles et al. (2015). It is unknown whether the productivity of puffins at FFC SPA has been affected by the fisheries impact on sandeel abundance, but it seems likely it will have been reduced. However, it is also unknown what the productivity of puffins at FFC SPA is (or any other life history parameters). It was therefore necessary to use the best available estimate of puffin productivity, using the "National" default value in the Seabird PVA Tool. This value was 0.7522 chicks per pair. Since puffins only lay a single egg per breeding attempt, the maximum possible productivity increase would be 0.2478 chicks per pair. However, a productivity level of 1 chick per pair would be highly unlikely to occur. The highest productivity rate recorded in the SMP database for breeding puffins was a productivity of 0.91 chicks per pair from the Farne Islands in 2009. Thus, an increase of 0.1578 chicks per pair was assigned as the "high" compensation scenario. Two further scenarios were also tested. The "medium" compensation scenario assumed that productivity would be increased by half that of the high impact scenario. The "low" impact scenario was half that of the medium impact scenario. If these levels of increase in productivity cannot compensate for any of the three impact scenarios (Table 98) then compensation that increases productivity will not be effective.

**Potential effect of increasing productivity**

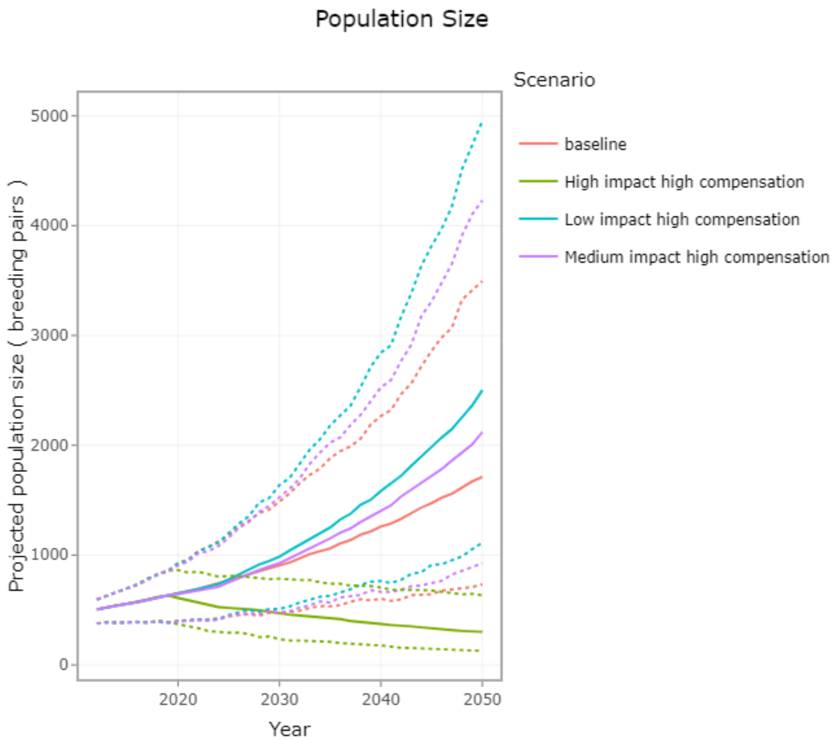
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on productivity (Figure 38).




Low compensation




Medium compensation




High compensation

**Figure 38 Projected population size of breeding puffins (pairs) at FFC SPA comparing baseline with the low, medium and high impact scenarios combined with low, medium and high compensation scenarios to increase productivity.**

For the low and medium impact scenarios the additional productivity from all compensation scenarios resulted in the projected population size increasing (Table 101). For all of the high impact scenarios the projected population growth was less than one for all compensation scenarios. The medium impact with low compensation still showed positive population growth, but the rate was lower than the baseline scenario. The population size increases shown are all likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, for the low and medium impacts the results suggested that even the low and medium increases in productivity that would be reasonably expected to follow sandeel fishery closure is likely to result in increases in the puffin population at FFC SPA. However, in all cases the high impact scenario was too great for the population to overcome, even at a maximum level of productivity (for the given levels of immature and adult survival).

**Table 101 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario. Scenarios where the median annual growth rate is greater than the baseline are shaded grey.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 1.0338 |
| Low | Low | 1.0365 |
| Low | Medium | 1.0399 |
| Low | High | 1.0463 |

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Medium | Low | 1.0307 |
| Medium | Medium | 1.0343 |
| Medium | High | 1.0403 |
| High | Low | 0.9678 |
| High | Medium | 0.9714 |
| High | High | 0.9770 |

There was no target population size set specifically for puffin at the FFC SPA, as it is not a qualifying feature in its own right but a component of the seabird assemblage. Consequently, no comparisons between scenarios on the ability of compensation to allow population to reach or exceed the conservation target were possible.

Examination of the counterfactuals of population size showed a larger difference in the CPS between the low and high, and medium and high, impact scenarios than between the low and medium impact scenarios, despite any level of compensation tested (Figure 39). This suggests that, for the survival parameters used in the model, the level of impact in the high scenario would never be compensated for, even if the productivity could be raised to the maximum likely value of 0.91 chicks per pair. It is also important to note that among low and medium impact scenarios, the median CPS for all but one scenario (medium impact/low compensation) was greater than one. For the medium impact low compensation scenario, the median CPS value was below one (0.9201) but the upper 95% confidence interval was above one (1.0300) indicating that, based on the model parameters used, the low and potentially also the medium impacts on survival could be effectively compensated by any of the levels of increased productivity assumed to arise from closure of sandeel fisheries as a compensation measure.

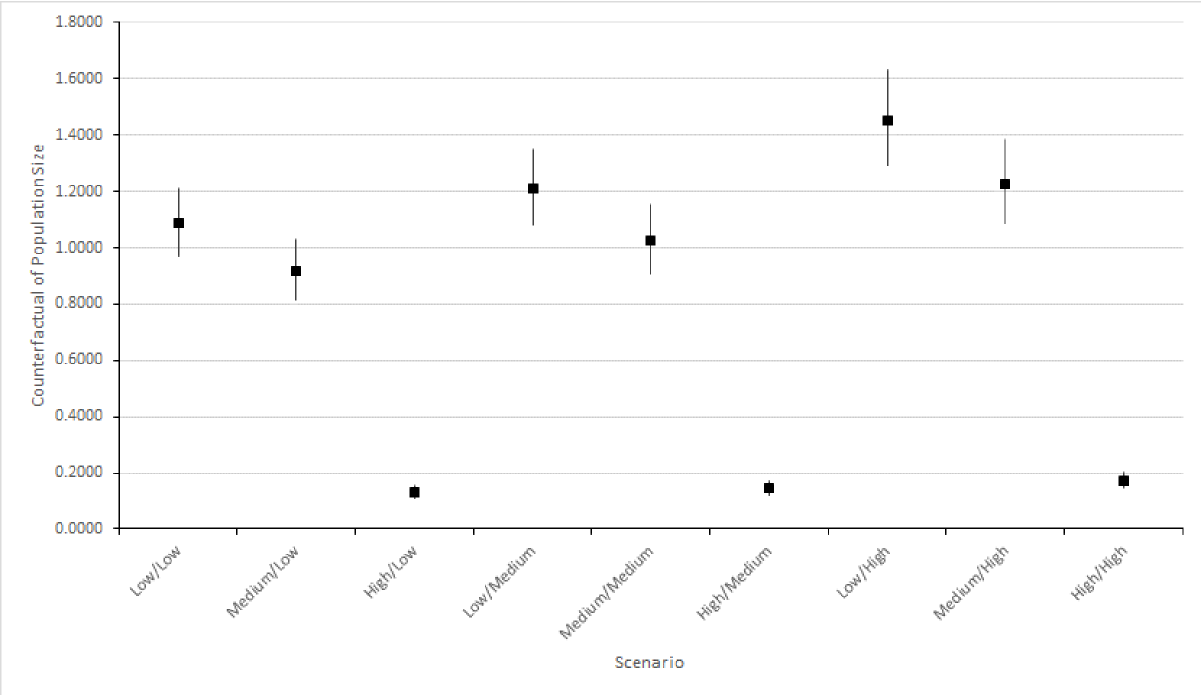



**Figure 39 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed a similar pattern to the CPS. There was a greater difference between the high impact scenarios and the low and medium impact scenarios than between the low and medium impact scenarios (Figure 40). Again, all the compensation scenarios were unable to overcome the negative effects of the high impact scenario. All of the high impact scenario CGR values were below one. The only low or medium impact scenario below one was also the medium impact low compensation scenario, but again this was only just below one (0.9973) and the upper 95% confidence interval was above one (1.0006).

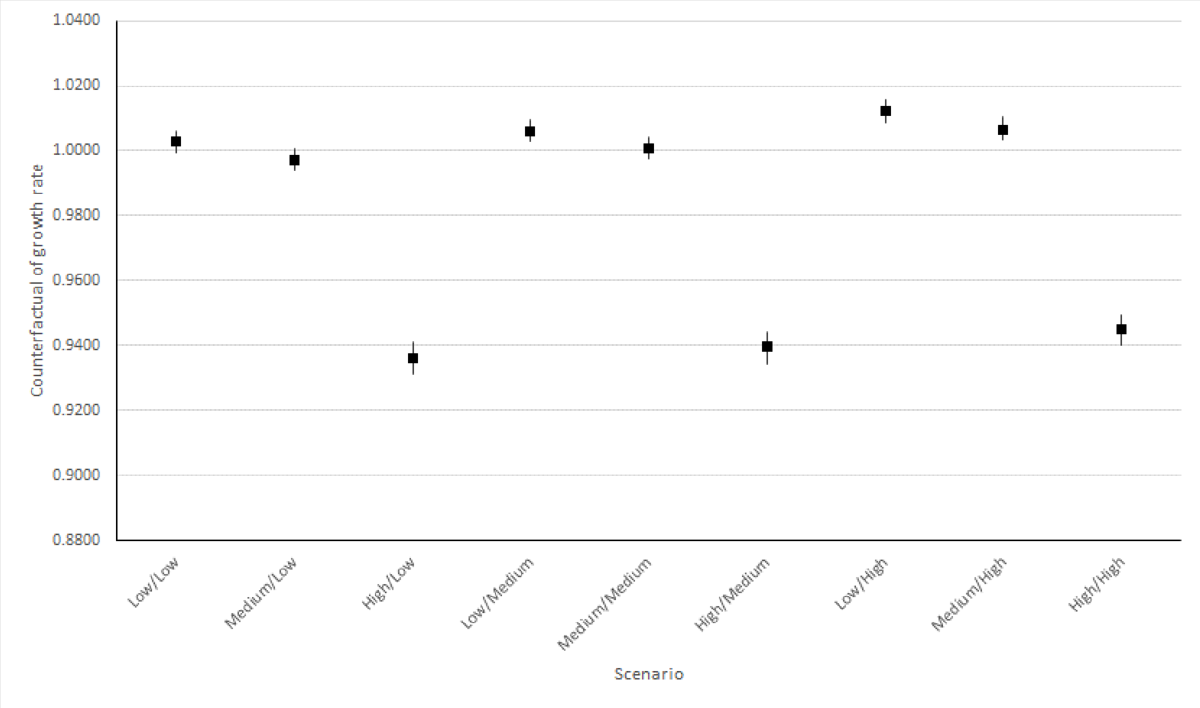


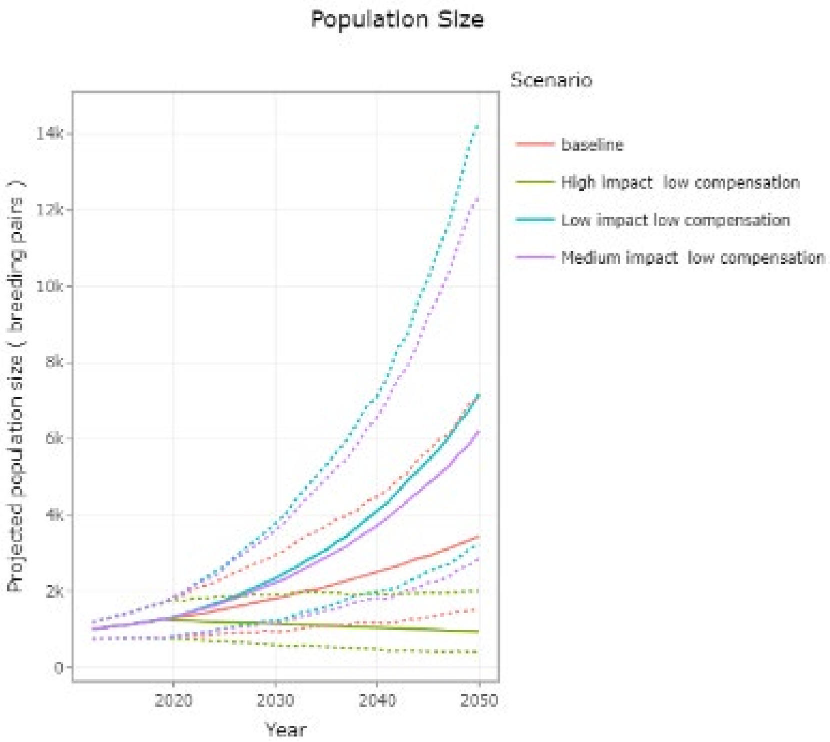

**Figure 40 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

The comparison of mean CGR values (and 95% confidence intervals) does suggest that it is likely that if closure of the sandeel fishery would result in any of the increased levels of puffin productivity considered here, that this would have positive benefits to the FFC SPA puffin population at low and medium impact levels, but at high impact levels compensation would be unlikely to be successful. However, it is important to note that all of the demographic parameters used in the model were not derived from the FFC SPA population. The productivity values used at baseline was relatively high (0.7522) compared with the maximum likely value of 0.91. So, increases in productivity were already constrained. If the productivity of the population is actually smaller than this, then greater improvements may be possible, but it would still be unlikely that the high impact scenario could be compensated for.

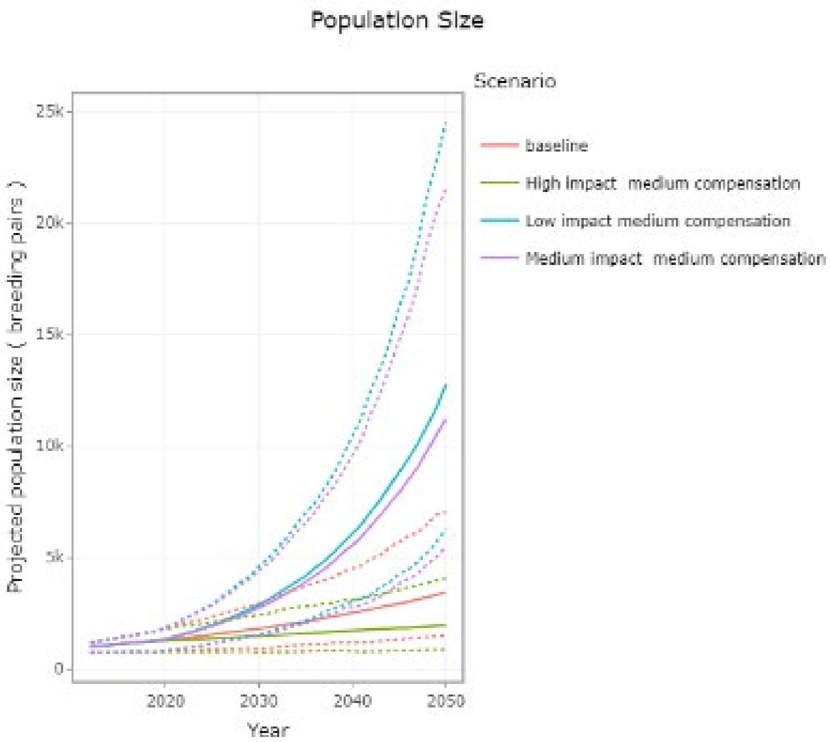
**Potential effect of increasing adult survival**

Since there is evidence of adult survival being reduced due to low sandeel stocks (Miles et al. 2015), it was assumed that the adult survival of puffins at FFC SPA could be increased if the sandeel and sprat fishery in UK waters was closed. Evidence of higher adult survival of puffins prior to the current high level of fisheries pressure on the North Sea sandeel stock can be obtained from adult survival rates obtained in the 1970's. The adult survival rate used in the population model here was 0.907. Harris et al. (2005) showed that there was very little variation in survival rates of puffin between five colonies in Wales, Scotland and Norway across the period from 1982 to 2002 (0.930 ± SE 0.005 for Skomer, 0.935 ± 0.006 for the Isle of May, 0.935 ± 0.022 for Fair Isle, 0.935 ±0.013 for Røst, and 0.935 ± 0.016 for Hornøya). Thus, an increase in adult survival to 0.93 was assumed to be a suitable "low" compensation scenario (under the assumption that the adult survival rate used in the baseline scenario was correct). Harris et al. (2005) cited studies in the 1970's showing higher adult survival of puffin at Skomer (0.95; Ashcroft 1979) and the Isle of May (0.975; Harris et al. 1997). These were assumed to be suitable medium and high compensation scenarios respectively.

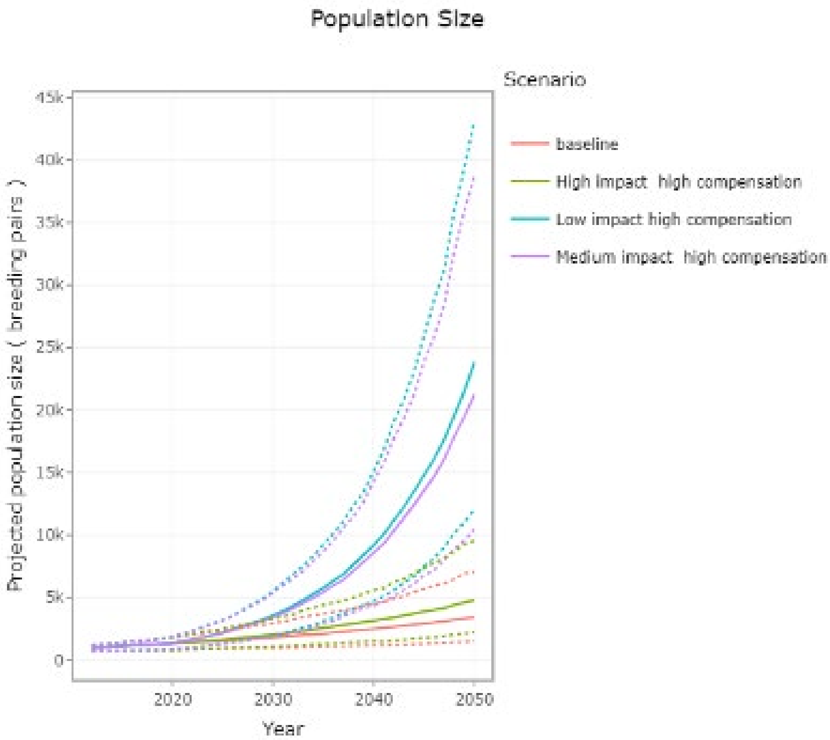
Projected population change of the baseline population (no impact) was compared with the low, medium, and high impact scenarios combined with the three potential levels of compensation on adult survival (Figure 41).




Low compensation




Medium compensation



High compensation

**Figure 41 Projected population size of breeding puffins (pairs) at FFC SPA comparing baseline with the low, medium and high impact scenarios combined with low medium and high compensation scenarios to increase adult survival.**

For only one scenario did the improved adult survival from all compensation scenarios result in the projected population size not increasing (Table 102); the high impact low compensation scenario. The projected annual population growth rate was more than one for all scenarios except the high impact low compensation scenario. The population size increases shown are all likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, the results suggested that if closure of the sandeel fishery would result in any of the increased levels of adult puffin survival considered here, that this would be likely to result in increases in the puffin population at FFC SPA unless the impact is high and the increase in adult survival achieved through compensation is low.

**Table 102 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 1.0337 |
| Low | Low | 1.0585 |
| Low | Medium | 1.0783 |
| Low | High | 1.0998 |
| Medium | Low | 1.0534 |
| Medium | Medium | 1.0741 |

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Medium | High | 1.0962 |
| High | Low | 0.9916 |
| High | Medium | 1.0153 |
| High | High | 1.0447 |

There was no target population size set specifically for puffin at the FFC SPA, as it is not a qualifying feature in its own right but a component of the seabird assemblage. Consequently, no comparisons between scenarios on the ability of compensation to allow population to reach or exceed the conservation target were possible.

Examination of the counterfactuals of population size showed a larger difference in the CPS between the low and high, and medium and high, impact scenarios than between the low and medium impact scenarios, despite any level of compensation tested (Figure 42). This suggests that, for the survival parameters used in the model, the level of impact in the high scenario would only be compensated for with a high compensation scenario (i.e. increasing the adult survival rate to 0.975). It is also important to note that among low and medium impact scenarios, the CPS for all compensation scenarios was greater than one.

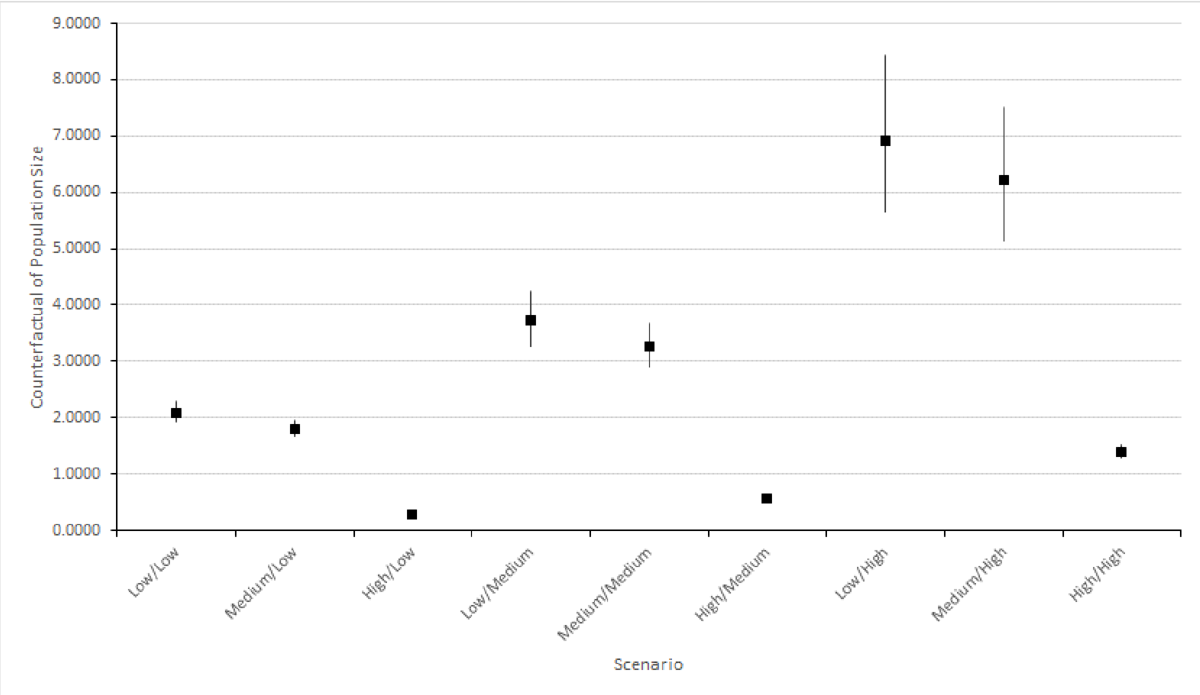



**Figure 42 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed a similar pattern to the CPS. There was a greater difference between the high impact scenarios and the low and medium impact scenarios than between the low and medium impact scenarios (Figure 43). The low and medium compensation scenarios were unable to overcome the negative effects of the high impact scenario. Only the high compensation scenario CGR value was greater than one when the impact scenario was high. It is also important to note that among low and medium impact scenarios, the CGR for all compensation scenarios was greater than one.

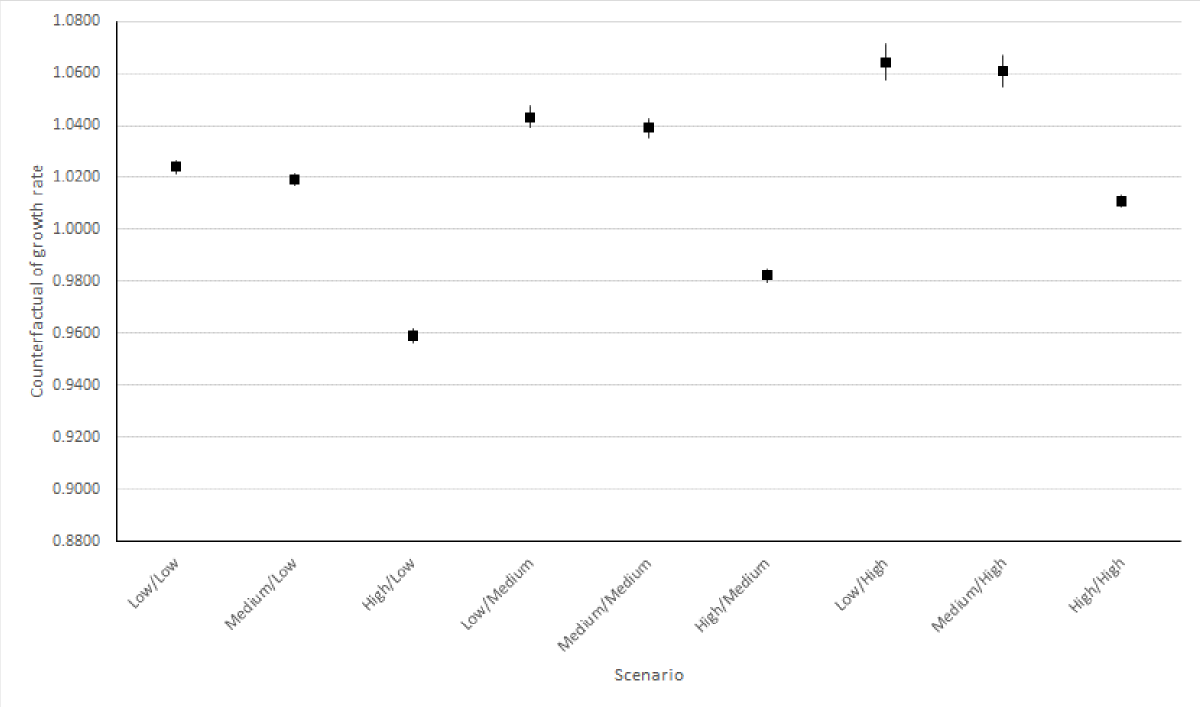



**Figure 43 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

The comparison of mean CGR values (and 95% confidence intervals) does suggest that if closure of the sandeel fishery would result in any of the increased levels of adult puffin survival considered here, that this would have positive benefits to the FFC SPA puffin population at low and medium impact levels, but at high impact levels compensation would be unlikely to be successful. However, it is important to note that all of the demographic parameters used in the model were not derived from the FFC SPA population. The adult survival value used was relatively low (0.907) compared with those shown in Harris et al (2005), but the adult survival of puffins at FFC SPA is unknown. If the adult survival rate in the FFC puffin population is currently higher than the value used this would limit the benefits of sandeel and sprat fisheries closures in UK waters.

Due to a lack of site-specific demographic parameters for the puffin population at FFC SPA it is difficult to draw strong conclusions on the potential benefit of sandeel and sprat fisheries closure from PVA modelling. However, the evidence of a benefit from other sites remains strong and so there remains a good likelihood of a benefit to the population, it is simply harder to quantify. Given the inaccessible characteristics of the nesting sites of puffins at FFC SPA it is unlikely that site specific data will be available in the future. However, monitoring of relative population change would be possible using the birds on the water at the colony counting method.

#### 6.7.1.2 *Compensation ratios*

**Potential effect of increasing productivity**

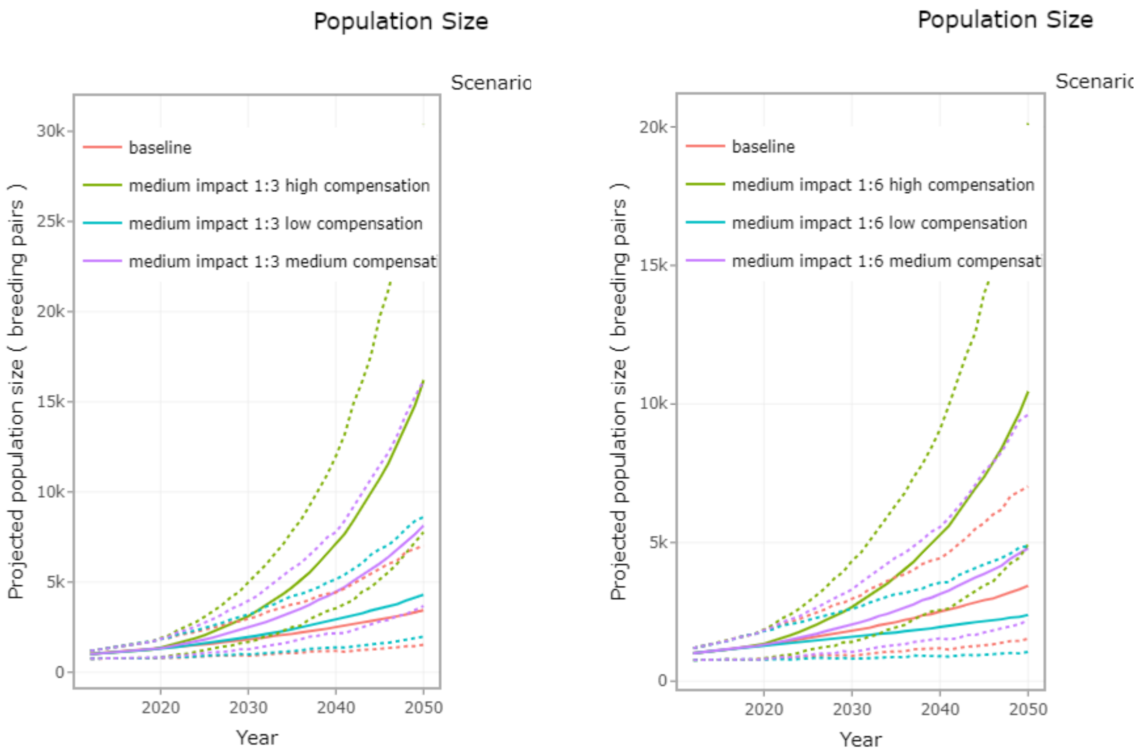
The levels of impact that compensation measures would need to overcome were calculated for 1:3 and 1:6 ratios (Table 103). The 1:1 ratio impacts were tested above and the High impact scenario was considered too large to be compensated for at any level. With the higher ratios, the high impact scenario at 1:3 and 1:6 was much larger and exceeded the high impact ratio at 1:1 and so could not be compensated. The medium impact level at both 1:3 and 1:6 was between the medium impact 1:1 ratio and the high impact 1:1 ratio. The low impact scenarios were below or equal to the medium impact scenario tested at 1:1, so these could be compensated from closure of the sandeel and sprat fisheries. Consequently, the medium impact scenarios at 1:3 and 1:6 were tested using a PVA for both increases in adult survival and increases in productivity. The low impact scenarios at both 1:3 and 1:6 was small enough that the testing above encompassed these levels, so PVAs were not run for these.

**Table 103 Low, medium, and high impact scenarios at 1:3 and 1:6 ratios.**

| Impact scenario | Ratio | Low | Medium | High |
|---|---|---|---|---|
| Additional mortality (birds) | 1:3 | 5.4 | 32.7 | 343.5 |
| Additional mortality (rate) | | 3% | 18% | 188.4% |
| Additional mortality (birds) | 1:6 | 10.8 | 65.2 | 687 |
| Additional mortality (rate) | | 6% | 36% | 374.8% |

**Potential effect of increasing adult survival**

Projected population change of the baseline population (no impact) was compared with the medium impact scenario at a 1:3 and 1:6 ratio combined with the three potential levels of compensation on adult survival (Figure 44).




Medium impact 1:3 | Medium impact 1:6

**Figure 44 Projected population size of breeding puffins at FFC SPA comparing the baseline with the medium impact scenarios at 1:3 and 1:6 ratios combined with low, medium, and high compensation scenarios.**

All of the scenarios tested resulted in positive population growth (Figure 44, Table 104). Only the low compensation compared to the 1:6 medium impact scenario was less than the projected baseline population growth rate. The CPS values were greater than one except the 1:6 ratio with low compensation, which was well below one. This was also the case with the CGR values, although the CGR value for the 1:6 ratio and low compensation was not much lower than one.

**Table 104 PVA metrics from assessment of the medium impact scenarios with 1:3 and 1:6 ratios.**

| Impact scenario | Compensation scenario | Median growth rate | CPS median (LCI – UCI) | CGR median (LCI – UCI) |
|---|---|---|---|---|
| Medium impact (1:3) | Baseline | 1.0338 | - | - |
| | Low | 1.0410 | 1.2477 (1,1515 – 1.3597) | 1.0071 (1.0049 – 1.0095) |
| | Medium | 1.0628 | 2.3706 (2.1464 – 2.6213) | 1.0282 (1.0254 – 1.0311) |
| | High | 1.0865 | 4.7117 (4.0193 – 5.4549) | 1.0513 (1.0462 – 1.0562) |
| Medium impact (1:6) | Baseline | 1.0338 | - | - |
| | Low | 1.0217 | 0.6932 (0.6373 – 0.7517) | 0.9883 (0.9859 – 0.9906) |
| | Medium | 1.0447 | 1.3973 (1.2931 – 1.5260) | 1.0109 (1.0088 – 1.0134) |
| | High | 1.0712 | 3.0182 (2.7018 – 3.3965) | 1.0363 (1.0331 – 1.0398) |

This assessment suggests that the compensation available from closing the UK sandeel and sprat fishery would likely be sufficient to overcome the medium impact scenario with a 1:3 ratio. At the 1:6 ratio it is unlikely that the low compensation scenario would be sufficient.

It is therefore possible to conclude that 1:3 ratio and 1:6 could be compensated with the medium and high compensation scenarios. The 1:6 scenario could not be compensated by the low compensation scenario but could by the medium and high compensation scenarios.

#### *6.7.1.3 Eradication of rats and other invasive mammal predators*

Population level assessment of the effects of rat and other invasive mammal predators on offshore islands on the FFC SPA puffin population is not necessary. In the absence of any scope to deliver benefits to puffins at FFC through predator eradication there, looking to enhance the population of puffins at another site may be considered as an acceptable form of compensation with aim of maintaining the integrity of network of SPAs for breeding puffins in the UK.

Stanbury et al. (2017) identified islands in UK by the benefit of eradicating rats and invasive mammal predators to breeding seabirds. For each for the 25 islands identified by Stanbury et al. (2017) those with an SPA where puffin was a feature (either in its own right, or as a named feature of the breeding seabird assemblage) was identified (Table 105). For each island the citation population was recorded, and the most recent population size was determined from the SMP database. The numerical change in population size and percentage change since designation was calculated and the current Site Condition Monitoring status recorded. The presence of feral cats, brown and black rats, and American mink were determined using the information in Stanbury et al. (2017).

Some of the islands identified by Stanbury et al. (2017) either did not have areas on them designated as SPAs, or the seabird assemblage did not include puffin as a named feature of the SPA breeding seabird assemblage. The absence of a SPA or puffin appearing as a named feature of the assemblage did not necessarily mean the absence of breeding puffins on that island. So, for those islands where there was not a SPA the SMP database was used to determine whether breeding puffins were present, what their population size was around the Seabird 2000 count, what the most recent population count was and whether this had changed (Table 106).

**Table 105 Top 25 islands prioritised for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) and the SPAs designated for their puffin populations, their citation population size, current population size, change in population size since designation, percentage change in population size since designation, current Site Condition Monitoring (SCM) status, and the presence of key invasive mammal predators. Percent change is highlighted as green if the population has grown since designation, amber if it has declined slightly and red if it has declined markedly. SCM status is highlighted as green if the population is Favourable and red if Unfavourable.**

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Foula | Foula SPA | 48,000 | 6,351 in 2016 | -41,649 | -87 | UNc | Y | N | N | N |
| Fair Isle | Fair Isle SPA | 23,000 | 6,666 in 2015 | -16,334 | -71 | UD | Y | N | N | N |
| Westray | West Westray | Not a qualifying feature | | | | | Y | N | N | N |
| Garbh Eilean and Eilean an Taighe, Shiants | Shiant Isles | 77,000 | 64,695 in 2015 | -12,305 | -16 | FM | N | N | Y | N |
| Rousay | Rousay | Not a qualifying feature | | | | | Y | Y | N | N |
| Rathlin Island | Rathlin Island | 2,398 | 695 in 2011 | -1,703 | -71 | | Y | Y | N | N |
| Colonsay and Oronsay | North Colonsay & Western Cliffs SPA | Not a qualifying feature | | | | | Y | Y | N | N |
| Unst | Hermaness, Saxa Vord & Vallafield SPA | 55,000 | 1,757 in 2017 | -53,243 | -97 | UD | Y | Y | N | N |
| Yell | None | | | | | | Y | N | N | N |
| Rum | None | | | | | | Y | N | N | N |
| Papa Westray | North Hill and Holm SPA | Not a qualifying feature | | | | | Y | N | N | N |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brow n rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Fetlar | Fetlar | Not a qualifying feature | | | | | Y | N | N | N |
| Inchkeith | None | | | | | | N | Y | N | N |
| Hoy | Hoy | 3,500 | 361 in 2017 | -3,139 | -90 | UD | Y | Y | N | N |
| Flotta | None | | | | | | Y | Y | N | N |
| Tiree | None | | | | | | Y | Y | N | N |
| Inchmarnock | None | | | | | | N | Y | N | Y |
| Stronsay | None | | | | | | Y | Y | N | N |
| Eilean Mhuire, Shiant Islands | As above | | | | | | N | N | Y | N |
| Gairsay | None | | | | | | Y | Y | N | N |
| North Ronaldsay | None | | | | | | Y | N | N | N |
| Muck | None | | | | | | N | Y | N | N |
| Housay, Out Skerries | None | | | | | | Y | Y | N | N |
| South Havra, Shetland | None | | | | | | Y | N | N | N |

| Island | SPA | Citation population | Current population | Change from designation | Percent change | SCM | Feral cat | Brow n rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|
| Herm, Channel Islands | None | | | | | | Y | Y | Y | N |

**Table 106 Islands prioritised for invasive alien vertebrate eradication in the UK based on the eradication benefit of feasible and sustainable (from Stanbury et al. 2017) where the site was either not designated for their puffin populations, or puffin was not a named feature in the assemblage of more than 20,000 breeding individuals.**

| Island | SPA | Seabird assemblage | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Westray | West Westray | Y | 877 | 1999 | 38 | 2017 | -839 | -95.7% | Y | N | N | N |
| Rousay | Rousay | Y | 53 | 1999 | 104 | 2016 | 51 | 96.2% | Y | Y | N | N |
| Colonsay and Oronsay | North Colonsay and Western Cliffs | Y | 1 | 2000 | 0 | 2019 | -1 | | Y | Y | N | N |
| Yell | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Rum | Rum | Y | 17 | 1999 | no count | no count | ? | ? | Y | N | N | N |
| Papa Westray | North Hill and Holm SPA | N | no count | no count | no count | no count | | | Y | N | N | N |

| Island | SPA | Seabird assemblage | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fetlar | Fetlar | Y | 1090 | 1999 - 2002 | no count | no count | ? | ? | Y | N | N | N |
| Inchkeith | None | N | 1373 | 2000 | 1600 | 2018 | 227 | 16.5% | N | Y | N | N |
| Flotta | None | N | no count | no count | 6 | 2019 | 6 | | Y | Y | N | N |
| Tiree | None | N | no count | no count | 0 | 2019 | | | Y | Y | N | N |
| Inchmarnock | None | N | no count | no count | no count | no count | | | N | Y | N | Y |
| Stronsay | None | N | no count | no count | 1 | 2018 | 1 | | Y | Y | N | N |
| Gairsay | None | N | no count | no count | no count | no count | | | Y | Y | N | N |
| North Ronaldsay | None | N | no count | no count | no count | no count | | | Y | N | N | N |
| Muck | None | N | no count | no count | no count | no count | | | N | Y | N | N |

| Island | SPA | Seabird assemblage | SMP | SMP (year) | SMP (most recent) | SMP (most recent year) | Change | Percent change | Feral cat | Brown rat | Black rat | American mink |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Housay, Out Skerries | None | N | no count | no count | no count | no count | | | Y | Y | N | N |
| South Havra, Shetland | None | N | 28 | 2000 | 10 | 2016 | -18 | -64.3% | Y | N | N | N |
| Herm, Channel Islands | None | N | no count | no count | no count | no count | | | Y | Y | Y | N |

There were fewer breeding puffin populations in SPAs on the islands identified and ranked by Stanbury et al. (2017) than for guillemot. Puffin populations had declined and were in unfavourable condition on Foula, Fair Isle, Unst and Hoy. There were also declines on Rathlin Island, but no Site Condition Monitoring information is available. The Shiants SPA had declined but was in favourable condition. Recently, rat eradication has been completed on all of the islands in the Shiants, so it is no longer available as a compensation location. No populations had increased. Among the islands where there were no SPAs, or puffin was not a named feature of the SPA, declines were noted on Westray and South Havra, Shetland. Increases were noted from Rousay and Inchkeith with very small populations noted from Colonsay and Oronsay, Flotta and Stronsay. While the declines in the colonies shown in Table 105 and Table 106 are likely to be for a variety of reasons, it is likely that the presence of rats and other invasive mammal predators are a contributing factor. There is strong evidence from the islands of Lundy, Canna, Ailsa Craig and Handa that eradication of brown rats results in population increases or recolonisation (see above). It is therefore likely that eradication of rats and other invasive mammal predators from these islands could contribute to the compensation for losses of breeding adult puffins predicted for the FFC SPA population.

Comparing the low, medium and high impact scenarios (Table 107) with the declines shown in Table 105 and Table 106 indicates that there is potential to compensate for even the high impact scenario through eradication of rats and other invasive mammal predators from a variety of islands, mostly in Scotland. Islands with a change in their population size since designation smaller than the impact scenario could have lower potential to compensate. This is summarised in Table 107, but note that rat eradication has already been completed on the Shiant Islands. It is important to note, that these islands would likely have been designated with the same rat and other invasive mammal predator population as they have now, so it is likely that population would grow if predation pressure was reduced or removed, including those islands where the probability may be lower (i.e. Amber shaded in Table 107).

**Table 107 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios (Green = high probability, Amber = lower probability, Red = no probability).**

| Islands suitable for compensation | Compensation | | |
|---|---|---|---|
| | Low | Medium | High |
| | 1.8 | 10.9 | 114.5 |
| Foula | | | |
| Fair Isle | | | |
| Garbh Eilean and Eilean an Taighe, Shiants* | | | |
| Rathlin Island | | | |
| Unst | | | |
| Hoy | | | |
| Westray | | | |
| Rousay | | | |
| Colonsay and Oronsay | | | |
| Inchkeith | | | |

| Islands suitable for compensation | Compensation | | |
|---|---|---|---|
| | Low | Medium | High |
| | 1.8 | 10.9 | 114.5 |
| Flotta | Red | Red | Red |
| Stronsay | Red | Red | Red |
| South Havra, Shetland | Green | Amber | Red |

* Invasive terrestrial mammals have already been eradicated from the Shiants.

### 6.7.1.4 *Compensation ratios*

Increasing the compensation ratios to 1:3 and 1:6 resulted, unsurprisingly, in fewer potentially suitable islands being available for compensation (Table 108). The only island suitable for High impacts at either 1:3 or 1:6 ratios was Westray. At the medium impact level six islands were probably or possibly suitable and at the low impact level there were eight possible islands.

**Table 108 Islands with rats and invasive mammal predators compared with low, medium, and high compensation scenarios at 1:3 and 1:6 ratios (Green = high probability, Amber = lower probability, Red = no probability).**

| Islands suitable for compensation | Change | 1:3 | | | 1:6 | | |
|---|---|---|---|---|---|---|---|
| | | Low | Medium | High | Low | Medium | High |
| | | 5.4 | 32.7 | 343.5 | 10.8 | 65.4 | 687 |
| Foula | -87 | Green | Green | Red | Green | Green | Red |
| Fair Isle | -71 | Green | Green | Red | Green | Green | Red |
| Garbh Eilean and Eilean an Taighe, Shiants* | -16 | Red | Red | Red | Red | Red | Red |
| Rathlin Island | -71 | Green | Green | Red | Green | Amber | Red |
| Unst | -97 | Green | Green | Red | Green | Green | Red |
| Hoy | -90 | Green | Green | Red | Green | Green | Red |
| Westray | -839 | Green | Green | Green | Green | Green | Green |
| Rousay | 51 | Red | Red | Red | Red | Red | Red |
| Colonsay and Oronsay | -1 | Red | Red | Red | Red | Red | Red |
| Inchkeith | 645 | Red | Red | Red | Red | Red | Red |
| Flotta | 6 | Red | Red | Red | Red | Red | Red |
| Stronsay | 1 | Red | Red | Red | Red | Red | Red |
| South Havra, Shetland | -18 | Green | Red | Red | Amber | Red | Red |

* Invasive terrestrial mammals have already been eradicated from the Shiants.

### 6.7.2 EAA 2

#### 6.7.2.1 *Closure of sandeel or sprat fisheries*

Population level assessment was not completed for any other SPAs in the EAA 2 area. Ten SPAs designated for their breeding puffin populations also occurred within EAA 2.

All of these colonies would be expected to benefit from a closure of UK water to sandeel fisheries, as all of these colonies occur within foraging range of sandeel stocks that are currently or previously depleted through fishing. The sandeel box off the east coast of Scotland has likely positively affected the puffin populations in SPAs on the east coast of Scotland, which is only the Forth Islands SPA. It is important to note that the sandeel box is a fisheries management tool, not a seabird conservation tool, and that current management of the stock that includes the sandeel box allows a take of the stock that could deplete the stock despite the presence of the box. This stock has not experienced the same level of take since the box was put in place, though fishing occurred within the stock in 2021.

#### 6.7.2.2 *Eradication of rats and other invasive mammal predators*

The majority of islands identified in Table 108 occurred in EAA 2 (Table 109). Since eradication of mammals from FFC SPA is not a suitable compensation measure, the next level of preferred location would be the islands occurring in EAA 2 shown in Table 109.

**Table 109 EAA for islands identified as potentially suitable for compensation for puffin through eradication of rats and other invasive mammal predators.**

| Islands suitable for compensation | EAA |
|---|---|
| Foula | 2 |
| Fair Isle | 2 |
| Garbh Eilean and Eilean an Taighe, Shiants* | 4 |
| Rathlin Island | 4 |
| Unst | 2 |
| Hoy | 2 |
| Westray | 2 |
| Rousay | 2 |
| Colonsay and Oronsay | 4 |
| Inchkeith | 2 |
| Flotta | 2 |
| Stronsay | 2 |
| South Havra, Shetland | 2 |

### 6.7.3 All other EAAs in the UK

#### 6.7.3.1 *Closure of sandeel or sprat fisheries*

The closure of UK waters to sandeel fisheries would be expected to mainly benefit colonies in areas that are currently, or have previously been, subject to sandeel fisheries. There are 10 SPAs designated for their puffin populations in all other EAAs in the UK, but much of the areas of sea within the foraging range of birds in these SPAs has not experienced the same take of sandeels from stocks as those in the North Sea. It is therefore considered unlikely that closure of sandeel fisheries would have the same level of benefit to puffin populations in all other EAAs in the UK.

#### 6.7.3.2 *Eradication of rats and other invasive mammal predators*

The islands with potential for compensation outside EAA 2 are shown in Table 109. These islands would be in the final tier of location hierarchy, with sites in EAA 2 preferred. Only three islands were identified and one of these (Shiant Islands) has already had invasive mammals eradicated.

## 6.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the two recommended compensation measures was estimated for the closure of sandeel or sprat fisheries compensation approach and the eradication of rats and other invasive mammals approach. The summary table for the closure of sandeel or sprat fisheries compensation method is shown in Table 110, and the PVA method in Table 111. The confidence in the eradication of rats and other invasive mammals compensation method is shown in Table 112 and the confidence in the assessment method is shown in Table 113. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 114 and Table 115 (closure of sandeel or sprat fisheries) and in Table 116 and Table 117 (eradication of rats and other invasive mammals).

**Table 110 Assessment of confidence in the recommended compensation method of closure of sandeel or sprat fisheries.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Puffins feed chicks mostly on sandeels | n/a | Tasker & Furness 1996 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Sandeel collapse reduced breeding success | n/a | Miles et al. 2015 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Puffins cannot maintain breeding success when sandeel abundance is low | n/a | Shetland Bird Reports and Fair Isle Bird Observatory Reports | Medium | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Supplementary feeding increases chick growth rate | n/a | Fitzsimmons et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | LOW | MEDIUM |
| Sandeel stocks improved by limiting fishing effort | n/a | Lindegren et al. (2018) | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Recovery slowed by predation pressure | n/a | Saraux et al. 2020 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Winter diet varies between years | n/a | Glew et al. (2019) | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Importance of healthy stocks of forage fish for puffin populations | n/a | Fayet et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Puffins forage on sandeels in winter | n/a | Harris et al. 2005b | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 111 Assessment of confidence in the inputs to PVA assessing closure of sandeel or sprat fisheries approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 5 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Upper constraint on productivity | 1 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 980 pairs in 2015 | SACO | Medium | Low | Medium | Robust | MEDIUM | HIGH | MEDIUM |
| Productivity rate per pair | mean: 0.7522, sd: 0.1289 | PVA app "National" default value | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Adult survival rate | mean: 0.907, sd: 0.108 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 0 to 1 | mean: 0.8283, sd: 0.077 | Calculated from survival to breeding age of 0.39 (from Harris 1983), s.d. from PVA app "National" default value. | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.8283, sd: 0.077 | Calculated from survival to breeding age of 0.39 (from Harris 1983), s.d. from PVA app "National" default value. | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Age class 2 to 3 | mean: 0.8283, sd: 0.077 | Calculated from survival to breeding age of 0.39 (from Harris 1983), s.d. from PVA app "National" default value. | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.8283, sd: 0.077 | Calculated from survival to breeding age of 0.39 (from Harris 1983), s.d. from PVA app "National" default value. | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 112 Assessment of confidence in the eradication of rats and other invasive mammals method to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Rat eradication from offshore islands benefits puffin populations | n/a | Booker et al. 2019, Luxmoore et al. 2019, Zonfrillo 2002, 2007 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Presence of rats on islands with puffins | n/a | Stanbury et al. 2017 | Robust | Robust | Robust | Medium | MEDIUM | HIGH | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 113 Assessment of confidence in the eradication of rats and other invasive mammals approach to compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Citation population size | n/a | SiteLink | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |
| Current population size | n/a | SMP database | Robust | n/a | n/a | Robust | ROBUST | HIGH | VERY HIGH |
| Site Condition | n/a | SiteLink | Robust | n/a | n/a | Medium | MEDIUM | HIGH | MEDIUM |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 114 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Puffins feed chicks mostly on sandeels | Tasker & Furness (1996) reviewed evidence from Runde, Fair Isle, Isle of May and Farne Islands. This showed the high importance of sandeels in puffin chick diet across several years, albeit during 1970s to 1980s. This was robust data with a medium applicability, as it was from the same region as FFC SPA, but no data was from FFC. Confidence was therefore high. |
| Sandeel collapse reduced breeding success | Miles et al. 2015 was a very robust study across multiple years on Fair Isle. Since the location was not at FFC SPA, but still in the North Sea applicability was scored as medium. Confidence was therefore high. |
| Puffins cannot maintain breeding success when sandeel abundance is low | This evidence comes from bird reports and bird observatory reports. These are not peer reviewed scientific publications, so the quality of evidence was assessed as medium. However, these reports contain robust evidence collected using agreed, standardised methods. An evidence score of robust was therefore supported. Since this evidence was from Shetland the applicability score was medium, resulting in high confidence. |
| Supplementary feeding increases chick growth rate | A robust study was undertaken by Fitzsimmons et al. 2017. However, the study was undertaken in eastern Canada, so the applicability was assessed as low, resulting in medium confidence. |
| Sandeel stocks improved by limiting fishing effort | With evidence from the modelling by Lindegren et al. 2018 showing stock recovery is possible with reduced fishing mortality, and a general concept that one third of the available stock should be available for natural predators from Cury et al. 2011 the evidence was assessed as robust. While the evidence from Lindegren et al. 2018 applies directly to the stock needed for puffins at FFC SPA, the evidence from Cury et al. 2011 was not from the FFC SPA puffin population directly, so the applicability was assessed as medium. This resulted in a recommended confidence score of high. |
| Recovery slowed by predation pressure | Saraux et al (2020) studies the responses of depleted prey populations to predation pressure. This was a very robust study. The study used data from a very wide variety of locations around the world, including Shetland. Thus, the applicability score was medium, so the overall confidence was high. |
| Winter diet varies between years | The study by Glew et al (2019) provided a robust study, which was from the Isle of May, so assessed as medium applicability. Thus, the confidence score was high. |
| Importance of healthy stocks of forage fish for puffin populations | Fayet et al (2017) undertook a very robust study of puffins from 12 colonies across the North Atlantic. The colonies included the Isle of May, so the applicability was scored as medium. The overall confidence was high. |
| Puffins forage on sandeels in winter | Harris et al (2005b) was a robust meta-analysis of puffin data from multiple colonies. These colonies included three colonies in the North Sea. So, this robust study had a medium applicability giving a high confidence. |
| **OVERALL CONFIDENCE SCORE** | All but one of the confidence scores were high, with the other a medium. There were multiple sources of very robust data, including several meta-analyses. Thus, an overall confidence of high had merit. |

**Table 115 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of puffins is well established and not in question. It is not variable between populations and is directly applicable to the FFC SPA population. |
| Upper constraint on productivity | Puffins lay a single egg and so productivity cannot be above this. |
| Initial population size | Counts of breeding puffins are well known to be challenging, especially in the type of habitat used by puffins at FFC SPA. The count used was based on a standardised and approved method but confidence in its accuracy was low. However, based on the subsequent counts of individual puffins on the sea it is likely that this was a reasonable starting population to use as about 2,000 individuals, or more, have been counted in recent years. |
| Productivity rate per pair | The productivity rate was based on the default "National" value in the Seabird PVA Tool. This was considered a reasonable alternative to site based values. These data are based on robust data in peer reviewed publications that have been selected by CEH as suitable to parameterise PVA models generally. Their applicability can only be medium as they were not site specific data. |
| Adult survival rate | The adult survival rate evidence is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. A medium score was therefore given. With robust evidence and medium applicability, the confidence score was high. |
| Age specific survival rates from 0 to 4 years | These survival rates were based on the survival from fledging to breeding age from a robust published source. They were from another North Sea razorbill colony, but one relatively close to the FFC SPA and so their applicability was considered medium. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was high or medium, so an overall score of high was given. |

**Table 116 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation method from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Rat eradication from offshore islands benefits puffin populations | Multiple studies on puffin colonies around the UK have provided robust evidence that eradication of rats from offshore islands both benefits existing populations of puffins and can result in recolonisation by puffins to islands previously used for nesting. These islands occurred in similar areas to those proposed for rat eradication here, so the applicability was scored as high. Therefore, an overall confidence in the compensation method was high. |
| Presence of rats on offshore islands | Stanbury et al. (2017) was only used to identify potential islands for compensation. The quality, consistency and amount of evidence |

| Metric | Narrative |
|---|---|
| | were all scored as robust, but the type of evidence was scored as medium as the study only determined the presence/absence of predators and not their absolute or relative abundance. This was considered sufficiently important to result in an overall evidence score of medium. Applicability was high as this evidence was for the islands that could be used for compensation. The overall confidence score was therefore high. |
| **OVERALL CONFIDENCE SCORE** | Both confidence scores were high. There were multiple sources of robust data that the eradication of rats from offshore islands was successful where it has been applied. The evidence for islands with rats was weaker. As islands get larger with larger human populations confidence in the ability to create and maintain rat free islands declines. **The suitable islands are relatively large and inhabited, so the confidence in rat eradication on these islands is better considered as medium.** |

**Table 117 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from eradication of rats and other invasive mammals.**

| Metric | Narrative |
|---|---|
| Citation population size | The citation population size was obtained from the Citation document from NatureScot SiteLink. This is a legal document, so the value is not in question. Evidence is robust and applicability is high. So, confidence was very high. |
| Current population size | The current population size was the most recent whole SPA count available from the SMP database. The most recent count was used for each site, so these may be different years between different SPAs. However, these are the best available evidence and the count method and source were considered robust. Applicability was high. So, confidence was very high. |
| Site Condition | Site Condition of each puffin SPA population was obtained from the NatureScot SiteLink website. Site Condition Monitoring is a count based methodology, but the assignment of condition scores is not based on site or species specific methodologies, so the type of evidence was scored as medium. Applicability was high. An overall score of medium was considered appropriate as the site condition monitoring methodology is not designed for tracking populations change, but for broad scale assessment of the UK SPA network. |
| **OVERALL CONFIDENCE SCORE** | The confidence scores were very high (two) or medium (one). The applicability scores were high. Thus, an overall confidence score of high was considered appropriate. |

With an overall assessment of high in the compensation method and the assessment method for the closure of the sandeel and sprat fishery, the assessment of confidence in the proposed compensation methods against the three impact scenarios needs to be carefully considered. The population level assessment (6.7.1.1) showed that the low and medium impact scenarios could be compensated for by the low, medium and high compensation scenarios that increased productivity, so confidence in these was high with the exception of the medium impact low compensation scenario, which was assessed as medium confidence as the CPS and CGR values were less than one (Table 118). The high impact scenario could not be compensated for, even with the high compensation scenarios, so confidence in these was low (Table 118).

**Table 118 Assessment of confidence in the impact/compensation scenarios for puffins from sandeel or sprat fisheries closure increasing productivity.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | MEDIUM | LOW |
| Medium compensation | HIGH | HIGH | LOW |
| High compensation | HIGH | HIGH | LOW |

As with the compensation affecting productivity there needs to be careful consideration of the confidence that the compensation measures would be sufficient to overcome the impact scenarios if closure of the sandeel and sprat fisheries increased adult survival. The population level assessment (6.7.1.1) also showed that the low and medium impact scenarios could be compensated for by the low, medium and high compensation scenarios that increased adult survival, so confidence in these was high (Table 119). The high impact scenario could be compensated for by the high compensation scenario, so confidence in this was high. The high impact medium compensation scenario was projected by the PVA to have a median population growth rate greater than one and a CGR value only a little below one, so this was assessed as having medium confidence. There was low confidence that the high impact scenario could be compensated by low compensation measures.

**Table 119 Assessment of confidence in the impact/compensation scenarios for puffins from sandeel or sprat fisheries closure increasing adult survival.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | HIGH | LOW |
| Medium compensation | HIGH | HIGH | MEDIUM |
| High compensation | HIGH | HIGH | HIGH |

The rat and other invasive mammal eradication measures also had a high confidence in the approach and the assessment process. The confidence in the ability to compensate for the three impact scenarios varied between islands (Table 112 & Table 113). The assessment provided in Table 107 showed that there was high potential for the low and medium impacts to be compensated for through rat eradication on six islands, but only potential for rat eradication being successful in compensating for the high impact scenario on one island (Westray). However, as islands get larger, and their human populations are larger confidence in the ability to create and maintain rat free islands must decline. All of the islands considered suitable are relatively large and inhabited, so the confidence in rat eradication on these islands is better considered as medium.

## 6.9 Future monitoring and adaptive management

### 6.9.1 Closure of sandeel or sprat fisheries

If compensation through closure of the UK sandeel fishery was applied, it would be important that suitable monitoring is put in place to demonstrate that this has been effective at:

- Increasing the sandeel stock available to the puffin population at FFC SPA; and
- Increasing the population size at FFC SPA.

Methods for suitable monitoring of sandeel stocks would need to be established with experts in this field and is beyond the scope of this study. However, fishery-independent assessment methods have been developed by Norway (using acoustic sampling) and by Marine Scotland (using grab samples) as described in the latest sandeel benchmark stock assessment (ICES 2017). Monitoring would need to determine overall abundance of the stock and perhaps also the stock within the foraging range of puffins from the FFC SPA to ensure that the compensation measure is having the desired effect on the prey resource for the population at FFC SPA.

In addition to assessing the stock at both the relevant ICES stock area and within the foraging range of the FFC SPA puffin population, monitoring of the change in the population at the colony would be important.

These monitoring measures need to be connected to adaptive management decision making. The proposed monitoring needs to be considered together when adapting the management to the results of the monitoring. The aim of the proposed compensation is to increase the puffin population at the FFC SPA colony. Monitoring of sandeel stocks is needed to determine whether recovery of the stock was as expected, below the level expected or above the level expected. Similarly adaptive management will need to consider whether action is necessary if the change in population size is above or below the expected value. Ultimately the need to adapt management actions will need to be based on whether the population size at the SPA changes as a result of the proposed compensation method. Given the difficulties of estimating the actual breeding population at FFC SPA, continuing the monitoring of change in puffins on the water at the colony in the early morning at the start of the breeding season would be useful in assessing whether compensation has resulted in positive population change, or at least stability even with increasing predicted impacts. It may be necessary to move to other compensation mechanisms should the closure of the sandeel fishery ultimately prove to be unsuccessful.

#### 6.9.2 Eradication of rats and other invasive mammal predators

After a suitable location for rat eradication is selected it will be important to collect data on the current population size of the puffin population on that island. At least one breeding season count using the SMP methodology should be completed to provide an up-to-date baseline condition for future comparison. It may also be helpful to use the method of monitoring at FFC SPA (i.e. early morning counts early in the breeding season of adult puffins on the sea near the colony) where puffin burrows are difficult to access. In addition, evidence should be collected from the island, preferably around the colony, to demonstrate that rats or other invasive mammals are present. This would likely need to be carefully designed for the specific island and colony being considered for eradication. Suitable methods for collecting these data would be chew sticks, camera traps, live traps and kill traps. Following application of an eradication method suitable for the invasive mammals present (there may need to be more than one) it will be necessary to continue monitoring to ensure eradications method have been successful. Should either the targeted species, or other invasive mammal species, be detected further eradication measures would likely be necessary. Monitoring for the presence/absence of mammals should continue throughout the period of compensation. Details on the intensity and frequency of this monitoring would need to be site specific. Several important characteristics will need to be considered when determining this frequency and intensity. These include distance from nearest other mammal population (more remote island will need less monitoring), type and frequency of boat visits (islands only visited occasionally by small passenger only vessels are less exposed to re-introduction of invasive mammalian predators than more frequent visits by larger cargo vessels), presence and type of working agriculture on the island (islands with working farms are more likely to accidentally introduce invasive mammal predators than those with none), and presence and size of the human population living on the island (reintroduction risk of invasive mammals increases with the presence of human populations on the island).

Monitoring of the breeding adult population would be necessary to demonstrate that the colony increases in size following the removal of invasive mammal predators. Initially annual counts of the colony size are advised, with this frequency being reduced as recovery continues. The level of reduction in monitoring frequency should be related to the level of recovery observed. Monitoring should continue throughout the period compensation is required, at least periodically. If invasive mammal predators appear to have re-established then monitoring frequency should increase following additional eradication, particularly if the population was suppressed by the re-establishment.

#### 6.9.3 Future research

One of the key elements of future research would be whether the predicted improvements to adult survival and productivity from sandeel fisheries closure could be combined in a robust manner. It seems likely that both effects would act simultaneously on populations, which would likely greatly increase the response of populations to sandeel fisheries closure. However, there was insufficient empirical evidence to support combining the increases shown here.

Future research on the effects of rats on puffin populations, particularly in relation to eradication programmes, would be to understand the non-lethal effects on puffins on offshore islands. It is apparent that in some locations rat removal has allowed more nest sites to become available for new recruits, but it is unknown whether this is the primary driver of population increase or whether it is decline in adult or nest mortality.

### 6.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for puffins at FFC SPA: closure of sandeel and sprat fisheries in all UK waters, rat eradication, and prevention of oil spills. Oil spills have become very rare in the UK through both strong legislation and strong application of best practice, so it was concluded that this would no longer be a suitable compensation measure. Since Furness et al. (2013) was published some new evidence has been published on the benefits of fisheries closure and invasive mammal eradication.

PVA suggested that the FFC SPA population should be increasing and that this increase would remain likely with low and medium impact levels. However, the high impact scenario resulted in a rapid projected decline in the FFC SPA population. Three levels of compensation measure were then assessed against the three levels of impact. The increase of productivity and increase in adult survival of puffins that might follow from sandeel fisheries closure were both tested in the PVA model. The PVA results testing an increase in productivity suggested that the low and medium level impacts could be compensated by low, medium, and high levels of compensation, though confidence in the low compensation being sufficient for the medium impact was less certain. None of the compensation scenarios were sufficient to overcome the high impact scenario. The results for the testing of compensation scenarios that increased adult survival also found that low, medium, and high compensation scenarios could all compensate for low and medium level impacts, though again with lower confidence in the medium impact low compensation scenario, but not for high level impacts.

Assessment of compensation through eradication of invasive mammal predators from offshore islands was able to identify several islands where the recorded decline in the population size of puffins was larger than the impact scenarios being considered. Most of these island colonies were able to compensate for low, medium, and high levels of impact.

Confidence in the PVA assessment process was high. However, confidence was low in the ability of fisheries closures to compensate for high levels of impact, whether this compensation was through increasing productivity or adult survival. However, confidence was high for compensating the low and medium impact scenarios for both productivity and adult survival compensation scenarios. There was high or medium confidence in the ability of rat eradication from offshore islands to compensate for low, medium, or high impact scenarios across multiple islands, including those ranked most highly by Stanbury et al. (2017).

# 7 NORTH NORFOLK COAST SPA – BREEDING SANDWICH TERN

North Norfolk Coast SPA (NNC SPA) is located east of The Wash on the northern coastline of Norfolk and covers an area of nearly 8,000 hectares extending approximately 40 km from Holme to Weybourne. A variety of coastal habitats occur within the SPA, including intertidal mudflats and sandflats, coastal waters, saltmarshes, shingle, sand dunes, freshwater grazing marshes and reedbeds. The site is important within Europe as one of the largest areas of undeveloped coastal habitat of its type, and at designation was the fourth most important wetland site for waterfowl in Britain. The coastal waters along the North Norfolk Coast SPA are shallow. They support large populations of small fish, including sandeel and sprat, which provide food for breeding tern populations in the SPA. Breeding Sandwich terns are one of the many features of the SPA.

## 7.1 Conservation status of Sandwich tern

The biogeographic population (defined as the subspecies *S. s. sandvicensis*) was estimated at 74,000 pairs, of which 11,000 pairs breed in Great Britain and 3,700 pairs in all-Ireland (Mitchell et al. 2004). Sandwich tern breeding numbers in the UK increased from the 1920s to the mid-1980s, after major reductions caused by human exploitation and hunting (JNCC 2020). National surveys showed an increase in the UK population of 33% from 1969 to 1986, but a decrease of 15% from 1986 to 2000 (JNCC 2020). JNCC SMP data show no clear long-term trend for UK breeding numbers between 1986 and 2018, with the index in 2018 almost the same as in 1986 (JNCC 2020).

Stroud et al. (2016) identified that the SPA suite with breeding Sandwich tern as a designated feature has 13 qualifying sites in Great Britain, three in Scotland (Forth Islands SPA; Loch of Strathbeg SPA; Ythan Estuary, Sands of Forvie and Meikle Loch SPA), nine in England (Alde-Ore Estuary SPA; Chichester and Langstone Harbours SPA; Coquet Island SPA; Duddon Estuary SPA; Farne Islands SPA; Foulness SPA; Morecambe Bay SPA; North Norfolk Coast SPA; Solent and Southampton Water SPA) and one in Wales (Ynys Feurig, Cemlyn Bay and The Skerries SPA, now known as Anglesey terns SPA). The SPAs in Great Britain were estimated to hold 72% of the Great Britain breeding population of Sandwich terns present in 2000 (Stroud et al. 2016). Three sites in Northern Island also qualify (Carlingford Lough; Larne Lough; and Strangford Lough). North Norfolk Coast SPA held 3,700 pairs of Sandwich terns at designation, the largest breeding population of the species in the UK SPA suite. Numbers have decreased at many of the SPA sites, but have increased at some, including North Norfolk Coast SPA, such that the overall change since designation is small. Similarly, the JNCC seabird monitoring index for Sandwich tern suggests that current numbers in England (in 2020) are very similar to numbers present in 1986; the index in 2020 being essentially the same as in 1986 despite periods in the mid-1990s and early 2010s when the index fell below 100 (JNCC, 2020).

Within the boundary of the North Norfolk Coast SPA, Sandwich terns breed at two principal colonies; Blakeney Point and Scolt Head (JNCC, 2020; Perrow et al., 2017). Alternative breeding locations within the SPA, such as Stiffkey/Holkham, have been unused since 2004 (JNCC, 2020). JNCC Seabird Monitoring Programme database presents breeding numbers and breeding success for the two main colonies (Table 120).

**Table 120 Numbers of pairs of Sandwich terns (or AONs) and breeding success (chicks per pair) for colonies in NNC SPA. Data from JNCC SMP database except for Scolt Head 2019 and 2020 which have not yet been entered into the database (as on 16/02/2021). Counts for that site in 2019 and 2020 are from local ornithologists' unpublished counts but will most likely be added to the database.**

| Year | Scolt Head | | Blakeney Point | | Stiffkey/Holkham | Total in NNC SPA |
|---|---|---|---|---|---|---|
| | pairs | Chicks/pr | pairs | Chicks/pr | pairs | pairs |
| 1969 | 3850 | | 0 | | 96 | 3946 |
| 1970 | 4022 | | 35 | | 0 | 4057 |
| 1971 | 4400 | | 0 | | 0 | 4400 |
| 1972 | 4800 | | 1 | | 350 | 5151 |
| 1973 | 800 | | 0 | | 3500 | 4300 |
| 1974 | 1610 | | 0 | | 1400 | 3010 |
| 1975 | 3200 | | 0 | | 0 | 3200 |
| 1976 | 4000 | | 47 | | 0 | 4047 |
| 1977 | 3000 | | 1750 | | 91 | 4841 |
| 1978 | 2100 | | 3000 | | 0 | 5100 |
| 1979 | 2000 | | 3650 | | 0 | 5650 |
| 1980 | 1200 | | 2400 | | 0 | 3600 |
| 1981 | 300 | | 3850 | | 0 | 4150 |
| 1982 | 250 | | 3200 | | 0 | 3450 |
| 1983 | 500 | | 3300 | | 0 | 3800 |
| 1984 | 1700 | | 2500 | | 0 | 4200 |
| 1985 | 2500 | | 1000 | | 0 | 3500 |
| 1986 | 2600 | 0.86 | 1000 | | 0 | 3600 |
| 1987 | 3089 | 1.03 | 475 | | 0 | 3564 |
| 1988 | 2775 | 0.43 | 1000 | | 0 | 3775 |
| 1989 | 1052 | 0 | 1500 | | 0 | 2552 |
| 1990 | 0 | - | 3000 | 1.00 | 0 | 3000 |
| 1991 | 320 | 0 | 3000 | 0.27 | 0 | 3320 |
| 1992 | 280 | 0 | 4000 | 1.00 | 0 | 4280 |
| 1993 | 853 | 1.17 | 3000 | 0.87 | 0 | 3853 |
| 1994 | 2406 | 1.03 | 1000 | 0.95 | 0 | 3406 |
| 1995 | 1588 | 0.82 | 1450 | 1.00 | 0 | 3038 |
| 1996 | 450 | No data | 3500 | 0.86 | 0 | 3950 |
| 1997 | 220 | 0 | 3000 | 0.50 | 0 | 3220 |
| 1998 | 650 | 0 | 3000 | 0.58 | 0 | 3650 |
| 1999 | 1000 | 0.65 | 3200 | 0.63 | 0 | 4200 |

| Year | Scolt Head | | Blakeney Point | | Stiffkey/Holkham | Total in NNC SPA |
|---|---|---|---|---|---|---|
| | pairs | Chicks/pr | pairs | Chicks/pr | pairs | pairs |
| 2000 | 4200 | 1.19 | 75 | 0 | 0 | 4275 |
| 2001 | 4000 | 1.00 | 250 | 1.01 | 221 | 4471 |
| 2002 | 3050 | 0.90 | 750 | 0.73 | 800 | 4600 |
| 2003 | 650 | 0.77 | 2900 | 0.79 | 250 | 3800 |
| 2004 | 1800 | 0.34 | 1260 | 0.24 | 18 | 3078 |
| 2005 | 1900 | 0.87 | 1650 | 0.55 | 0 | 3550 |
| 2006 | 2500 | 0.80 | 950 | 0.86 | 0 | 3450 |
| 2007 | 1800 | 0 | 1800 | 0.78 | 0 | 3600 |
| 2008 | 280 | 0.01 | 2400 | 0.64 | 0 | 2680 |
| 2009 | 0 | - | 3100 | 0.42 | 0 | 3100 |
| 2010 | 480 | 0 | 2500 | 0.36 | 0 | 2980 |
| 2011 | 0 | - | 3562 | 0.52 | 0 | 3562 |
| 2012 | 400 | 0 | 3735 | 0.59 | 0 | 4135 |
| 2013 | 550 | 0 | 4120 | 0.44 | 0 | 4670 |
| 2014 | 1050 | 0.60 | 2859 | 0.19 | 0 | 3909 |
| 2015 | 3550 | 0.90 | 1113 | 0.01 | 0 | 4663 |
| 2016 | 3365 | 0.80 | 451 | 0.39 | 0 | 3816 |
| 2017 | 4665 | 0.94 | 3 | 0 | 0 | 4668 |
| 2018 | 4685 | 0.85 | 165 | 0.12 | 0 | 4850 |
| 2019 | 3805 | No data | 788 | 0.51 | 0 | 4593 |
| 2020 | 4160 | 0.72 | 2425 | 0.45 | 0 | 6585 |

The data in Table 120 indicate that there has been no significant long-term trend in breeding numbers in the SPA (1969-2020, Figure 45, $r=0.063$, 50 d.f., n.s.) although numbers have fluctuated, and were below average through 2003-2011 (Figure 45), and some autocorrelation is evident. However, there has been a significant short-term recent increase in numbers between 2008 and 2020 (Figure 46, $r=0.858$, 11 d.f., $p<0.01$), though in this case autocorrelation is much more evident. The (as yet unconfirmed) number in 2020 appears to be particularly high, although follows the recent short-term trend of increasing numbers.

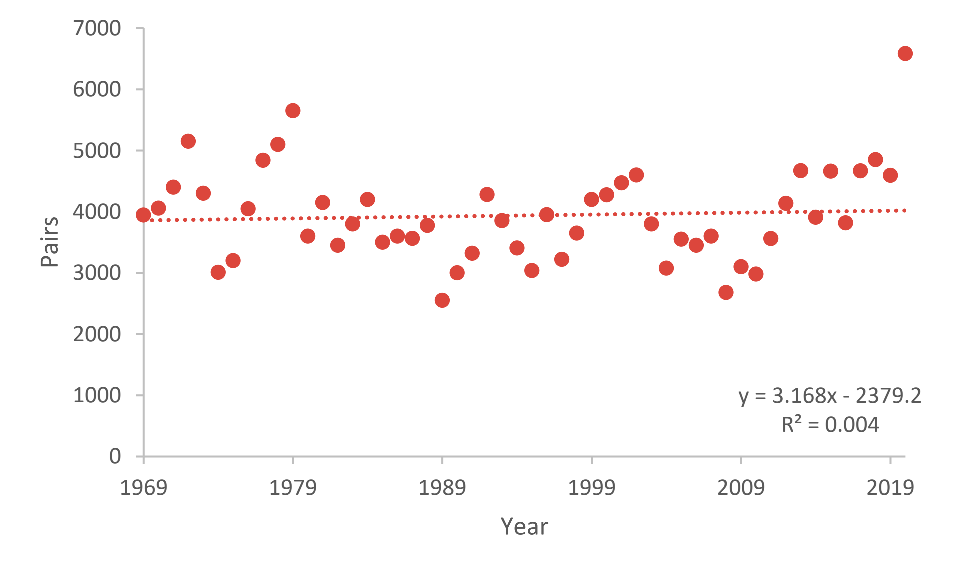



Figure 45 Long-term trend (1969-2020) in breeding numbers of Sandwich terns in NNC SPA. Data from Table 120.

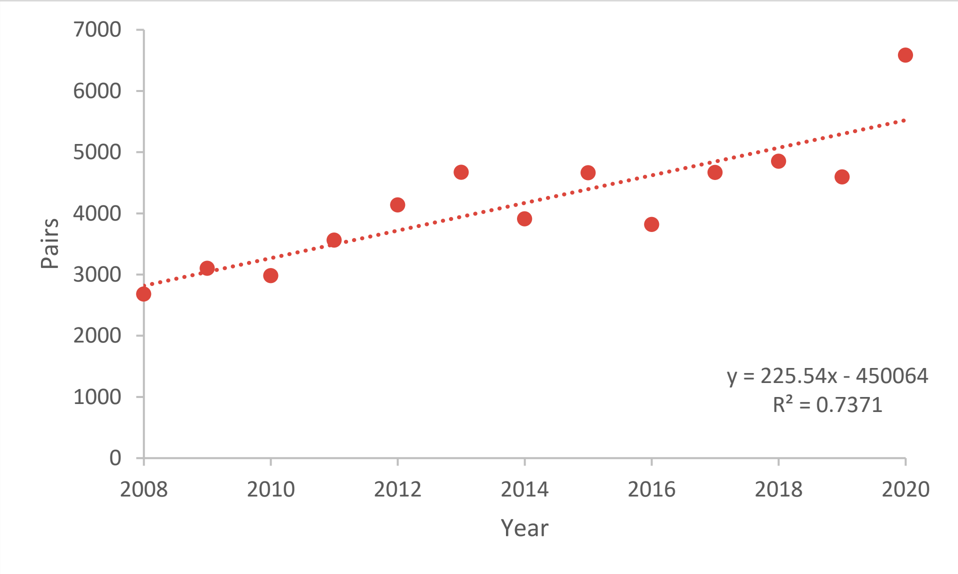



Figure 46 Short-term trend (2008-2020) in breeding numbers of Sandwich terns in NNC SPA. Data from Table 120.

Sandwich terns nest in a relatively small number of large and dense, highly synchronous, colonies on large areas of bare ground, in areas with extensive sheltered shallow waters nearby. While breeding they feed predominantly on small pelagic fish, in the British Isles on sandeels, sprats and young herring. Their fishing success is severely hampered by strong winds and rough seas (Dunn 1973, Taylor 1983, Stienen et al. 2000), so they tend to feed mainly in sheltered bays and estuaries (Mitchell et al. 2004), although they may commute to shallow banks further out to sea where those hold stocks of forage fish. After breeding, they migrate to spend the winter off West Africa.

Woodward et al. (2019) list the foraging range of breeding Sandwich terns as mean 9 km, mean maximum 34.3 km, maximum 80 km. However, these distances are likely to apply more along the coast than directly out to sea, given the preference of UK breeding Sandwich terns to remain near the coast.

Sandwich terns are highly vulnerable to mammal predators and declines at colonies are most often related to an increase in predator access, especially to foxes, but also rats, stoats and American mink. Predators can cause complete abandonment of a colony, or periodic breeding failure (Mitchell et al. 2004). Predation by gulls can also influence breeding success but tends to be less of a problem than predation by mammals. Sandwich tern nesting habitat is dynamic, with influences of coastal erosion and flooding potentially leading to habitat loss, and of plant succession potentially leading to habitat becoming overgrown and unsuitable for this species (Mitchell et al. 2004). Sandwich terns have been affected by chemical pollution, with very large decreases in breeding numbers in the Netherlands in the 1960s (Mitchell et al. 2004) but that pressure has been reduced. Breeding success can be strongly affected by forage fish abundance and breeding failures have been related to reductions in stocks of sandeel, sprat and juvenile herring. Overwinter survival may be influenced by fisheries off West Africa affecting abundance of forage fish in that region (Mitchell et al. 2004), and deliberate trapping of birds at the West African coast for sport and food has been identified as affecting survival, especially of immature birds.

### 7.2 Citation population size

The SPA citation (dated 30 January 1996) states that the site qualifies under Article 4.1 of the Birds Directive by supporting up to 4,500 pairs of Sandwich terns (no reference to the source of this count is provided). However, the Natura 2000 Standard Data Form, dated 25 January 2016, lists the qualifying breeding Sandwich tern as a 5-year mean (1992-1996) of 3,700 pairs. JNCC Seabird Monitoring Programme database lists counts in NNC SPA as 4,280 pairs in 1992, 3,853 pairs in 1993, 3,406 pairs in 1994, 3,038 pairs in 1995, and 3,950 pairs in 1996. The mean of these counts is 3,705 pairs, which could reasonably be rounded to 3,700 as in the Natura 2000 form.

### 7.3 Conservation objectives

The site's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site.

More detailed conservation objectives have since been added online, last updated 13 September 2019 (Natural England 2020). For Sandwich tern at NNC SPA these are:

- Restore the size of the breeding population to a level which is above 4,500 pairs, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Maintain safe passage of birds moving between nesting and feeding areas;
- Reduce the frequency, duration and/or intensity of disturbance affecting roosting, nesting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;
- Restrict predation and disturbance caused by native and non-native predators;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System (www.apis.ac.uk);
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable habitat (either within or outside the site boundary) which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding) at levels described in site specific supporting notes;
- Maintain the distribution, abundance and availability of key food and prey items (eg. sandeel, sprat) at preferred sizes. The availability of an abundant food supply is critically important for successful breeding, adult fitness and survival and the overall sustainability of the population;
- Maintain the availability of shallow sloping nesting sites, grading to <30 cm above water level, restricting the probability that they will flood;
- Maintain vegetation cover which should be <10% throughout areas used for nesting, providing sufficient bare ground for the colony as a whole;
- Restrict aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;
- Maintain water quality at mean winter dissolved inorganic nitrogen levels where biological indicators of eutrophication (opportunistic macroalgal and phytoplankton blooms) do not affect the integrity of the site and features, avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

The aim to restore Sandwich tern breeding numbers to above 4,500 pairs may merit re-appraisal, given that the numbers at designation averaged 3,705 pairs (1992-1996) and did not exceed 4,500 pairs in any of the years on which the designation was based. A running mean for breeding numbers of Sandwich terns at NNC SPA also indicates that a population size exceeding 4,500 pairs has not been sustained since counts began in 1969, although numbers occasionally exceeded 4,500 pairs in some individual years (Table 120).

A Site Improvement Plan was published in December 2014, outlining the prioritized issues for the site and features, and the proposed measures to address those issues.

### 7.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely; and
- the populations of each of the qualifying features.

There are three main sources of impact on Sandwich terns from offshore wind farm development: mortality due to collisions with operational turbines, displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain the structure and function of the habitat and supporting processes of the qualifying features could be affected through the displacement of Sandwich terns from the wind farm, if birds from the SPA used this area for foraging prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of birds will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival. That impact on survival may be a carry-over effect on reduced winter survival as birds are in poorer condition at the end of the breeding season than would have been the case in the absence of the wind farm.

The maintenance of the population of each of the qualifying features could be affected directly through collision mortality and indirectly through impact to energy budgets from displacement and barrier effects.

## 7.4 Location of compensation

Cook et al. (2011) defined five Ecological Assessment Areas for Sandwich tern (Figure 34). The NNC SPA occurs within EAA 1. Consequently, the hierarchy of the locations of compensation are:

1. NNC SPA;
2. EAA 1; and
3. All other EAAs in the UK.

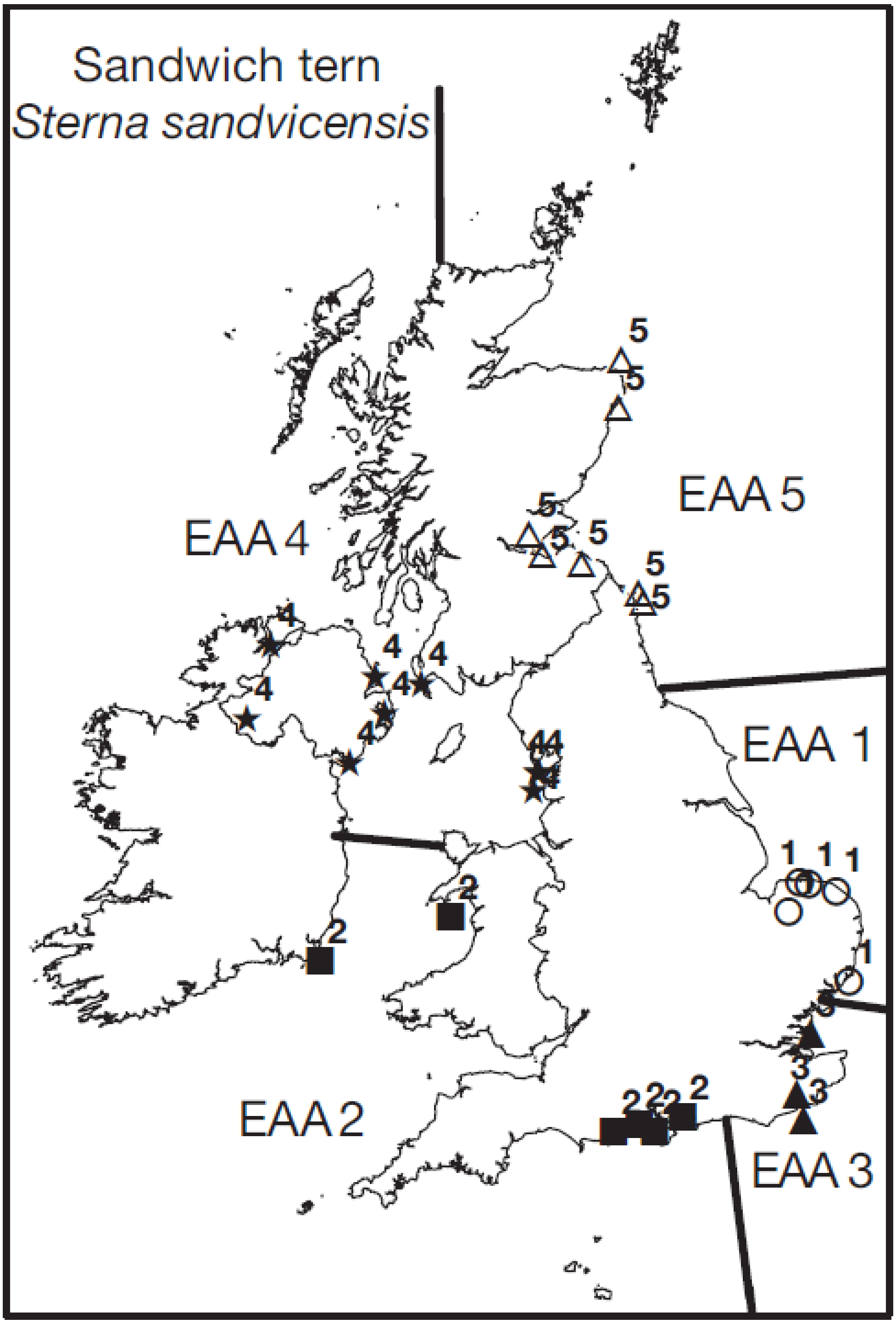



**Figure 47 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for Sandwich tern by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

## 7.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding Sandwich terns were developed based on the eight potential compensation measures reviewed by Furness et al. (2013). The eight potential measures listed were:

1. Fencing out foxes from colonies;
2. Stoat control/eradication;
3. Flood and vegetation control at colonies;
4. Closure of sandeel and sprat fisheries close to Sandwich tern colonies;
5. Exclusion of large gulls;
6. Mink eradication;
7. Feral cat eradication; and
8. Rat eradication.

The first three of these potential measures were considered highly likely to be effective with high confidence in that assessment based on evidence. While there was strong evidence that closure of sandeel and sprat fisheries would benefit related seabird species, there was weaker evidence in this regard specifically for Sandwich tern. There was a lack of clear evidence that this species would benefit from measures 5 to 8, except possibly in a few particular locations.

Recent research emphasises the specialist diet of Sandwich tern, and the importance of high densities of small pelagic fish near to colonies if this species is to breed successfully. In Belgium and The Netherlands this species feeds almost exclusively on just three prey types, small Clupeids (herring and sprat), sandeels, and Nereis worms (Courtens et al. 2017). Foraging effort and breeding success are strongly influenced by food availability, with adult body condition at colonies where forage fish are scarce being reduced by high breeding effort, suggesting that shortage of forage fish probably affects adult survival as well as colony breeding success (Stienen et al. 2015, Fijn et al. 2017). Food shortage is also implicated as a cause of reduced productivity at several of the main UK colonies (Furness et al. 2013). Frederiksen and Wanless (2006) concluded that "Sandwich terns may have been affected by reduced sandeel availability during the 1990s in a similar way to black-legged kittiwakes". These results strengthen the evidence that measures to increase abundance of sandeels and sprats in waters near to Sandwich tern colonies can be expected to result in an increase in breeding success and probably an increase in adult survival of Sandwich terns.

Thus, all of the first four measures appear to be potential approaches to use to compensate for impacts to Sandwich terns at NNC SPA.

The key biological questions for compensation measures for Sandwich terns at NNC SPA are provided in Table 121.

**Table 121 Key Biological Questions in assessing the potential for compensation of breeding Sandwich terns at NNC SPA.**

| No. | Key Biological question |
|---|---|
| **Fencing out foxes** | |
| 1 | Is there evidence of foxes causing breeding failure or site desertion in: |

| No. | Key Biological question |
|---|---|
| 1.1 | NNC SPA? |
| 1.2 | EAA 1? |
| 1.3 | All other EAAs in the UK? |
| 2 | Are predator proof fences effective at excluding predators from seabird colonies? |
| 3 | Is there evidence of successful measures to prevent fox predation on Sandwich tern colonies in: |
| 3.1 | NNC SPA? |
| 3.2 | EAA 1? |
| 3.3 | All other EAAs in the UK? |
| **Stoat control/eradication** | |
| 1 | Is there evidence of stoats causing breeding failure or site desertion in: |
| 1.1 | NNC SPA? |
| 1.2 | EAA 1? |
| 1.3 | All other EAAs in the UK? |
| **Flood and vegetation control** | |
| 1 | Is there evidence that flooding, or vegetation development can cause breeding failure or site desertion in: |
| 1.1 | NNC SPA? |
| 1.2 | EAA 1? |
| 1.3 | All other EAAs in the UK? |
| **Closure of sandeel and sprat fisheries** | |
| 1 | Is there evidence for sandeels being important in the diet of Sandwich terns in: |
| 1.1 | NNC SPA? |
| 1.2 | EAA 1? |
| 1.3 | All other EAAs in the UK? |
| 2 | Is there evidence for sandeel abundance affecting Sandwich tern demographics |
| 3 | Is there evidence of Sandwich tern populations being limited by productivity in: |
| 3.1 | NNC SPA? |
| 3.2 | EAA 1? |
| 3.3 | All other EAAs in the UK? |

## 7.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 7.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

### 7.6.1 Fencing out foxes

There has been evidence of fox activity at NNC SPA Sandwich tern colonies causing breeding failure and site desertion in a few years. Fencing of colonies to exclude foxes would allow Sandwich tern productivity to increase at colonies where this predator is present. In the UK, some examples of using electric fences to exclude foxes from colonies have been successful, but electric fences are not fully effective in excluding predators and require frequent maintenance. A more effective alternative is the use of predator-proof fences, as deployed in Hawai'i at Ka'ena Point Natural Area Reserve (Young et al. 2012). These two metre tall fences prevent predators (including rats and mice) from entering the protected area. Predators (in their case dogs, cats, mongoose, rats and mice) were eradicated within the enclosed 20 ha (which took three months to complete for all predators except mice which were eradicated within an additional six months). This was the first predator proof fence constructed in the United States at the time of its completion (Young et al. 2012) but the same approach has been used extensively in New Zealand, and has been used at a few sites in Europe, including the Azores where it has been deployed to exclude predators from ground-nesting seabird colonies (Furness et al. 2013, RSPB 2020, Xcluder 2020). Such completely predator-proof fencing may be unnecessary to protect colonies just from foxes but might be especially appropriate for colonies subject to predation by rats, stoats or mink as well as by foxes.

#### *7.6.1.1 Answers to the key biological questions (7.5).*

The answers to the key biological question in relation to compensation through fencing out foxes are shown in Table 122.

**Table 122 Answers to Key Biological Questions in assessing the potential for compensation through fencing out foxes.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence of foxes causing breeding failure or site desertion in: | |
| 1.1 | NNC SPA? | Yes, there is evidence of fox predation at NNC SPA (Walsh et al. 1991) |
| 1.2 | EAA 1? | Yes. Since 1995 there have been very few nesting attempts in the Alde-Ore Estuary (AOE) SPA which Ratcliffe et al. (2000) attributed to fox predation. |
| 1.3 | All other EAAs in the UK? | Yes. There is evidence of fox predation causing breeding failure at multiple colonies in all other EAAs in the UK. See Section 7.7.3. |
| 2 | Are predator proof fences effective at excluding predators from seabird colonies? | Yes. There is good evidence from Young et al. (2012), Furness et al. (2013), RSPB (2020), and Xcluder (2020) that suitable predator proof fencing is effective at excluding predators, such as foxes. |
| 3 | Is there evidence of successful measures to prevent fox predation on Sandwich tern colonies in: | |
| 3.1 | NNC SPA? | No. No evidence could be found through the literature search of fox management measures being applied in NNC SPA. |
| 3.2 | EAA 1? | No. No evidence could be found through the literature search of fox management measures being applied in AOE SPA. |
| 3.3 | All other EAAs in the UK? | Yes. There is good evidence for fox management measures positively affecting Sandwich tern colonies in all other EAA's in the UK. See Section 7.7.3.1. |

### 7.6.2 Stoat control/eradication

It is unclear whether stoat predation is a problem at NNC SPA Sandwich tern colonies, so this measure would not be appropriate at this SPA unless evidence on the extent of predation by stoats could be gathered (e.g. by remote camera technology).

#### *7.6.2.1 Answers to the key biological questions (7.5).*

The answers to the key biological question in relation to compensation through stoat control/eradication are shown in Table 123.

**Table 123 Answers to Key Biological Questions in assessing the potential for compensation through stoat control/eradication.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence of stoats causing breeding failure or site desertion in: | |
| 1.1 | NNC SPA? | No. There was no evidence found of stoat predation specifically at NNC SPA. |
| 1.2 | EAA 1? | No. There was no evidence found of stoat predation specifically at SPA colonies in EAA 1. |
| 1.3 | All other EAAs in the UK? | Yes. Furness et al. (2013) presented evidence of stoat predation at Sandwich tern colonies at the Ythan Estuary, Sands of Forvie and Meikle Loch SPA and Morecambe Bay SPA and Duddon Estuary SPA. In addition, stoat (and crow) control has been applied at the Ynys Feurig, Cemlyn Bay and The Skerries SPA. |

### 7.6.3 Flood and vegetation control

There is evidence that flooding/erosion, and vegetation development at NNC SPA Sandwich tern colonies can cause breeding failure and site desertion. NNC SPA Site Improvement Plan states "Investigate the options for adaptive site management in light of ecological changes likely to occur due to increased frequency and duration of saline inundation". It is well known that Sandwich terns prefer to nest on areas of flat bare coastal habitat, and that these sites can be at risk of flooding and erosion by tidal inundation and by intense rainfall and runoff. Sandwich terns may also abandon nesting areas if too much vegetation develops on the nesting area. Long-term breeding success of Sandwich terns at NNC SPA could potentially be improved by engineering works that maintain Sandwich tern preferred nesting habitat in optimal condition, engineered to minimize risk of flooding and erosion and to minimize risk of excessive vegetation development on the nesting area.

#### *7.6.3.1 Answers to the key biological questions (7.5).*

The answers to the key biological question in relation to compensation through flood and vegetation control are shown in Table 124.

**Table 124 Answers to Key Biological Questions in assessing the potential for compensation through flood and vegetation control.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that flooding or vegetation development can cause breeding failure or site desertion in: | |
| 1.1 | NNC SPA? | Yes. There have multiple instances of colonies in NNC SPA being affected by flooding, including complete colony breeding failure (e.g. Thompson et al. 1997). |
| 1.2 | EAA 1? | Yes. The Alde Ore Estuary SPA site improvement plan states, "Flood wall breaches in December 2013 (due to tidal surge) has led to flooding". However, Sandwich tern was not mentioned as a specific issue in relation to flooding, presumably as there is not a breeding colony at present. |
| 1.3 | All other EAAs in the UK? | Yes. Vegetation control has been raised as an important issue in the management plan for the Farnes Islands SPA, where rank nitrophilous vegetation has replaced previously short cropped vegetation and much of the former nesting sites for terns, including Sandwich terns, is no longer suitable. |

## 7.6.4 Closure of sandeel and sprat fisheries.

Whereas measures that allow forage fish stocks to recover are likely to benefit Sandwich terns, the relatively short foraging range of breeding Sandwich terns (a mean range of only about 9 km), suggests that measures throughout the UK North Sea will have relatively little local relevance to NNC SPA, as sandeel fishing occurs further offshore than the foraging range of Sandwich tern. It is also unclear whether NNC SPA Sandwich terns feed mainly on sprats or on sandeels, or a combination of both forage fish species. Local sprat stocks may be more relevant for this population, in which case constraints on sprat fishing may be more beneficial than constraints on sandeel fishing. More evidence on diet of NNC SPA Sandwich terns may be required but could be obtained quite easily by cameras set up to record birds returning to nest sites carrying fish.

### *7.6.4.1 Answers to the key biological questions (7.5).*

The answers to the key biological question in relation to compensation through closure of sandeel and sprat fisheries are shown in Table 125.

**Table 125 Answers to Key Biological Questions in assessing the potential for compensation through closure of sandeel and sprat fisheries.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence for sandeels being important in the diet of Sandwich terns in: | |
| 1.1 | NNC SPA? | Yes. Sandeels are known to form an important part of the diet of Sandwich terns in North Norfolk (Perrow et al. 2010). |
| 1.2 | EAA 1? | Unknown. No evidence could be found of the diet of Sandwich terns at Alde-Ore Estuary SPA in EAA 1. |
| 1.3 | All other EAAs in the UK? | Yes. Frederiksen and Wanless (2006) reported that the closure of the sandeel fishery off the east coast of Scotland resulted in the productivity of Sandwich tern colonies in the region being 0.24 chicks per pair higher than before the closure. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2 | Is there evidence for sandeel abundance affecting Sandwich tern demographics | Yes. Frederiksen and Wanless (2006) reported that the closure of the sandeel fishery off the east coast of Scotland resulted in the productivity of Sandwich tern colonies in the region being 0.24 chicks per pair higher than before the closure. |
| 3 | Is there evidence of Sandwich tern populations being limited by productivity in: | |
| 3.1 | NNC SPA? | Yes. Breeding success of Sandwich tern has been monitored every year at Scolt Head since 1986 and at Blakeney Point since 1990. The productivity at both colony locations in the SPA have been very variable ( Figure 48). From 2011 to 2020 the mean productivity at Scolt Head was 0.60 chicks per pair (between 0 and 0.94) and at Blakeney it was 0.32 chicks per pair (between 0 and 0.59). |
| 3.2 | EAA 1? | No. There is no direct evidence from AOE SPA, the only other SPA colony in EAA 1, that the colony was limited by sandeels prior to its extirpation. |
| 3.3 | All other EAAs in the UK? | Yes. Frederiksen and Wanless (2006) showed that Sandwich tern colonies had higher productivity when sandeels were more abundant. |

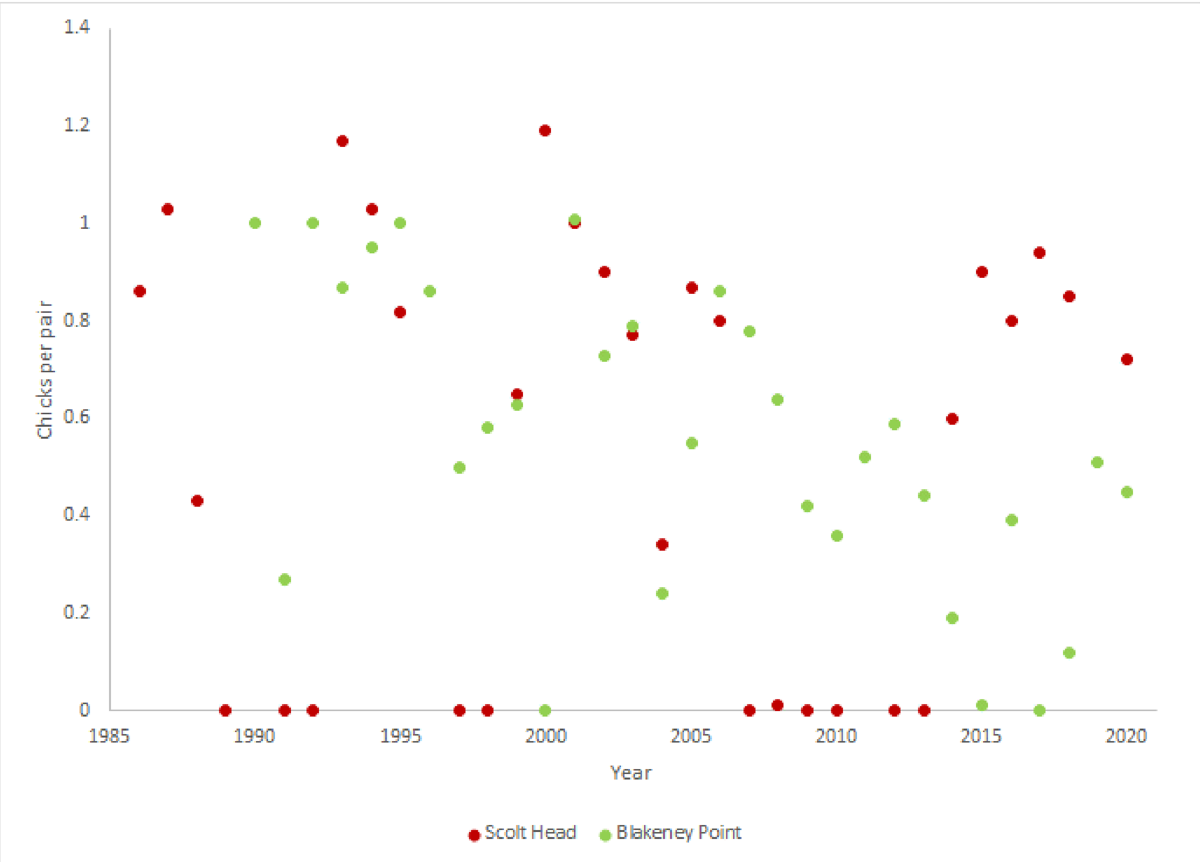



**Figure 48 Breeding success of Sandwich terns at NNC SPA from 1986 to 2020 (from SMP database).**

## 7.7 Population level assessment

### 7.7.1 North Norfolk Coast SPA

All the population level assessments for NNC SPA Sandwich terns were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult mortality rate. For a population size of 3,457 pairs (from the DECC strategic AA in 2012) and an adult survival rate of 0.898 a 1% increase in baseline mortality would be 7 additional birds being killed per annum.

The medium impact scenario was based on a pro-rata increase in the installed capacity of proposed Round 4 offshore wind farms (an additional 7GW) compared to the current installed capacity (26GW). This resulted in an estimated mortality of 25 birds per annum.

The high impact scenario was based on the current in-combination impact on the population from all offshore wind farms (7 birds killed per annum) pro-rated to the 2050 net zero target of 100GW of installed capacity. This is an additional 74GW of additional capacity compared to the current level of installed, consented or planned capacity (26GW). This results in an additional mortality of 268 adult birds per annum, or 37.9% increase in adult mortality rate. Impact levels are summarised in Table 126.

**Table 126 Values for low, medium and high impact scenarios for Sandwich terns at NNC SPA.**

| Impact scenario | Low | Medium | High |
|---|---|---|---|
| Additional mortality (birds) | 7 | 25 | 268 |
| Additional mortality (rate) | 1% | 3.5% | 37.9% |

**Table 127 PVA input parameters for predator control compensation scenarios.**

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 3 | | | PVA app default |
| upper constraint on productivity | 3 chicks per pair | | | PVA app default |
| Initial population size | 3,563 in 2012 | | | DECC strategic AA |
| Productivity rate per pair | mean: 0.4461, sd: 0.3281 | | | Overall mean chicks per pair |

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| | | | | (2011 - 2020) at Scot Head & Blakeney Point |
| Adult survival rate | Mean: 0.898, sd: 0.116 | | | PVA app "National" default value |
| Age class 0 to 1 | mean: 0.77, sd: 0.02 | | | PVA app "National" default value |
| Age class 1 to 2 | mean: 0.77, sd: 0.02 | | | PVA app "National" default value |
| Age class 2 to 3 | mean: 0.77, sd: 0.02 | | | PVA app "National" default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Impact on adult survival rate | Low | Medium | High | Calculated as above |
| | 0.000982594, se: NA | 0.003509264, se: NA | 0.03754913, se: NA | |

*7.7.1.1 Predator control*

It was assumed that full predator control would be provided and would be effective. While most of the predation on the Blakeney Point and Scolt Head colonies is likely from foxes, other predators, particularly stoats and rats, may be present. Based on these assumptions three levels of compensation were derived. The "low" compensation scenario assumed that only in the years that adult birds were present and attempting to nest, and breeding failure of the whole colony occurred, would the anti-predator measures increase the productivity. It was assumed that productivity would be increased from zero by the mean of the previous ten years of productivity at both colonies combined where this was above 0.3 chicks per pair. This was 0.6425 chicks per pair (± 1 SD 0.1925). Where there were no nesting pairs present, the productivity at that colony was kept at zero, to replicate the stochastic characteristic of Sandwich tern colony movement. This resulted in the overall mean productivity across both colonies being 0.5883 chicks per pair an increase of 0.1422 chicks per pair (Table 128).

The "medium" compensation scenario was assumed to increase any productivity in the colony that was below the ten year mean of both colonies (i.e. 0.6425) to that level (i.e. any year with productivity below 0.6425 was increased to 0.6425, but values greater than this were not changed). This assumes that low productivity years were caused by predators and that these could be increased with sufficient anti-predator measures, but that years with higher productivity than the mean still occurred. This resulted in an overall mean across both colonies of 0.6979 (± 0.1010) chicks per pair, an increase of 0.2518 chicks per pair (Table 128).

The "high" compensation scenario was assumed to increase productivity to the mean of the top 50% of productivity values over the last ten years (2011 – 2020; 0.8017 ± 0.1253) in years where it had been less than this. Years when productivity had been higher than 0.8017 these values were unchanged. This resulted in an overall mean productivity of 0.8175 (± 0.0396) chicks per pair, an increase of 0.3714 chick per pair (Table 128).

**Table 128 Calculations for the change in productivity of Sandwich terns at NNC SPA because of predator control compensation. Green shaded cells are where productivity was assumed to increase as a result of predator control.**

| Year | Low compensation | | Medium compensation | | High compensation | |
|---|---|---|---|---|---|---|
| | Scolt Head | Blakeney | Scolt Head | Blakeney | Scolt Head | Blakeney |
| 2011 | No nests | 0.52 | No nests | 0.6425 | No nests | 0.8017 |
| 2012 | 0.6425 | 0.59 | 0.6425 | 0.6425 | 0.8017 | 0.8017 |
| 2013 | 0.6425 | 0.44 | 0.6425 | 0.6425 | 0.8017 | 0.8017 |
| 2014 | 0.6 | 0.19 | 0.6425 | 0.6425 | 0.8017 | 0.8017 |
| 2015 | 0.9 | 0.6425 | 0.9 | 0.6425 | 0.9000 | 0.8017 |
| 2016 | 0.8 | 0.39 | 0.8 | 0.6425 | 0.8017 | 0.8017 |
| 2017 | 0.94 | No nests | 0.94 | No nests | 0.9400 | No nests |
| 2018 | 0.85 | 0.12 | 0.85 | 0.6425 | 0.8500 | 0.8017 |
| 2019 | 0.6425 | 0.51 | 0.6425 | 0.6425 | 0.8017 | 0.8017 |
| 2020 | 0.72 | 0.45 | 0.72 | 0.6425 | 0.8017 | 0.8017 |
| | Mean | 0.5883 | Mean | 0.6979 | Mean | 0.8175 |
| | SD | 0.2213 | SD | 0.1010 | SD | 0.0396 |

**PVA results**

The baseline population projection was compared with the three impact scenarios (Table 126). The projections all showed a decline in the population size with time for baseline (unimpacted) and all three impact scenarios (Figure 49), with a much larger decline for the high impact scenario than the other two impact scenarios.

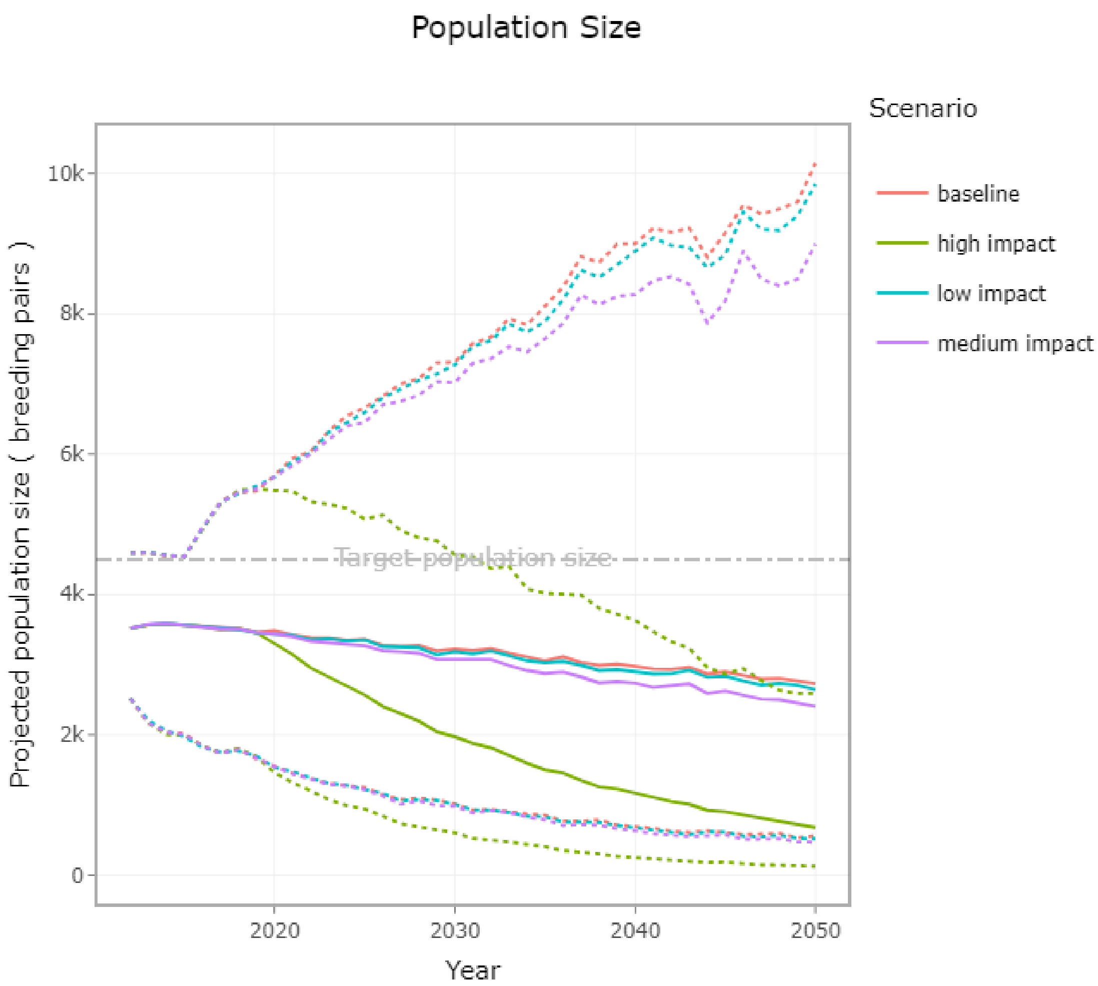


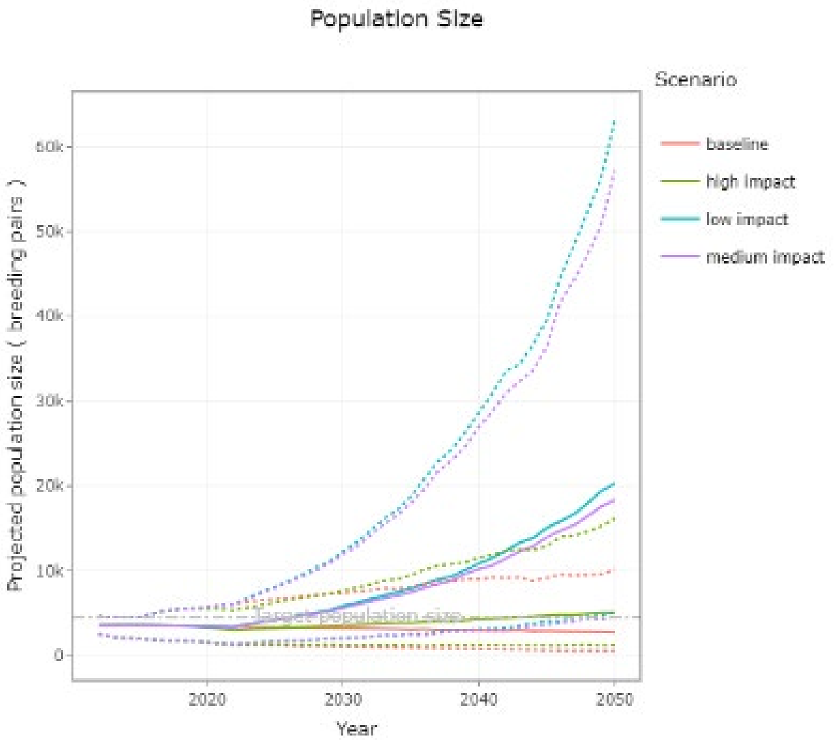

**Figure 49 Population projections for the baseline and low, medium and high impact scenarios.**

In none of the projected scenarios, including the baseline, was the target population size exceeded. The counterfactuals of both population size and growth rate from the projected population in 2050 showed relatively high levels of impact on the NNC SPA Sandwich tern population (Table 129).

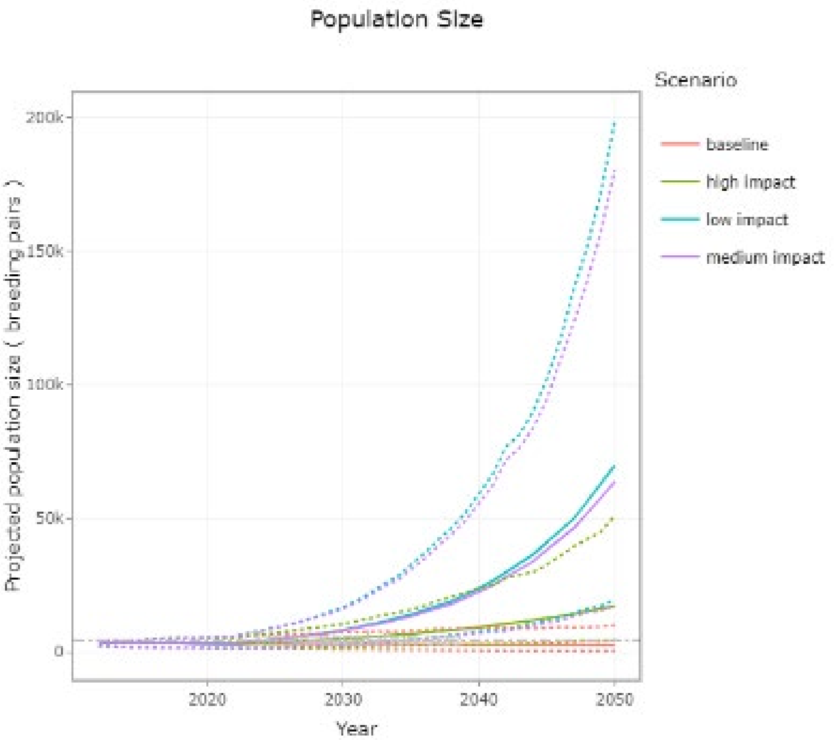
**Table 129 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9669 | 0.9659 (0.8766 – 1.0559) | 0.9989 | 0.9988 (0.9960 – 1.0013) |
| Medium | 0.8806 | 0.8827 (0.8040 – 0.9709) | 0.9959 | 0.9959 (0.9930 – 0.9988) |
| High | 0.2485 | 0.2474 (0.2072 – 0.2789) | 0.9561 | 0.9558 (0.9505 – 0.9594) |

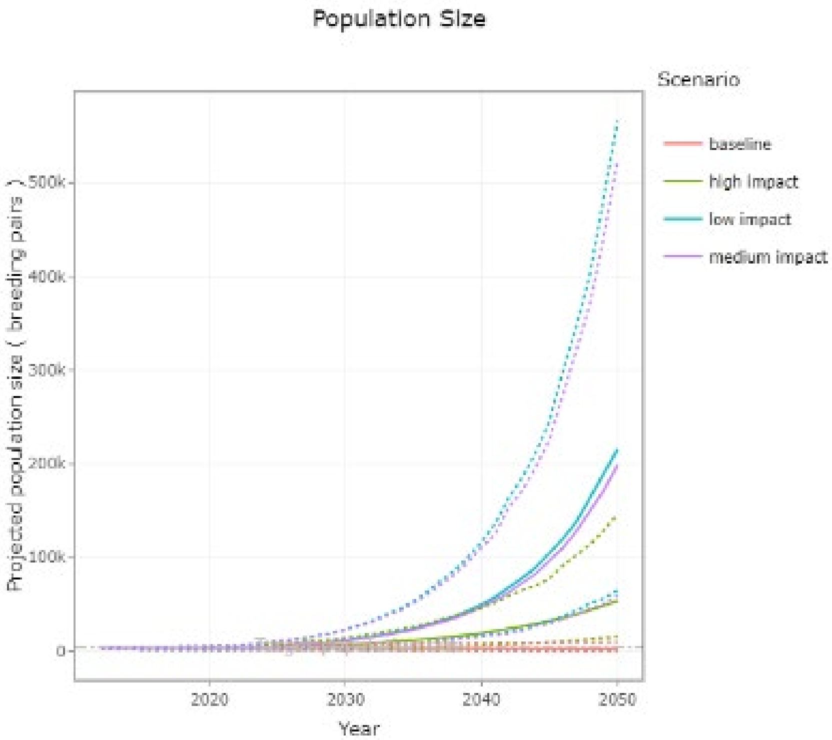
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on productivity (i.e. additional 0.1422 (low), 0.2518 (medium) or 0.3714 (high) chicks per pair) (Figure 50).




Low compensation




Medium compensation



High compensation

**Figure 50 Projected population size of breeding Sandwich tern (pairs) at NNC SPA comparing baseline with the low, medium, and high impact scenarios combined with low, medium and high compensation scenarios.**

In all scenarios the additional productivity resulted in the projected population size increasing (Table 130). The population size increases shown are all likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, all three scenarios suggested that if predator management is successful in delivering the simulated increase in productivity considered here, this measure is likely to result in substantial increases in the Sandwich tern population at NNC SPA.

**Table 130 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 0.9937 |
| Low | Low | 1.0595 |
| Low | Medium | 1.1022 |
| Low | High | 1.1431 |
| Medium | Low | 1.0565 |
| Medium | Medium | 1.0992 |
| Medium | High | 1.1397 |
| High | Low | 1.0136 |

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| High | Medium | 1.0541 |
| High | High | 1.0928 |

In all impact and compensation scenarios the target population size was exceeded within the time span of the population projection (30 years). The PVA showed that the population was more sensitive to the scale of change in compensation than the scale of the impacts modelled.

**Table 131 Year in which the projected median population size exceeded the target population size for each combination of impact and compensation scenario.**

| Impact | Compensation | Year target population size exceeded |
|---|---|---|
| Low | Low | 2027 |
| Low | Medium | 2025 |
| Low | High | 2024 |
| Medium | Low | 2027 |
| Medium | Medium | 2025 |
| Medium | High | 2024 |
| High | Low | 2043 |
| High | Medium | 2028 |
| High | High | 2026 |

Examination of the counterfactuals of population size showed a much stronger response to the level of compensation for the low and medium impact scenarios than the high impact scenarios (Figure 51). However, the low compensation scenarios all had CPS values greater than one, suggesting even this level of compensation was beneficial. The absolute scale of the CPS values, particularly for the high compensation scenarios, was largely due to the assumption of no density dependence in the population model. This is unrealistic, as the breeding population in the NNC SPA will be limited by available habitat. As such the CGR plot (Figure 52) is likely to be more informative.

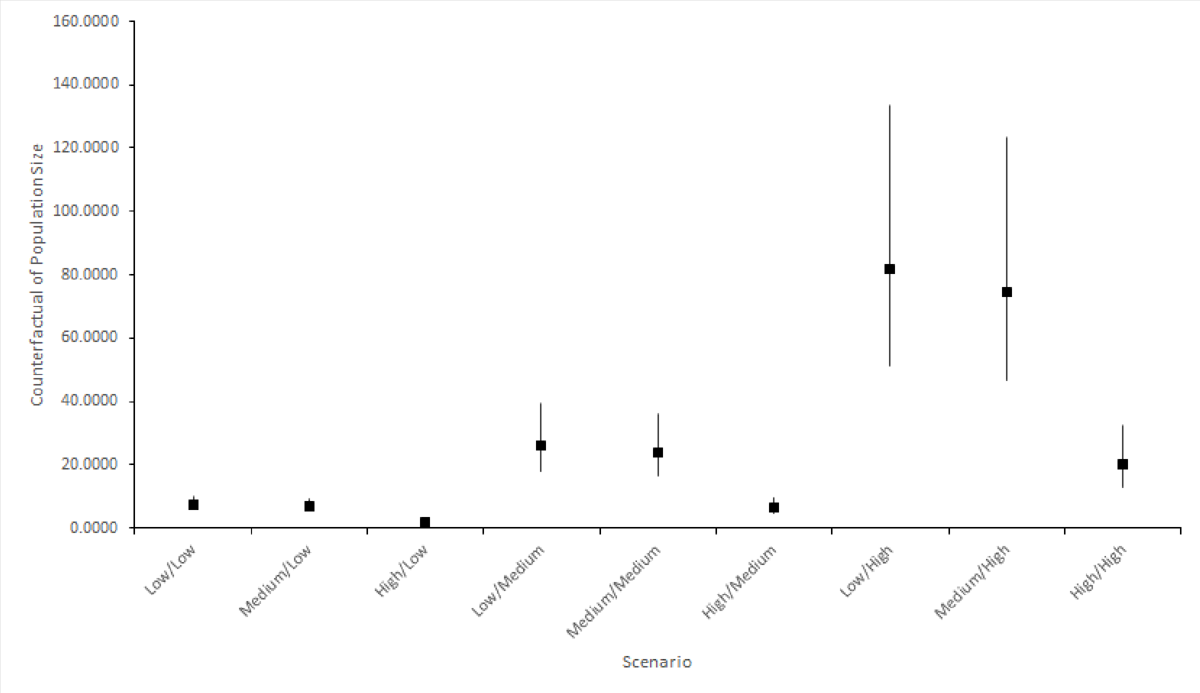



**Figure 51 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed a stronger response to the level of compensation from the low and medium impact scenarios than the high impact scenario (Figure 52). However, in all scenarios the CGR was above one, suggesting a positive effect on the population, even with a low level of compensation and high impacts being applied.

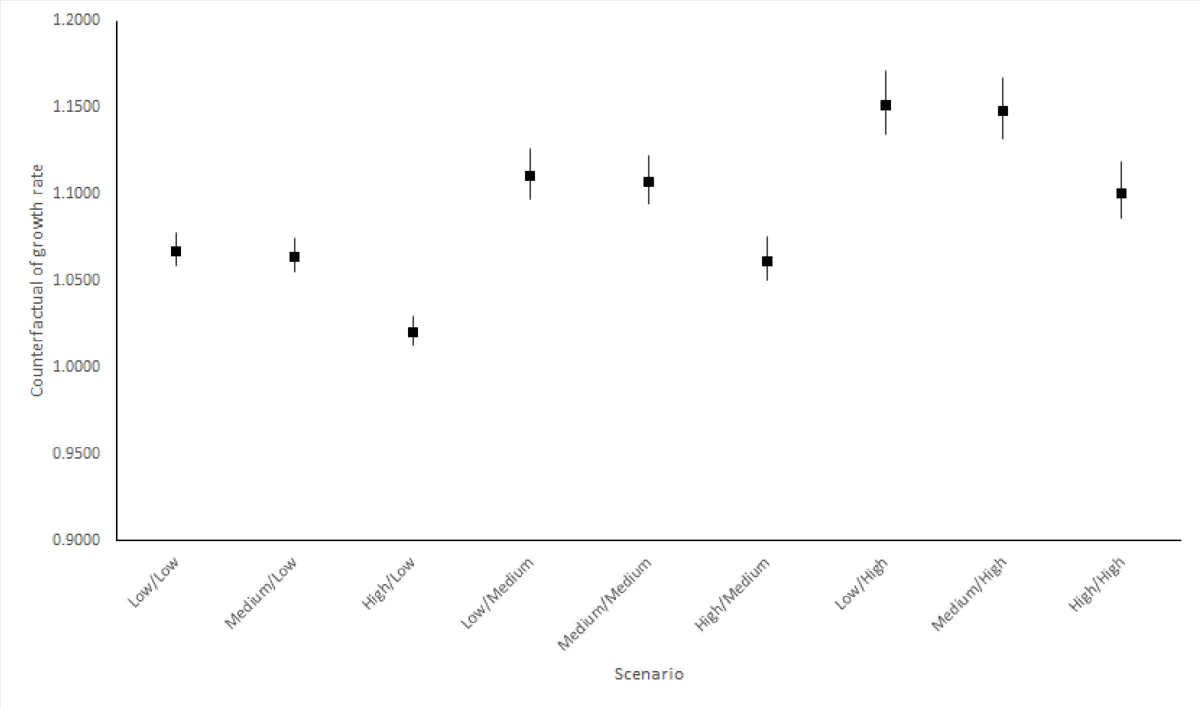



**Figure 52 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Overall, the application of even low compensation through predator management appears to have positive effects on the population of Sandwich terns at NNC SPA. It is important to interpret these results with a level of caution, as the increase in colony productivity is based on a series of assumptions. It is likely that any predator management plans at this SPA would be based on a variety of objectives, and that additional management, such as vegetation control and flood risk management, would be likely. As such, the assumptions made here are more likely to represent a reasonable range of potential compensation outcomes. In addition, the NNC SPA includes the habitat near Stiffkey and Holkham where Sandwich terns have nested successfully in the past. It may be possible to apply similar compensation management actions to this area and increase the overall protected nesting habitat within the SPA. Any detailed compensation plans would need to consider these further management actions in order to improve the likelihood of success.

*7.7.1.2 Sandeel fisheries closure*

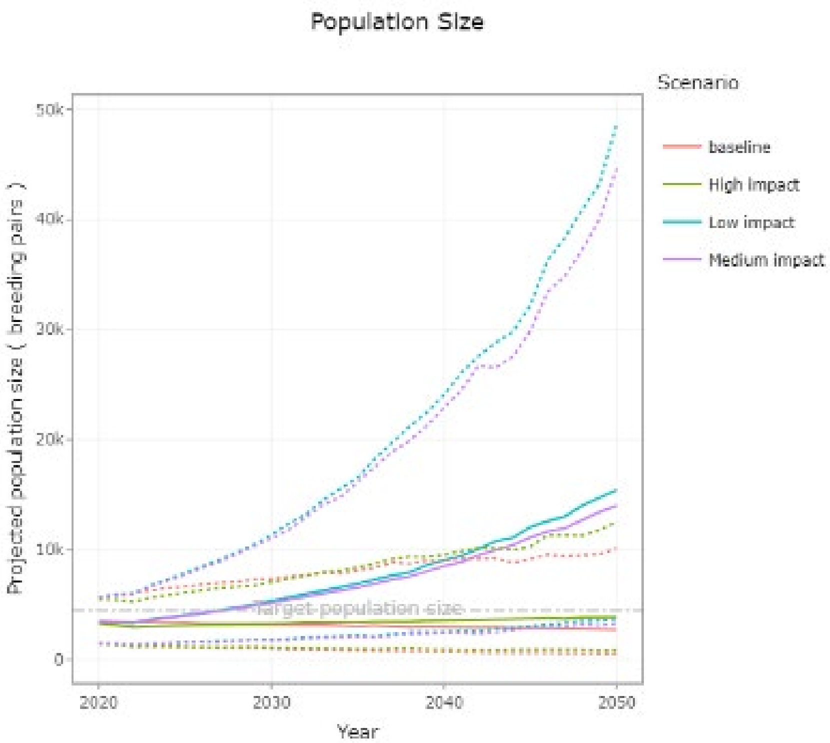
Frederiksen and Wanless (2006) reported that the closure of the sandeel fishery off the east coast of Scotland resulted in the productivity of Sandwich tern colonies in the region being 0.24 chicks per pair higher than before the closure. This was assumed to be a "high" compensation scenario if any fishery for the sandeel stock that determines the density of sandeel available in inshore waters and so likely to be exploited by Sandwich terns were closed. It was therefore assumed that half this value would represent the "low" compensation scenario, and the mid-point between the low and high values was the "medium" compensation scenario. Sandeels are known to form an important part of the diet of Sandwich terns in North Norfolk (Perrow et al. 2010), though in the years studied clupeids were a larger proportion of the diet. It is not known how the diet of

Sandwich terns studied in North Norfolk by Perrow et al. (2010) differed from the diet of birds studied by Frederiksen and Wanless (2006).

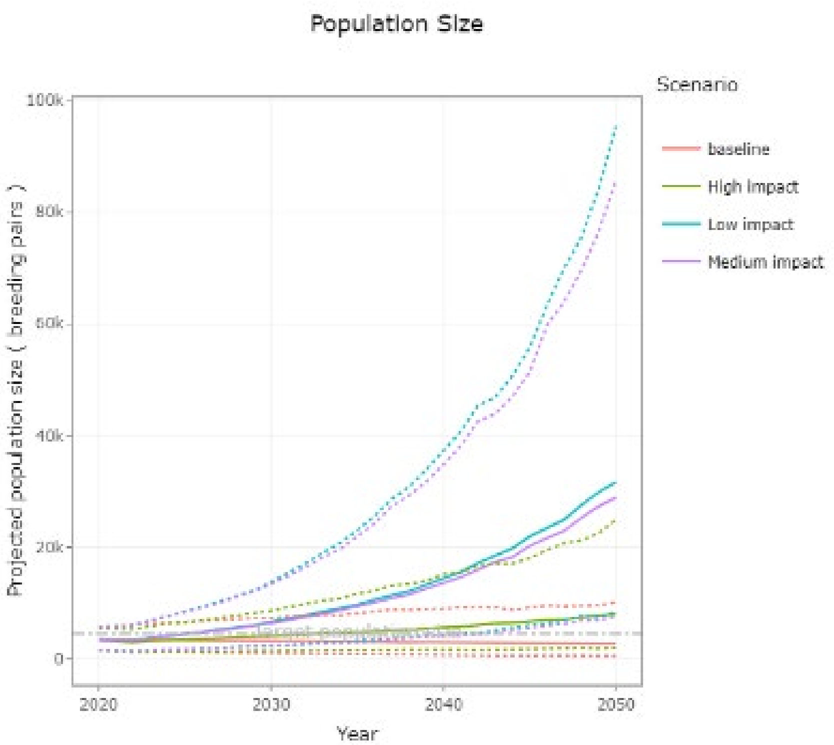
**Table 132 Values for low, medium and high sandeel compensation scenarios for Sandwich terns at NNC SPA.**

| Compensation scenario | "Low" | "Medium" | "High" |
|---|---|---|---|
| Increase in productivity (chicks per pair) | 0.12 | 0.18 | 0.24 |

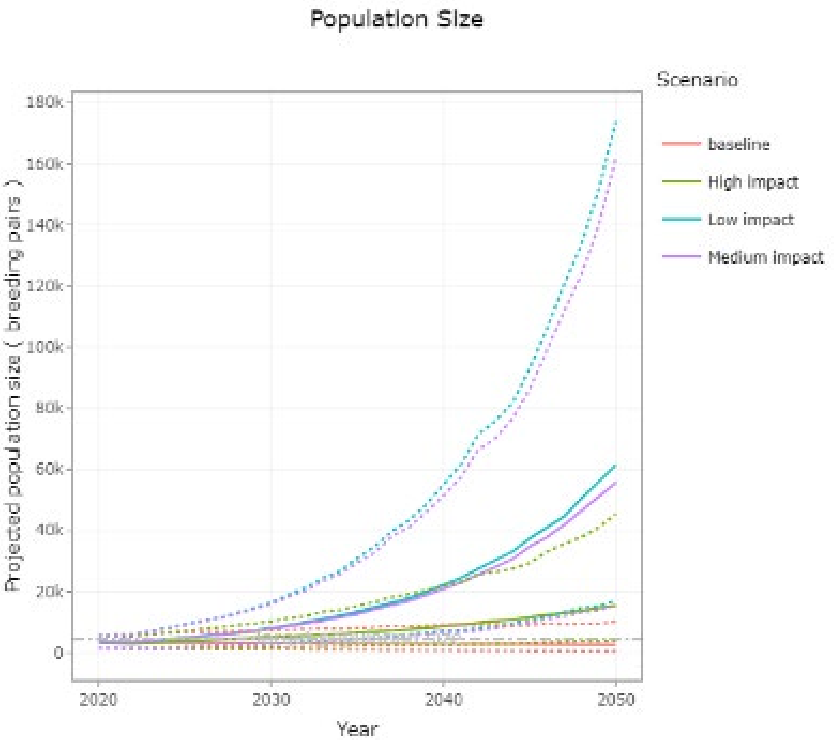
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on productivity (i.e. additional 0.12 (low), 0.18 (medium) or 0.24 (high) chicks per pair) (Figure 53).



Low compensation




Medium compensation




High compensation

**Figure 53 Projected population size of breeding Sandwich tern (pairs) at NNC SPA comparing baseline with the low, medium and high impact scenarios combined with low, medium and high compensation scenarios from sandeel fisheries closure.**

In all scenarios the additional productivity resulted in the projected population size increasing (Table 133). The population size increases shown are all likely to be unrealistic, as they are assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, all three scenarios suggested that if Sandwich tern at NNC SPA rely heavily on a sandeel stock that is commercially exploited, and that fishery is closed as a compensatory measure, the increase in productivity that might result is likely to result in increases in the Sandwich tern population at NNC SPA relative to a scenario in which productivity remains at its current level.

**Table 133 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 0.9937 |
| Low | Low | 1.0505 |
| Low | Medium | 1.0750 |
| Low | High | 1.0982 |
| Medium | Low | 1.0469 |
| Medium | Medium | 1.0720 |
| Medium | High | 1.0947 |
| High | Low | 1.0046 |
| High | Medium | 1.0281 |
| High | High | 1.0499 |

In all impact and compensation scenarios, excepting one of high impact but low compensation benefit, the target population size was exceeded within the time span of the population projection (30 years). The PVA showed that the population was more sensitive to the scale of change in compensation than the scale of the impacts modelled (Table 134).

**Table 134 Year in which the projected median population size exceeded the target population size for each combination of impact and compensation scenario.**

| Impact | Compensation | Year target population size exceeded |
|---|---|---|
| Low | Low | 2027 |
| Low | Medium | 2026 |
| Low | High | 2025 |
| Medium | Low | 2028 |
| Medium | Medium | 2026 |
| Medium | High | 2025 |
| High | Low | Not achieved |
| High | Medium | 2034 |
| High | High | 2029 |

Examination of the counterfactuals of population size showed a relatively small difference between the low and medium impact scenarios and that these impacts were successfully compensated by all three compensation scenarios, i.e. all values were above one (Figure 54). The high impact scenario was sufficiently large that the compensation scenarios were less likely to be successful, but these all showed that the population would increase as all scenarios had a median annual growth rate greater than one.

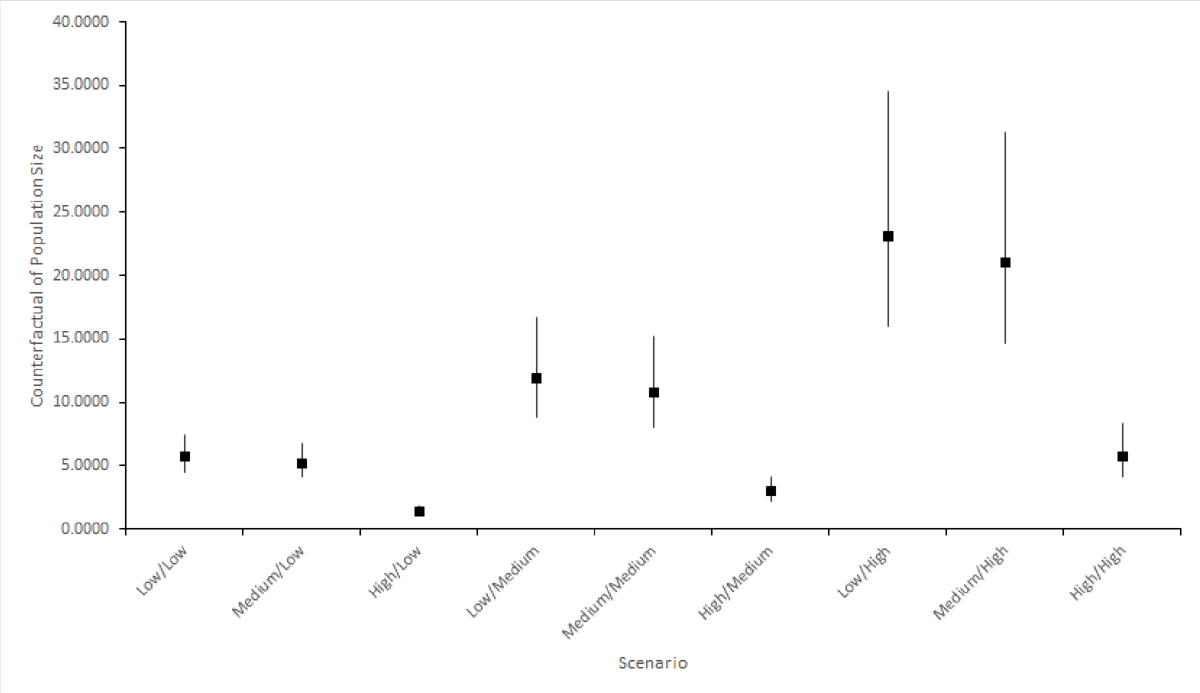



**Figure 54 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed that the population was less sensitive to the change from the low to medium impact scenario than the change from medium to high (Figure 55). The high impact scenario was sufficiently large that compensation was much less likely to be successful, but in these scenarios the CGR still remained greater than one.

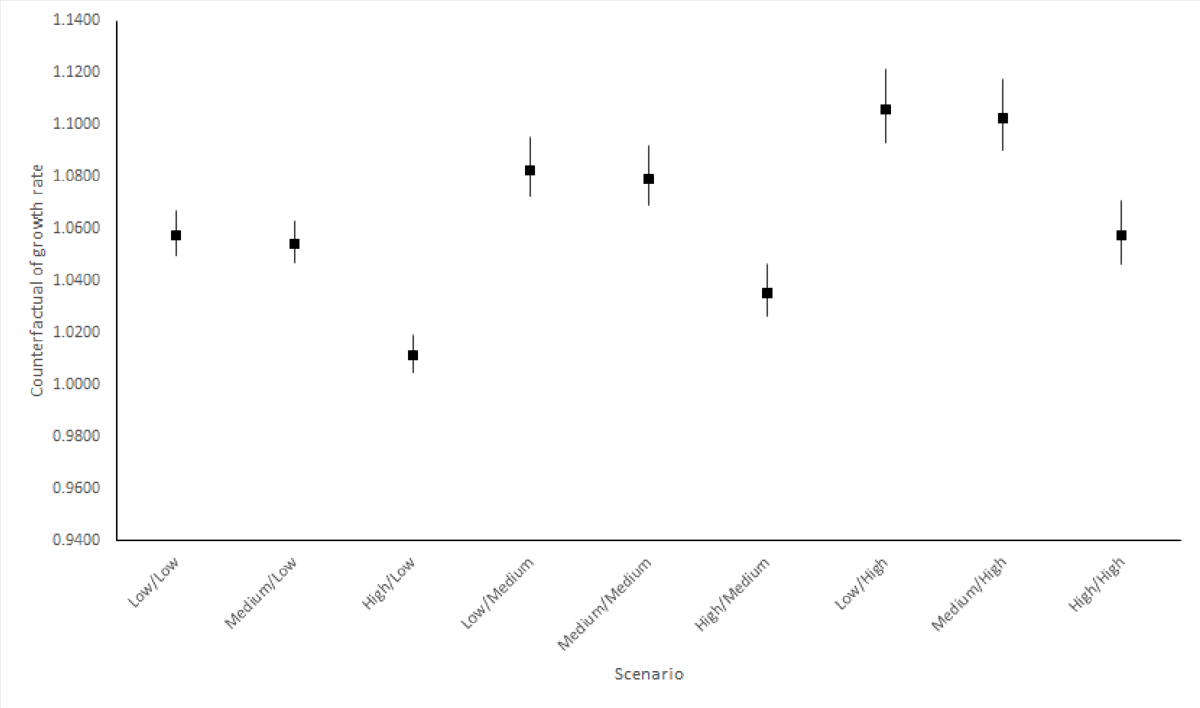



**Figure 55 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

The PVA results suggested that if Sandwich terns at NNC rely heavily on sandeel which they take from a stock that is currently heavily commercially fished, that the increases in productivity that might reasonably be expected to follow closure of that fishery would likely result in positive population change for the NNC SPA Sandwich tern population under all of the impact scenarios.

#### *7.7.1.3 Compensation ratios*

The levels of impact that compensation measures would need to overcome were calculated for 1:3 and 1:6 ratios (Table 135). The 1:1 ratio impacts were tested above and the high impact scenario was considered too large to be compensated for at any level. The high impact scenario at 1:3 and 1:6 was much larger than the high impact ratio at 1:1. These exceeded 100% of the current mortality level and so could not be compensated. The low and medium impact scenarios at both 1:3 and 1:6 ratios were all well below the high impact scenario at 1:1, so these scenarios could be compensated by the exclusion of foxes from the colony. Consequently, further PVA runs were unnecessary as the impacts were either well within the high impact scenario, or sufficiently larger than the high impact scenario that it is clear that the impact could not be compensated for.

**Table 135 Low, Medium and High impact scenarios at 1:3 and 1:6 ratios.**

| Impact scenario | Ratio | Low | Medium | High |
|---|---|---|---|---|
| Additional mortality (birds) | 1:3 | 21 | 75 | 804 |
| Additional mortality (rate) | | 3% | 11% | 114% |

| Impact scenario | Ratio | Low | Medium | High |
|---|---|---|---|---|
| Additional mortality (birds) | 1:6 | 42 | 150 | 1608 |
| Additional mortality (rate) | | 6% | 21% | 227% |

### 7.7.2 EAA 1

#### *7.7.2.1 Predator control*

There is one other SPA within the EAA 1 area: the Alde-Ore Estuary SPA. The Sandwich tern population in the SPA fluctuated between about 50 and 300 pairs in the 1980s and 1990s (JNCC SMP database). Since 1995 there have been very few nesting attempts in the SPA which Ratcliffe et al. (2000) attributed to fox predation. Applying the predator control methods described above to keep foxes from suitable habitat in the SPA may be a suitable compensation measure.

#### *7.7.2.2 Sandeel fisheries closure*

Population level assessment was not completed for the only other SPA in the EAA 1 area: the Alde-Ore Estuary SPA. While, in the absence of any other significant constraint, this colony would be expected to benefit from a closure of UK waters to sandeel fisheries, the absence of Sandwich terns from this colony, likely due to fox predation, means this measure would probably not benefit the colony without also addressing the predation issue.

### 7.7.3 All other EAAs in the UK

There are 14 further SPAs that occur within the remaining EAAs in the UK. Five are on the North Sea coast of Scotland and northeast England, and one, Foulness, is south of the NNC SPA on the Thames Estuary. Three are on the south coast of England and six are on the coast of the Irish Sea.

#### *7.7.3.1 Predator control*

Predator exclusion or control measures would likely benefit several SPAs elsewhere in the UK. The Loch of Strathbeg SPA colony suffered fox predation and no longer nest in the SPA. Other ground nesting terns and gulls currently suffer from otter predation. So, site specific measures to improve predator free nesting location would likely benefit this colony. The Ythan Estuary, Sands of Forvie and Meikle Loch SPA currently has a healthy Sandwich tern population, largely due to ongoing management to exclude foxes from the colony, so would not be a suitable site for compensation. Sandwich terns in the Forth Islands SPA have declined, though it seems this is likely due to a combination of herring gull predation pressure and growth of vegetation on some of the islands in the SPA. Vegetation management and provision of tern nest boxes to protect eggs and chicks have been successful on the Isle of May and these measures could be applied to other islands in the SPA, particularly Inchmickery and Fidra, where Sandwich terns have nested in the past. The colony on the Farne Islands has been declining since its peak in the early 1980s (JNCC SMP database). This has largely been due to the increase in rank vegetation following the extirpation of rabbits from the SPA. However, it appears that management plans are currently in place to better manage vegetation so this may not be a suitable site for compensation. The nearby Coquet Island SPA has a healthy Sandwich tern population, which has required ongoing management of vegetation by the RSPB. This would not be a suitable location for compensation. At Foulness SPA Sandwich terns have not bred since the late 1990s (JNCC SMP database). Ratcliffe et al (2000) stated that this was due to fox predation, so applying suitable fox exclusion methods (Short 2020) could be successful. Therefore, this is likely to be a suitable location for compensation measures. The Poole Harbour SPA Sandwich tern population has generally increased since the 1970s and has fluctuated around 150 to 250 pairs in the 2010s. However, there have been no counts in the JNCC SMP database since 2015, so the current population size is uncertain. It seems that this site would not be suitable for compensation, but more recent colony counts would be needed to confirm this. The Sandwich tern colony in Langstone Harbour, part of the Chichester and Langstone Harbours SPA, suffered from predation which largely stopped breeding (Mavor et al. 2008), with only sporadic breeding since (JNCC SMP database). This would therefore likely be a suitable location for compensation measures. There have been multiple records of Sandwich tern colonies in the Morecambe and Duddon Estuaries SPA being predated and this has had strong negative effects on colony size (Walsh et al. 1994, Thompson et al. 1996, Ratcliffe et al. 2000). This would also likely be a suitable location for compensation measures. At the remaining SPAs in all other EAAs in the UK there was no evidence for the presence or absence of predation causing limitation on Sandwich tern colonies.

#### *7.7.3.2 Sandeel fisheries closure*

The closure of UK waters to sandeel fisheries would be expected to mainly benefit colonies in areas that are currently, or have previously been, subject to sandeel fisheries. There are nine SPAs designated for their Sandwich tern populations in all other EAAs in the UK that are not on the North Sea coast. It is therefore considered unlikely that closure of sandeel fisheries would have the same level of benefit to Sandwich tern populations in those SPAs. However, there are six SPAs in other EAAs in the UK that are on the North Sea coast. Terns at these SPAs would be likely to benefit from sandeel fisheries closures in the UK. These SPAs are Loch of Strathbeg, Ythan Estuary, Sands of Forvie and Meikle Loch, Forth Islands, Coquet Island, and Farne Islands.

## 7.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the two recommended compensation measures were undertaken for the control of predators compensation approach and the closure of sandeel and sprat fisheries approach. The summary table for the predator control approach to compensation is given in Table 136 and the PVA assessment is shown in Table 137. The summary table for the sandeel and sprat fisheries closure approach to compensation is given in Table 138. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 139 (predator control method), Table 140 (PVA assessment method) and Table 141 (sandeel and sprat fisheries closure).

**Table 136 Assessment of confidence in the predator control method of compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Predator proof fences exclude predators | n/a | Young et al. 2012, Furness et al. 2013, RSPB 2020, Xcluder 2020 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Sandwich terns are affected by fox predation | n/a | Walsh et al. 1991, Ratcliffe et al. 2000 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 137 Assessment of confidence in the PVA assessment of the predator control and sandeel closure method comparing impact scenarios to baseline conditions.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 3 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 3 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 3,563 in 2012 | DECC strategic AA | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | mean: 0.4461, sd: 0.3281 | Overall mean chicks per pair (2011 - 2020) at Scot Head & Blakeney Point | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Adult survival rate | mean: 0.907, sd: 0.108 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 0 to 1 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age class 2 to 3 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Predator control effects on productivity | Low compensation = -0.1422 | Calculated as above | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | Medium compensation = -0.2518 | Calculated as above | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | High compensation = -0.3714 | Calculated as above | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| Sandeel fisheries closure effects on adult survival | Low compensation = -0.000982594 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | Medium compensation = -0.003509264 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | High compensation = -0.03754913 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 138 Assessment of confidence in the recommended compensation method of closure of sandeel or sprat fisheries.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Diet of Sandwich terns | n/a | Courtens et al. 2017 | Robust | Robust | Robust | Robust | Robust | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Shortage of forage fish affects adult survival and breeding success | n/a | Stienen et al. 2015, Fijn et al. 2017 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Food shortage reduced productivity at UK colonies | n/a | Furness et al. 2013 | Robust | Robust | Medium | Robust | ROBUST | MEDIUM | HIGH |
| Effects of sandeel shortage similar to effects on kittiwake | n/a | Frederiksen and Wanless 2006 | Robust | Robust | Medium | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 139 Narratives justifying the evidence and applicability scores for metrics used in confidence in the predator control method of compensation.**

| Metric | Narrative |
|---|---|
| Predator proof fences exclude predators | It is clear that there is robust evidence for the use of predator proof fencing at excluding the predators that affect Sandwich terns, particularly foxes. None of the evidence was from the SPA of interest, so applicability was medium. Overall confidence was therefore high. |
| Sandwich terns are affected by fox predation | It is very clear from multiple sources that nesting Sandwich terns are particularly affected by fox predation, so evidence was robust. None of the evidence presented here was from the SPA of interest so the applicability was medium. However, it is highly likely that Sandwich terns at NNC SPA would also be negatively affected by fox predation. Overall confidence was high. |
| **OVERALL CONFIDENCE SCORE** | The confidence in the assessment of the evidence was robust and the applicability was medium. Confidence in each measure was high, so an overall score of high was given. |

**Table 140 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from the closure of sandeel or sprat fisheries.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of Sandwich terns is well established and not in question. It is not variable between populations and is directly applicable to the NNC SPA population. So, confidence was very high. |
| Upper constraint on productivity | Sandwich terns lay up to three eggs and so productivity cannot be above this. Clutch size is usually one or two eggs, with less than 1% of clutches being three eggs (Smith 1975). Confidence was very high. |
| Initial population size | The Sandwich tern population at NNC SPA has had annual counts at each colony since 1969. The counts are based on agreed standardised methodologies, so the data are considered robust and sample size is high. These are counts of the colony being modelled so the applicability was also high. Thus, there was very high confidence in these data. |
| Productivity rate per pair | The productivity rate was based on long term monitoring of the productivity of the colonies within the spa. These used standardised agreed methodologies. Sample sizes were relatively large. Thus, confidence in the overall evidence was high. These were data from the colony being studied so applicability was also high. Thus, an overall confidence score of very high was given. |
| Adult survival rate | The adult survival rate evidence was based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. A medium score was therefore given. With robust evidence and medium applicability, the confidence score was high. |
| Age specific survival rates from 0 to 4 years | The adult survival rate evidence was based on high quality analyses of robustly collected data. These data have been agreed for |

| Metric | Narrative |
|---|---|
| | application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. A medium score was therefore given. With robust evidence and medium applicability, the confidence score was high. |
| **OVERALL CONFIDENCE SCORE** | The confidence in the metrics used as input value for the PVA were all either high or very high, so an overall confidence of high was given. |

**Table 141 Narratives justifying the evidence and applicability scores for metrics used in confidence in the closure of sandeel or sprat fisheries method of compensation.**

| Metric | Narrative |
|---|---|
| Diet of Sandwich terns | The diet of Sandwich terns was important in establishing whether closing the sandeel fishery would be beneficial. There was robust published evidence of this, though the evidence was from Belgium, so applicability was medium. Confidence was therefore high. |
| Shortage of forage fish affects adult survival and breeding success | There was robust evidence that Sandwich tern demographics are negatively affected by a shortage of forage fish. These data were from Belgium and the Netherlands, so applicability was medium, given a confidence score of high. |
| Food shortage reduced productivity at UK colonies | There was robust evidence presented in Furness et al. (2013) that UK Sandwich tern colonies have been negatively affected by food shortages. This evidence was from a variety of location, but not NNC SPA, so applicability was medium. Confidence was scored as high. |
| Effects of sandeel shortage similar to effects on kittiwake | The effects of sandeel shortages on Sandwich tern demographics was shown to be similar to that in kittiwakes, so the evidence was considered robust. These data were from the Isle of May, so applicability was medium, giving a confidence of high. |
| **OVERALL CONFIDENCE SCORE** | The confidence in the assessment of the evidence was robust and the applicability was medium. Confidence in each measure was high, so an overall score of high was given. |

### 7.8.1 Confidence in predator control

With an overall assessment of high in the compensation approach and in the PVA assessment method the assessment of confidence in the proposed predator control compensation against the three impact scenarios needs to be carefully considered. The population level assessment (7.7) showed that all impact scenarios could be compensated for if the modelled increases in productivity assumed for the low, medium, and high compensation scenarios could be achieved by improving the effectiveness of any current predator control measures at NNC SPA. There was little difference between the low and medium impact scenarios for all three compensation scenarios and PVA metrics were all positive for these combinations so, confidence in these was high (Table 142). However, the PVA metrics for the high impact scenario for all three compensation scenarios, while still being positive, was less clear. Consequently, the confidence in these was medium (Table 142). It is important to note that these conclusions are based on the assumption that these measures are not currently in place in the NNC SPA (or in other locations). No published information on current management in NNC SPA could be found in the literature search, so the methods described could be applicable.

**Table 142 Assessment of confidence in the impact/compensation scenarios for Sandwich terns from predator control measures increasing productivity.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | HIGH | MEDIUM |
| Medium compensation | HIGH | HIGH | MEDIUM |
| High compensation | HIGH | HIGH | MEDIUM |

### 7.8.2 Confidence in sandeel fisheries closure

As with the predator control compensation there needs to be careful consideration of the confidence that the compensation measures would be sufficient to overcome the impact scenarios if closure of the sandeel and sprat fisheries increased productivity. The population level assessment (7.7.1.2) also showed that the low and medium impact scenarios could be compensated for by the low, medium, and high compensation scenarios that increased productivity, so confidence in these was high (Table 143). However, it was apparent that compensation for the high impact scenario was less effective. Against a high level of impact, the low and medium compensation scenarios did result in CGR values above one (Figure 55) but not by as much as when combined with low or medium impact scenario, so these were assessed as having medium confidence. However, the high compensation scenario did result in a CGR value as high as the low and medium impacts with low compensation levels, so this was assessed as high confidence (Table 143).

**Table 143 Assessment of confidence in the impact/compensation scenarios for Sandwich terns from sandeel or sprat fisheries closure increasing productivity.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | HIGH | MEDIUM |
| Medium compensation | HIGH | HIGH | MEDIUM |
| High compensation | HIGH | HIGH | HIGH |

## 7.9 Future monitoring and adaptive management

### 7.9.1 Predator control

While evidence supports foxes having important effects on Sandwich tern colonies, a decision on whether other predators may be affecting the colonies may be needed to determine a suitable control measure. Foxes alone appear to be readily controlled through suitable electric fencing and night-time patrols by wardens during the breeding season. If other predators, such as stoats or mink, are present then different methods of exclusion would be necessary. Thus, it is recommended that further monitoring of the presence of terrestrial mammal predators is undertaken to determine which species are present both within and around the colonies. This could use a variety of methods (camera traps, live traps, kill traps, DNA traps).

If permanent anti-predator fencing is used it will be necessary to remove any predators from within the fenced area and demonstrate that they are absent. This would best be achieved in the winter following completion of the fencing, when number should be at their lowest. Even after predators have been removed from fenced areas it will be important to maintain some form of monitoring to enable action to remove any predators that enter the exclosure.

Monitoring should include counts of total colony size, including any nests outside exclosures, and plot counts of productivity. Careful monitoring of at least some nests may be helpful to determine whether other potential negative effects on productivity are occurring (e.g. food provisioning of chicks, disturbance of nests by other predators or human actions) to provide information for further potential management actions. For instance, if productivity doesn't increase, despite excluding terrestrial predators, it would be important to know the reasons for the constraint. Monitoring of the colony size and productivity should be annual, at least initially. If the compensation approach is proved to be successful from initial monitoring, then monitoring could be reduced in frequency or intensity. Monitoring should be continued throughout the period that compensation is required.

If the only predators that need to be excluded are foxes, then seasonal electric fencing and wardens undertaking regular night-time checks will likely suffice. Electric fencing typically needs checking on a daily basis to ensure that it is fully functional and has not been breached. Common problems with electric fencing for tern colonies include issues with earthing of fences in dry and well drained environments, vegetation growth shorting out parts of fencing and in sand dune habitats windblown sand covering the fence. Because of these issues fencing needs daily inspection and action to maintain its efficacy.

For any type of exclosure fencing there will need to be continual monitoring of the condition of the fencing, at least immediately before and through the breeding season. There will need to be budget and suitable staff or contractors available to undertake both regular maintenance or emergency repairs throughout the compensation period.

#### 7.9.2 Sandeel fisheries closure

If compensation through closure of the UK sandeel and sprat fisheries was applied, it would be important that suitable monitoring is put in place to demonstrate that this has been effective at:

- Increasing the sandeel and/or sprat stock available to the Sandwich tern population at NNC SPA;
- Increasing the population size of Sandwich terns at NNC SPA; and
- Increasing the productivity of Sandwich terns at NNC SPA.

Methods for suitable monitoring of sandeel or sprat stocks would need to be established with experts in this field and is beyond the scope of this study. Monitoring would need to determine overall abundance of the stocks and perhaps also the stocks within the foraging range of Sandwich terns from the NNC SPA to ensure that the compensation measure is having the desired effect on the prey resource for the population at NNC SPA.

In addition to assessing the stock at both the relevant ICES stock area and within the foraging range of the NNC SPA Sandwich tern population, monitoring of the change in the population at the colony would be important.

These monitoring measures need to be connected to adaptive management decision making. The proposed monitoring needs to be considered together when adapting the management to the results of the monitoring. The aim of the proposed compensation is to increase the Sandwich tern population at the NNC SPA colony. Monitoring of sandeel and/or sprat stocks is needed to determine whether recovery of the stock was as expected, below the level expected or above the level expected. Similarly adaptive management will need to consider whether action is necessary if the change in population size is above or below the expected value. Ultimately the need to adapt management actions will need to be based on whether the population size at the SPA changes as a result of the proposed compensation method. It may be necessary to move to other compensation mechanisms should the closure of the sandeel fishery ultimately prove to be unsuccessful.

#### 7.9.3 Future research

One of the more important elements missing from the assessment of the effects of sandeel fisheries closure on Sandwich tern populations was the effect on adult survival. It is possible that the adult survival of Sandwich terns can be shown to either be negatively impacted by sandeel stock declines, or positively affected by sandeel stock recovery. An assessment of the available data to undertake such an analysis would be a useful first step, followed by survival analysis from ringing data at suitable colonies. Since relatively small impacts on adult survival have greater effects on population growth rate than equivalent impacts on productivity, understanding any relationship between survival and sandeel stock would be particularly valuable.

### 7.10 Summary

The review found that the main forms of compensation recommended by Furness et al. (2013) remain the key methods that could be deployed for Sandwich terns at NNC SPA: fencing out foxes from colonies, stoat control/eradication, flood and vegetation control at colonies, closure of sandeel and sprat fisheries close to Sandwich tern colonies. Since Furness et al. (2013) no new evidence was found on the benefits of the recommended compensation measures.

PVA suggested that the NNC SPA population of Sandwich terns should be decreasing and that this decrease would remain likely with all impact levels. However, evidence suggests that the NNC SPA population has been stable in the longer term and increasing more recently. Three levels of compensation measure were then assessed against the three levels of impact. The potential increase in productivity from predator exclusion from colonies in the SPA were tested using the PVA. Results of the PVA suggested that population increase (from a decreasing baseline) was likely for all compensation levels combined with all impact levels tested.

The increase of productivity of Sandwich terns from sandeel and sprat fisheries closure were tested in the PVA model. The PVA results suggested that population would increase for all impact levels and all compensation levels. These results strongly suggest that the Sandwich tern population at NNC SPA is mostly limited by low productivity.

Confidence was high in the effects of predators on Sandwich terns and in methods to prevent this. Confidence in the PVA assessment process was also high. Confidence was high for compensating the low and medium impact scenarios and medium for the high impact scenarios where predators were excluded from colonies. There was mostly high confidence in the ability of sandeel and sprat fisheries closures to compensate for low or medium impact scenarios. For the high impact but low or medium compensation scenarios confidence was medium. However, it is important to take account of the assumptions made in this assessment. The review found no evidence of fox predation or the effects of sandeel availability on the NNC SPA directly. However, robust evidence was available from other Sandwich tern colonies that the types of compensation being suggested can be successful at increasing Sandwich tern colony size.

# 8 ALDE-ORE ESTUARY SPA – BREEDING LESSER BLACK-BACKED GULL

The Alde-Ore Estuary (AOE) SPA is on the east coast of Suffolk and is an estuary complex at the mouth of the Alde, Butley and Ore rivers. The SPA includes Havergate Island and Orfordness. The habitats within the SPA include vegetated shingle, intertidal mud, semi-improved grazing marsh, saltmarsh, and saline lagoons. The site was designated due to, among other features, nationally and internationally breeding and wintering waders, terns, gulls, ducks, swans, and geese, including breeding lesser black-backed gull.

## 8.1 Conservation status of Lesser black-backed gull

Lesser black-backed gull has an IUCN Red List classification of "Least Concern" and the UK population was listed in BOCC 2, 3, and 4 as amber. It is listed by the Birds Directive as a migratory species. The biogeographic population (subspecies *graellsii*, in British Isles south to Portugal) was estimated at 179,000 pairs, of which 110,000 pairs breed in Great Britain and 4,800 pairs in all-Ireland (Mitchell et al. 2004). National surveys found a 29% increase in breeding numbers in the UK from 1969 to 1986, and a 40% increase from 1986 to 2000 (JNCC 2020). JNCC SMP data (JNCC 2020) are unable to provide a trend in abundance in England as the confidence intervals were very wide. In general, there have been declines at coastal colonies throughout England and the rest of the UK, but apparent increases in urban colonies.

Stroud et al. (2016) identified that the SPA suite with breeding lesser black-backed gull as a designated feature has eight qualifying sites in Great Britain: two in Scotland (Ailsa Craig SPA; Forth Islands SPA), five in England (Alde-Ore Estuary SPA; Bowland Fells SPA; Isles of Scilly SPA; Morecambe Bay SPA, now known as Morecambe Bay and Duddon Estuary SPA; Ribble and Alt Estuaries SPA) and one in Wales (Skokholm, Skomer and Middleholm SPA, now known as Skomer, Skokholm and seas off Pembrokeshire SPA). The SPAs in Great Britain were estimated to hold 38% of the Great Britain breeding population of lesser black-backed gull present in 2000 (Stroud et al. 2016). Two sites in Northern Island also qualify: Rathlin Island SPA, and Lough Neagh and Lough Beg SPA.

Apart from the marine extensions at some SPAs for loafing seabirds close to colonies, no sites were listed in the 3rd UKSPA review as designated as marine areas for lesser black-backed gulls (Stroud et al. 2016).

The population size of the lesser black-backed gull at AOE SPA has varied considerably over time. There are two colonies within the SPA. One is located on Orford Ness and the other is on Havergate Island. The SMP database contains counts of lesser black-backed gull at Orford Ness from 1968, when 140 pairs were recorded, to 2018, when 97 pairs were counted (Figure 56). The population size increased rapidly when it peaked in 2000 at an estimated 23,000 pairs. The population then crashed the following year to 5,500 pairs and has declined since then.

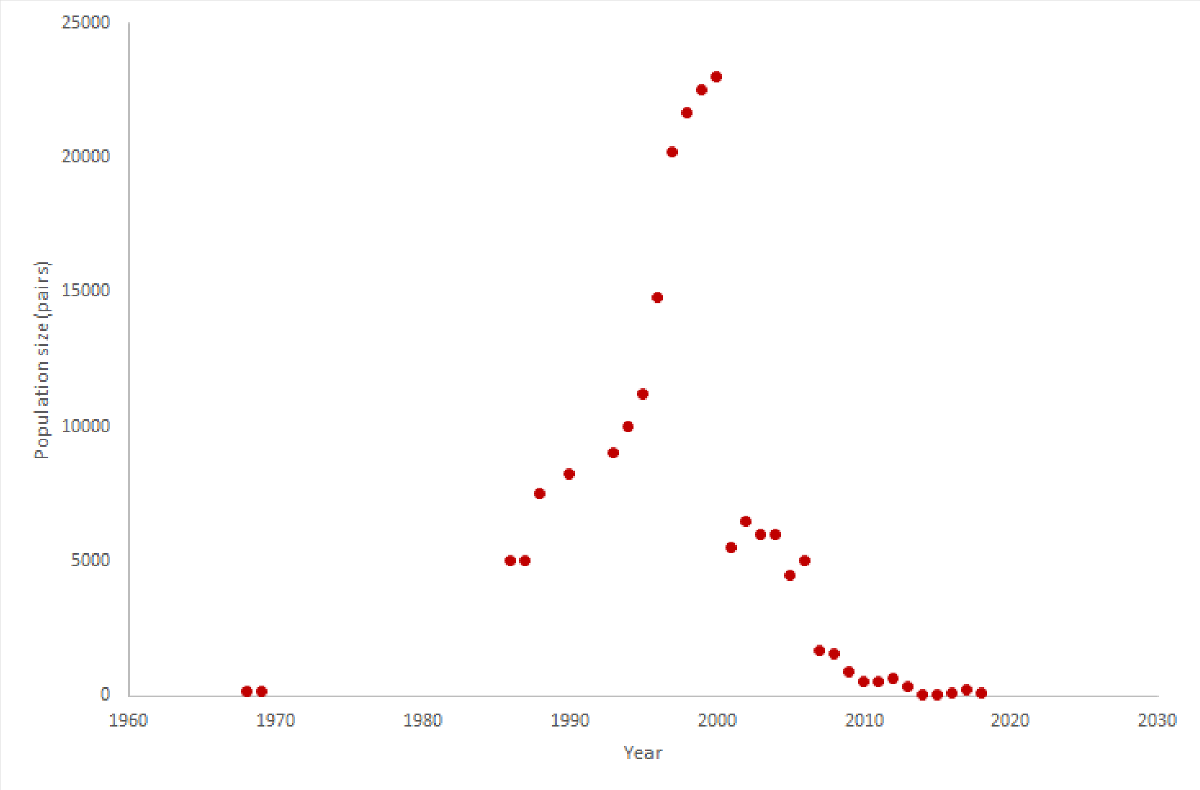



**Figure 56 Numbers of pair of lesser black-backed gull at Orford Ness, Alde-Ore Estuary SPA (data from SMP database).**

The SMP database also contains counts of lesser black-backed gull at Havergate Island from 1961, when two pairs were recorded, to 2018, when 1,670 pairs were counted. The population size has generally been smaller than at Orford Ness, but has shown a rapid increase since 2000, peaking at 2,403 pairs in 2015 with a relatively small decline since then (Figure 57).

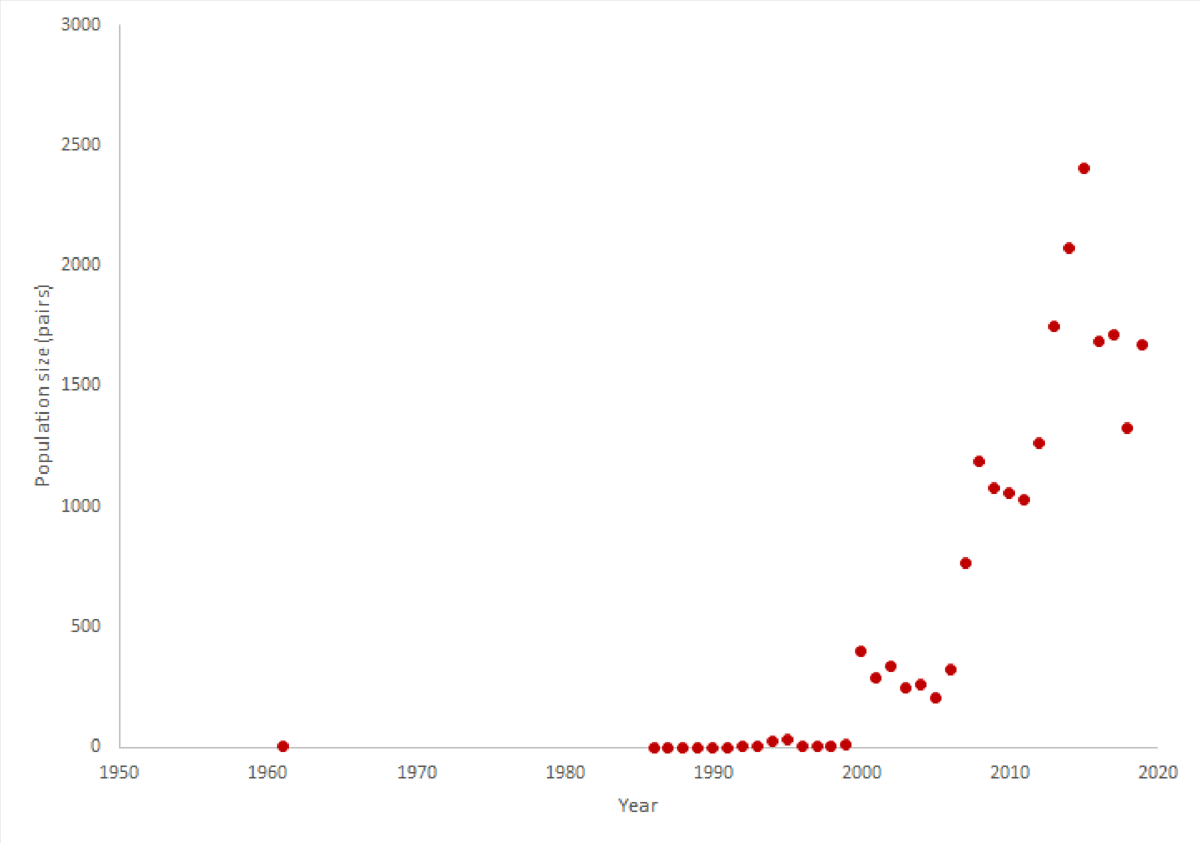



**Figure 57 Numbers of pair of lesser black-backed gull at Havergate Island, Alde-Ore Estuary SPA (data from SMP database).**

Lesser black-backed gulls nest in colonies on the ground, in locations away from mammal predators and human disturbance such as grassy islands, coastal sand-dunes, coastal wetlands and shingle, and moorland areas that can be far from the sea. Some nest in urban areas on roof-tops, and roof-top nesting has increased considerably. Lesser black-backed gulls feed on a wide range of diets, including marine fish caught at sea, or scavenged as discards from fishing boats. However, they can subsist on earthworms, small mammals, insects, and grain in areas of agricultural land. They can scavenge at landfill sites and from agriculture (such as from outdoor animal feeding troughs). After breeding, most migrate to north Africa and Iberia for the winter, although increasing numbers now overwinter in the UK.

Woodward et al. (2019) listed foraging range of breeding lesser black-backed gulls as: mean 43.3 km, mean maximum 127 km and maximum 533 km.

Mitchell et al. (2004) identified the main threats to lesser black-backed gull in the UK as culling of tens of thousands of breeding adults to reduce their impact on nesting terns or on drinking water quality, reductions in food supply due to changes in refuse disposal, and changes in fisheries practices (less discarding of fish). However, numbers culled are very uncertain as this species was on General Licences until 2019 so could be killed without a need to report numbers taken. Habitat change at colonies, where dense growth of taller plants and scrub can make sites unsuitable for ground-nesting, and attraction of predators such as foxes to large gull colonies can also have an impact. Breeding success can be strongly affected by fox predation, but also by American mink and by rats. A few colonies have been identified as affected by outbreaks of botulism, but this appears to have been limited in impact and less than seen in herring gulls. Lesser black-backed gull flight heights suggest moderate risk of collision with offshore wind farm turbines.

### 8.2 Citation population size

The SPA citation (dated January 1996) states that the site qualifies under Article 4.2 of the Birds Directive by supporting "12% of the British population" but gives no population size. However, the Natura 2000 Standard Data Form, dated December 2016, lists the population size as 14,070 pairs.

### 8.3 Conservation objectives

The site's conservation objectives are to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site.

Draft supplementary advice on conserving and restoring site features provides additional information on the conservation objective for lesser black-backed gull at AOE SPA. These are:

- Restore the size of the population to 14,074 pairs whilst avoiding deterioration from its current level as indicated by the latest mean peak count, or equivalent;
- Restore the abundance and structure of the assemblage at or above its current or target level (whichever is the higher) through restoring breeding productivity and adult survival;
- Maintain concentrations and deposition of air pollutants to at or below the site relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System;
- Maintain safe passage of birds moving between roosting and feeding areas. The maximum offshore distance reached was 159 km of breeding colonies;
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- The frequency, duration and/or intensity of disturbance in close proximity to nesting and/or feeding birds should not reach levels that substantially affect the feature;
- Restore the extent, distribution and availability of suitable breeding habitat which supports the feature for all necessary stages of its breeding cycle (courtship, nesting, feeding);

- Maintain the extent and distribution of predominantly medium to tall [i.e. 20-60 cm] grassland swards;
- Restore water quality and quantity to a standard which provides the necessary conditions to support the SPA feature, where the supporting habitats of the feature are dependent on surface water Current EA chemical quality; does not require assessment. Current EQ ecological quality: moderate potential Maintain Dissolved Oxygen (DO) at ≥ 5.7mg l-1 standardised to a salinity of 35 using 5th percentile of DO data (WFD High/Good boundary); and
- Maintain safe passage of birds moving between roosting and feeding areas.

A Site Improvement Plan (SIP) for AOE SPA was published in October 2014[3]. That identified several issues as pressures or threats to breeding lesser black-backed gulls. These were:

- Hydrological changes;
- Public Access/Disturbance;
- Inappropriate pest control; and
- Changes in species distributions;

The SIP identified the need for alternative habitat provision or habitat enhancement opportunities to alleviate the hydrological change pressure as a responsibility of National Trust, Natural England, Suffolk Wildlife Trust, Babcocks, Alde & Ore Estuary Partnership.

Reduction of bird disturbance and trampling of shingle vegetation was listed as a measure to alleviate public access and disturbance with responsibility on Eastern Inshore Fisheries, Conservation Authority (IFCA), Ministry of Defence (MoD), National Trust, Natural England, RSPB, Suffolk Coast & Heaths AONB, Suffolk Coastal District Council, Suffolk County Council, Suffolk Wildlife Trust, Marine Management Organisation (MMO), British Association for Shooting and Conservation (BASC), Joint Nature Conservation Committee (JNCC), Civil Aviation Authority (CAA), Suffolk Little Tern Group, EDF Energy, and Shingle Street residents.

Ensuring adequate protection of nesting birds from predators was the measure intended to tackle inappropriate pest control with National Trust, Natural England, and RSPB being the responsible organisations.

Understanding population dynamics and enabling boundary flexibility/better wider habitat provision was the measure recommended to determine the threat from changes in species distributions with the organisations responsible being the National Trust, Natural England, RSPB, Suffolk Coast & Heaths AONB, Suffolk Coastal District Council, Suffolk Wildlife Trust, British Trust for Ornithology (BTO), LIFE+ Little Tern Project, Suffolk Little Tern Group, and Shingle Street residents.

[3] http://publications.naturalengland.org.uk/publication/4884745984933888

### 8.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the populations of each of the qualifying features.

There is one main sources of impact on lesser black-backed gulls from offshore wind farm development: mortality due to collisions with operational turbines.

The maintenance of the population of each of the qualifying features could be affected directly through collision mortality.

## 8.4 Location of compensation

Cook et al. (2011) did not define Ecological Assessment Areas for lesser black-backed gull. Therefore, the EEAs for herring gull were used as the closest similar species available assessed by Cook et al. (2011). Cook et al. (2011) defined four Ecological Assessment Areas for herring gull (Figure 34). The AOE SPA occurs within EAA 4. Consequently, the hierarchy of the locations of compensation are:

1. AOE SPA;
2. EAA 4; and
3. All other EAAs in the UK.

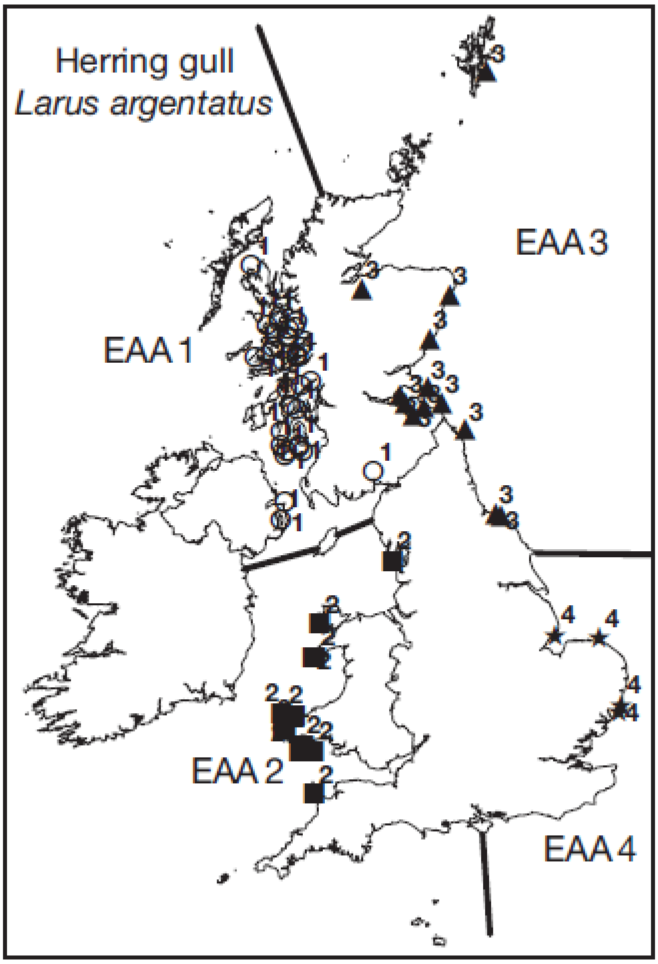



**Figure 58 Ecological Assessment Areas (EAAs) identified by Cook et al. (2011) for herring gull (as a proxy for lesser black-backed gull) by considering regions in which abundance at breeding colonies varies in a consistent fashion. Figures refer to the EAA to which each colony is assigned. Black bars mark boundaries of the EAAs.**

## 8.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for breeding lesser black-backed gulls were developed based on the five potential compensation measures reviewed by Furness et al. (2013). The five potential measures listed were:

1. Mink eradication at lesser black-backed gull colonies;
2. Fencing out foxes from colonies;
3. End culling of lesser black-backed gulls;
4. Closure of sandeel and sprat fisheries; and
5. Eradicate rats at lesser black-backed gull colonies.

Of these five potential measures, all except closure of sandeel and sprat fisheries were thought to be highly likely to be effective with high confidence in that assessment. There was only low confidence in the efficacy of closure of sandeel and sprat fisheries for lesser black-backed gull breeding success because this species does not rely heavily on sandeels but takes a diverse diet with much food derived from terrestrial rather than marine sources.

No culling of lesser black-backed gulls currently occurs at the AOE SPA. The recent changes to the licensed culling of lesser black-backed gulls suggests that culling will be more targeted and based on a narrow set of requirements for control. It now seems less likely that removal of a licensed cull could be an acceptable compensation measure as the problem resulting in the need to the cull would remain.

The key biological questions for compensation measures for lesser black-backed gulls at AOE SPA are provided in Table 144.

**Table 144 Key Biological Questions in assessing the potential for compensation of breeding lesser black-backed gulls at AOE SPA.**

| No. | Key Biological question |
|---|---|
| 1 | Is there evidence of terrestrial mammal predators causing breeding failure or site desertion in: |
| 1.1 | AOE SPA? |
| 1.2 | Colonies in EAA 4? |
| 1.3 | All other EAAs in the UK? |
| 2 | Are predator proof fences effective at excluding predators from seabird colonies? |
| 3 | Is there evidence of successful measures to prevent fox predation on lesser black-backed gull colonies in: |
| 3.1 | AOE SPA? |
| 3.2 | Colonies in EAA 4? |
| 3.3 | All other EAAs in the UK? |

### 8.6 Review of potential compensation measures

For each of the proposed compensation measures considered in 8.5 a review of the evidence base was completed to determine whether new relevant information was available to update the advice in Furness et al. (2013).

Literature published since 2013 does not add significantly to any of the above conclusions, and certainly does not contradict the conclusions reached in 2013. The most practical of the above measures is likely to be fencing out foxes from colonies, where predator-proof fencing can be established at mainland gull colonies. Eradication of rats or mink at island colonies would also be practical and appropriate, providing biosecurity measures can be put in place to minimize risk of recolonisation by rats or mink. However, it should be recognised that there could be an unintended consequence of this approach; increasing gull numbers on offshore islands with seabird colonies can result in conflicts resulting from impacts of gulls on smaller seabirds, and the latter may well be named features of the SPA. Management of predators on seabird islands to benefit gulls may result in declines in other seabirds, which may also be unwanted. Any proposed compensation measures that could impact on a SPA would require an Appropriate Assessment to be completed prior to any conservation management action (including compensation) taking place.

### 8.6.1 Fencing out terrestrial predators

It was assumed that predator management to compensate for impacts to the AOE SPA lesser black-backed gull colony would be applied preferentially to the SPA colony itself or in adjacent areas to the SPA that would benefit the colony, rather than to another colony elsewhere.

While there is no evidence of rat predation on the colony at AOE SPA, it may occur. However, it is apparent that fox predation is an important problem at the AOE SPA colony, in particular at the Orford Ness part of the SPA (Mavor et al. 2001). Fox predation directly, and the disturbance to the colony caused by foxes, has resulted in the population on Orfordness declining from 23,000 pairs in 2000 to a few tens of pairs nesting on the roofs of buildings.

Fox control can be carried out using lethal control, usually shooting at night using spotlights ('lamping'), or through anti-predator fencing. Electric mesh fencing can be effective at keeping foxes out of gull and tern colonies but is highly dependent on regular (daily) testing and maintenance. This can cause disturbance to the colony being protected, as well as other species of importance in the area. More effective, and permanent, anti-predator fencing solutions are available that require less maintenance to remain effective through the breeding season (e.g. Young et al. 2012).

In the AOE SPA anti-predator fencing could be installed at either Orford Ness or Havergate Island to enhance the breeding habitat for lesser black-backed gulls. There have been conflicts between gulls and other SPA qualifying features in the AOE SPA, most notably between lesser black-backed gulls and avocets at Havergate Island. Any fencing to increase the lesser black-backed gull colony at Havergate Island would need to carefully consider this interaction and whether the management objectives of the site could still be achieved with a larger gull colony on Havergate. It may be more appropriate to construct predator-proof fencing on part of Orfordness, especially because that area is already subject to considerable modification by earlier developments associated with weapons testing and military presence.

#### *8.6.1.1 Answers to the key biological questions (8.5).*

The answers to the key biological question in relation to compensation through fencing out terrestrial predators are shown in Table 145.

**Table 145 Answers to Key Biological Questions in assessing the potential for compensation through fencing out terrestrial predators.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence of terrestrial mammal predators causing breeding failure or site desertion in: | |
| 1.1 | AOE SPA? | Yes. There is no evidence of rat predation on the colony at AOE SPA. However, fox predation is an important problem at the AOE SPA colony, in particular at the Orford Ness part of the SPA (Mavor et al. 2001). |
| 1.2 | Colonies in EAA 4? | No. There are no other SPAs designated for their breeding lesser black-backed gull populations in EAA 4. |
| 1.3 | All other EAAs in the UK? | Yes. There is evidence of fox predation causing reductions in colonies size at Morecambe Bay and Duddon Estuary SPA. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2 | Are predator proof fences effective at excluding predators from seabird colonies? | Yes. There is good evidence from Young et al. (2012), Furness et al. (2013), RSPB (2020), and Xcluder (2020) that suitable predator proof fencing is effective at excluding predators, such as foxes. |
| 3 | Is there evidence of successful measures to prevent predation on lesser black-backed gull colonies in: | |
| 3.1 | AOE SPA? | No. There are predator management practices in place at Havergate Island, albeit not to increase the size of the lesser black-backed gulls colony. However, there are no predator control measures in place at other suitable nesting locations in the SPA (Orfordness). |
| 3.2 | Colonies in EAA 4? | No. There are no other SPAs designated for their breeding lesser black-backed gull populations in EAA 4. |
| 3.3 | All other EAAs in the UK? | Partly. There is existing management ongoing in the Morecambe Bay and Duddon Estuary SPA. However, this has had mixed success. The use of electric fences to exclude foxes at South Walney was studied by Davis et al. (2018). They found that foxes were able to enter the ex-closures due to problems with electricity supply to fences. Despite this they found productivity to be statistically significantly higher than expected when compared with unfenced areas. This provided evidence that controlling mammalian predator access to nesting lesser black-backed gulls can increase productivity. |

## 8.7 Population level assessment

### 8.7.1 AOE SPA

All the population level assessments for AOE SPA lesser black-backed gulls were based on three levels of potential impact. The low impact scenario was based on an impact of 1% of baseline adult mortality rate. For a population size of 2,000 pairs (JNCC 2014) and an adult survival rate of 0.885 a 1% increase in baseline mortality would be 4.6 additional birds being killed per annum.

The medium impact scenario was based on a pro-rata increase in the installed capacity of proposed Round 4 offshore wind farms (an additional 7GW) compared to the current installed capacity (26GW). This resulted in an estimated mortality of 14 birds per annum.

The high impact scenario was based on the current in-combination impact on the population from all offshore wind farms (53 birds killed per annum) pro-rated to the 2050 net zero target of 100GW of installed capacity. This is an additional 74GW of additional capacity compared to the current level of installed, consented, or planned capacity (26GW). This results in an additional mortality of 151 adult birds per annum, or 32.8% increase in adult mortality rate. Impact levels are summarised in Table 146.

**Table 146 Values for low, medium and high impact scenarios for lesser black-backed gulls at AOE SPA.**

| Impact scenario | Low | Medium | High |
|---|---|---|---|
| Additional mortality (birds) | 4.6 | 14 | 151 |
| Additional mortality (rate) | 1% | 3.0% | 32.8% |

The baseline population was compared against the low, medium and high impact scenarios using the Seabird PVA Tool with the input parameters shown in Table 147. The only compensation measure that could be effective was considered to be predator control scenarios, with focus on the application of suitable fencing (and other related management actions). It was assumed that providing fenced exclosures (or similarly effective predator control means) would allow the existing population of 2,000 pairs at the SPA to increase their productivity from a very low 0.33 chicks per pair. The low compensation scenario was based on the lower range of productivity values suggested in the NE "Alde Ore Estuary Special Protection Area: DRAFT Supplementary advice on conserving and restoring site features" document. This document suggests that the lower value of a range of productivity of lesser black-backed gulls breeding at natural sites, such as the AOE SPA, "without significant controlling factors" was 0.43 chicks per pair. This same document also suggested an upper value of productivity of lesser black-backed gulls breeding at natural sites of 0.69 chicks per pair. This was used as the high compensation scenario. The medium compensation scenario assumed that the national (UK) average between 1989 and 2018 of productivity at natural sites would be suitable; 0.52 chicks per pair (JNCC 2020).

**Table 147 PVA input parameters for predator control compensation scenarios.**

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| Density dependent? | No | | | n/a |
| Stochastic? | Yes | | | n/a |
| No. of simulations | 1000 | | | n/a |
| Random number seed | 1 | | | n/a |
| Burn in time (years) | 5 | | | n/a |
| Age at first breeding | 5 | | | PVA app default |
| upper constraint on productivity | 3 chicks per pair | | | PVA app default |
| Initial population size | 2,000 pairs in 2015 | | | JNCC (2014) |
| Productivity rate per pair | mean: 0.33, sd: 0.1756 | | | NE DRAFT Supplementary advice on conserving and restoring site features |
| Adult survival rate | mean: 0.885, sd: 0.056 | | | PVA app "National" default value |
| Age class 0 to 1 | mean: 0.82, sd: 0.056 (SD assumed same as other age clases) | | | PVA app "National" default value |

| Model parameter | Compensation level | | | Source |
|---|---|---|---|---|
| | Low | Medium | High | |
| Age class 1 to 2 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Age class 2 to 3 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Age class 3 to 4 | mean: 0.8283, sd: 0.077 | | | PVA app "National" default value |
| Years impacts begin and end | 2020 to 2050 | | | n/a |
| **Impact scenarios** | | | | |
| Impact on productivity rate | 0 | | | n/a |
| Impact on adult survival rate | Low | Medium | High | Calculated as above |
| | 0.00115, se: NA (4.6 birds) | 0.0035, se: NA (14 birds) | 0.03775, se: NA (151 birds) | |
| **Predator control measures (fencing)** | | | | |
| Impact on productivity rate | -0.29 | -0.56 | -1.07 | Low and High from NE DRAFT Supplementary advice on conserving and restoring site features. Medium (JNCC 2020) |
| First year to include in outputs | 2020 | | | n/a |
| Final year to include in outputs | 2050 | | | n/a |
| Target population size | 14,074 pairs | | | SACO TPS for AOE SPA |

**PVA results**

The baseline population projection was compared with the three impact scenarios (Table 147). The projections all showed a decline in the population size with time for baseline (unimpacted) and all three impact scenarios (Figure 59).

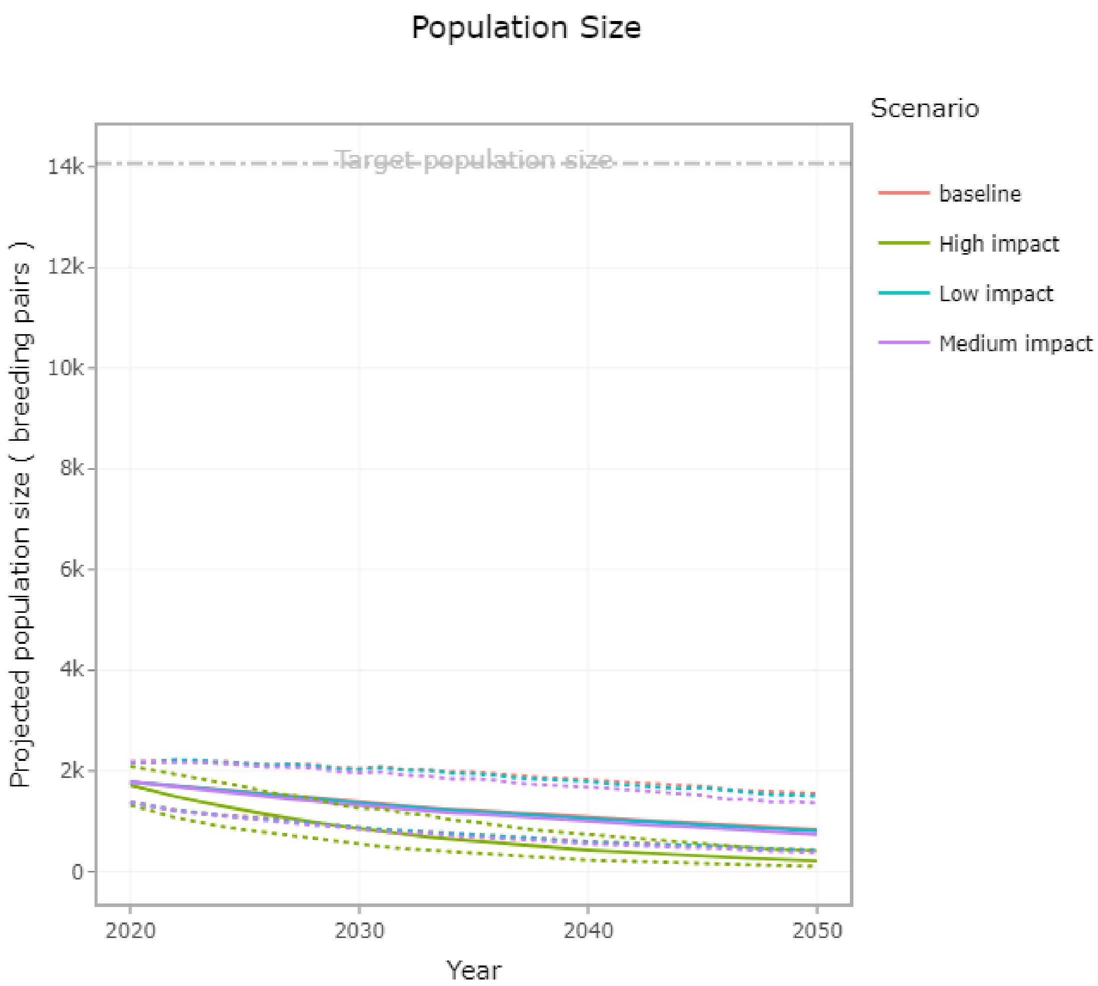


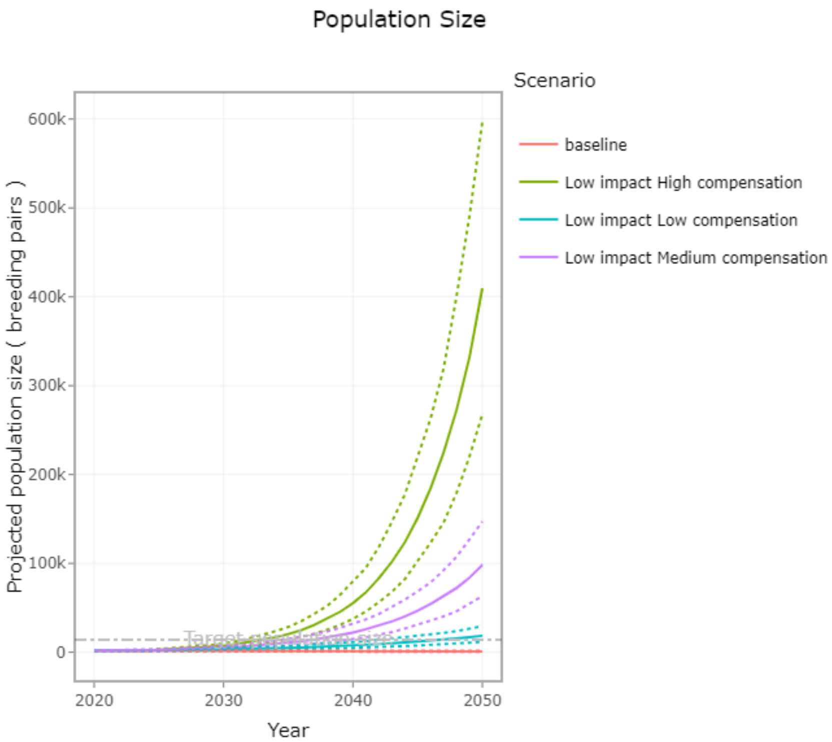

**Figure 59 Population projections for the baseline and low, medium and high impact scenarios.**

The starting population for all scenarios was well below the target population size. The baseline population projection, and the low, medium and high impact scenarios, remained below the target population size, as did the upper 95% confidence interval of the mean. The counterfactuals of both population size and growth rate from the projected population in 2050 showed smaller population sizes and lower growth rates for all three scenarios compared to the baseline (Table 148). The low impact scenario was the only one where the difference could be considered small. For both the medium and high impact scenarios the effects on the population were relatively large and important.

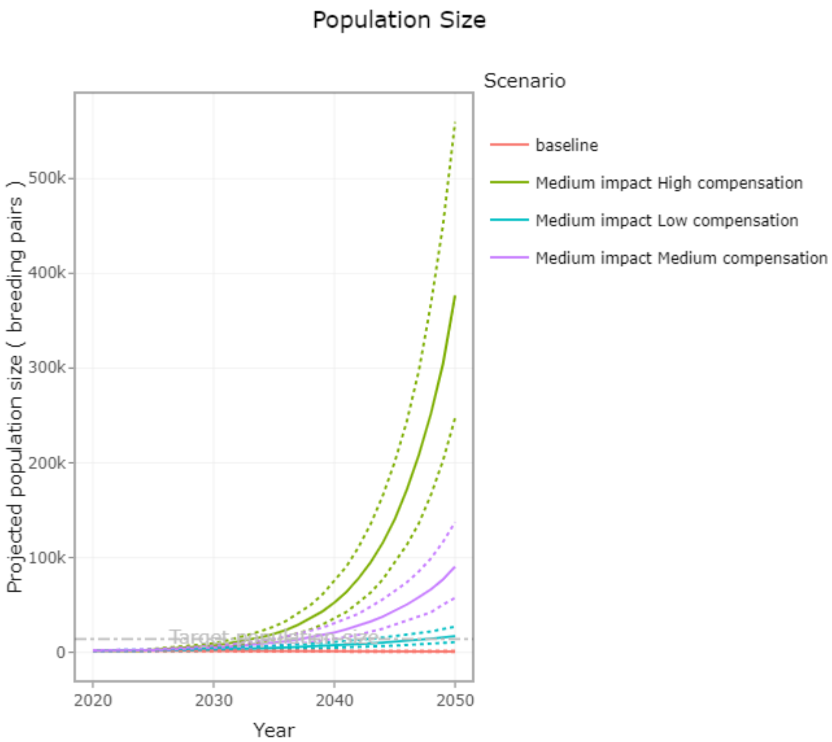
**Table 148 Counterfactual of population size and growth rate metrics for the baseline population projection in 2050 compared with low, medium and high impact scenarios.**

| Impact scenario | CPS (median) | CPS (mean ± 95% CI) | CGR (median) | CGR (mean ± 95% CI) |
|---|---|---|---|---|
| Low | 0.9621 | 0.9655 (0.8629 - 1.0757) | 0.9988 | 0.9988 (0.9952 - 1.0021) |
| Medium | 0.8836 | 0.8865 (0.7938 - 1.0031) | 0.9960 | 0.9961 (0.9926 - 0.9997) |
| High | 0.2576 | 0.2578 (0.2174 - 0.2987) | 0.9572 | 0.957 1 (0.9521 - 0.9618) |

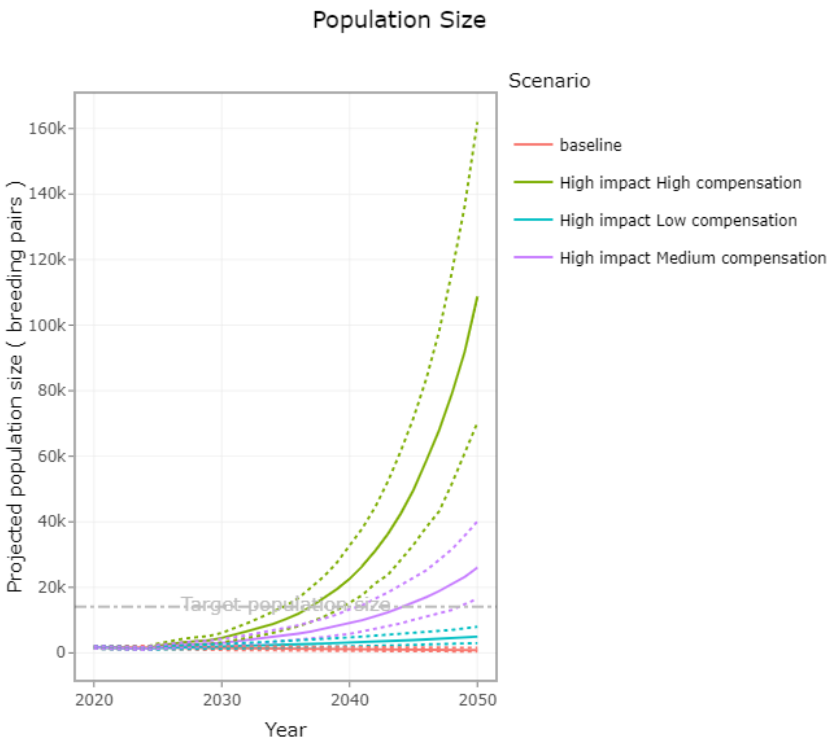
Projected population change of the baseline population (no impact) was compared with the low, medium and high impact scenarios combined with the three potential levels of compensation on productivity (Figure 60). In all combinations of impact and compensation scenarios the projected population sizes increased, including the low impact, high compensation scenario. Even in the high impact scenarios the compensation measures were projected to result in increasing population sizes.




Low impact



Medium impact




High impact

**Figure 60 Projected population size of breeding lesser black-backed gulls (pairs) at AOE SPA comparing baseline with the low, medium and high impact scenarios combined with low, medium and high compensation scenarios.**

In all impact and compensation scenario combinations the projected population growth rate was greater than one (Table 149). The baseline projected median growth rate was less than one. The population size increases shown are likely to be unrealistic, as the models were assumed to be density independent. However, PVA model results are best interpreted as relative differences rather than as absolute predictions of a likely future condition. Consequently, it is apparent that if exclusion of predators achieved any of the levels of increase in productivity simulated, the predator management actions are predicted to have sufficient benefit to cause the population to grow intrinsically.

**Table 149 Median annual growth rate of the projected populations for each combination of impact scenario and compensation scenario. Scenarios with a higher growth rate than the baseline are shaded grey.**

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Baseline (no impact) | Baseline (no compensation) | 0.9750 |
| Low | Low | 1.0778 |
| Low | Medium | 1.1370 |
| Low | High | 1.1909 |
| Medium | Low | 1.0750 |
| Medium | Medium | 1.1341 |

| Impact | Compensation | Median annual growth rate |
|---|---|---|
| Medium | High | 1.1877 |
| High | Low | 1.0324 |
| High | Medium | 1.0896 |
| High | High | 1.1410 |

In all but one impact and compensation scenario the target population size was exceeded within the time span of the population projection (30 years; Table 150). It was only for the high impact and low compensation scenario where the target population size was not achieved within 30 years. The PVA showed that the population was more sensitive to the scale of change in the impact from medium to high than the scale of the compensation modelled.

**Table 150 Year in which the projected median population size exceeded the target population size for each combination of impact and compensation scenario.**

| Impact | Compensation | Year target population size exceeded |
|---|---|---|
| Low | Low | 2047 |
| Low | Medium | 2037 |
| Low | High | 2033 |
| Medium | Low | 2048 |
| Medium | Medium | 2038 |
| Medium | High | 2034 |
| High | Low | Not achieved |
| High | Medium | 2037 |
| High | High | 2045 |

Examination of the counterfactuals of population size showed relatively little difference in the CPS between the low and medium impact scenarios within each compensation level (Figure 61). The CPS under low and medium impacts were much greater than under high impacts, across all three compensation levels. There was less difference between the low and medium compensation scenarios than between the medium and high compensation scenarios for the low and medium impact levels. Across the high impact scenarios, the benefits of compensation were much less, but note that the CPS values were all very large.

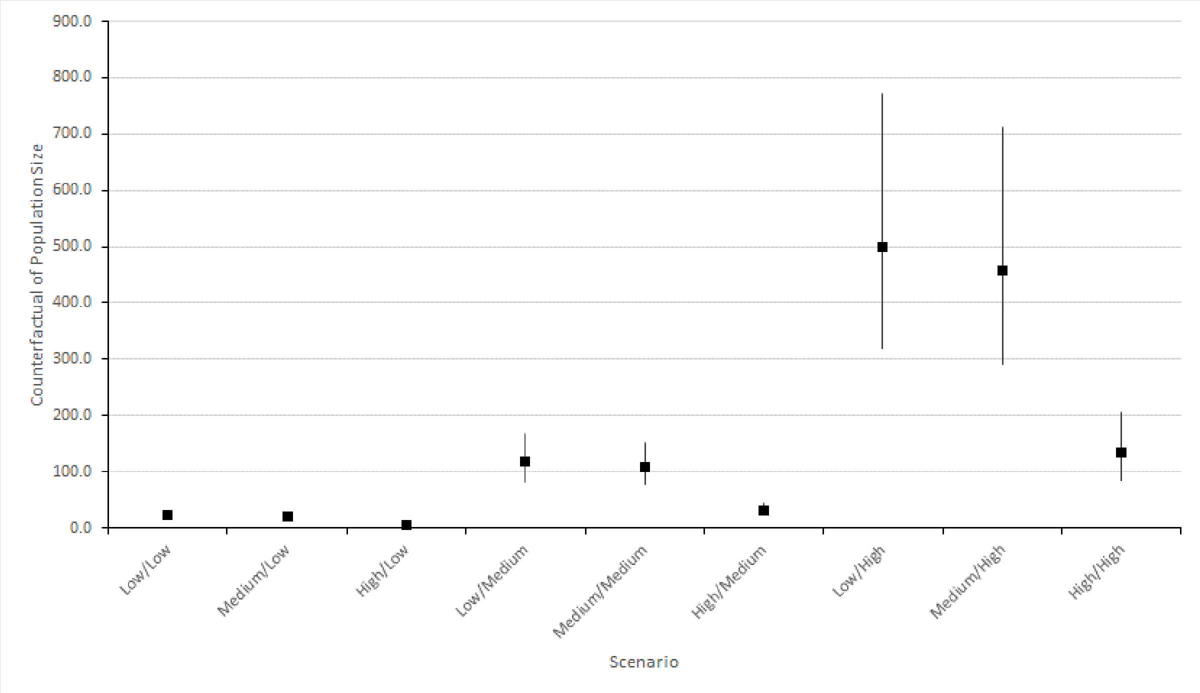



**Figure 61 Comparison of the mean CPS (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

Comparison of the CGR also showed relatively little difference between the low and medium impact scenarios across all three compensation levels (Figure 62). There was however a smaller difference between the CGR values for each level of compensation. The larger increase in the CPS values between the medium and high compensation values than the low and medium compensation scenarios was not replicated in the CGR values. Across the high impact scenarios, the benefits of compensation were apparently much less, with the high compensation level needed to bring the CGR value to the same level as the low compensation values for the low and medium impact scenarios. However, in all cases the CGR was greater than one, showing positive population consequences across all scenarios.

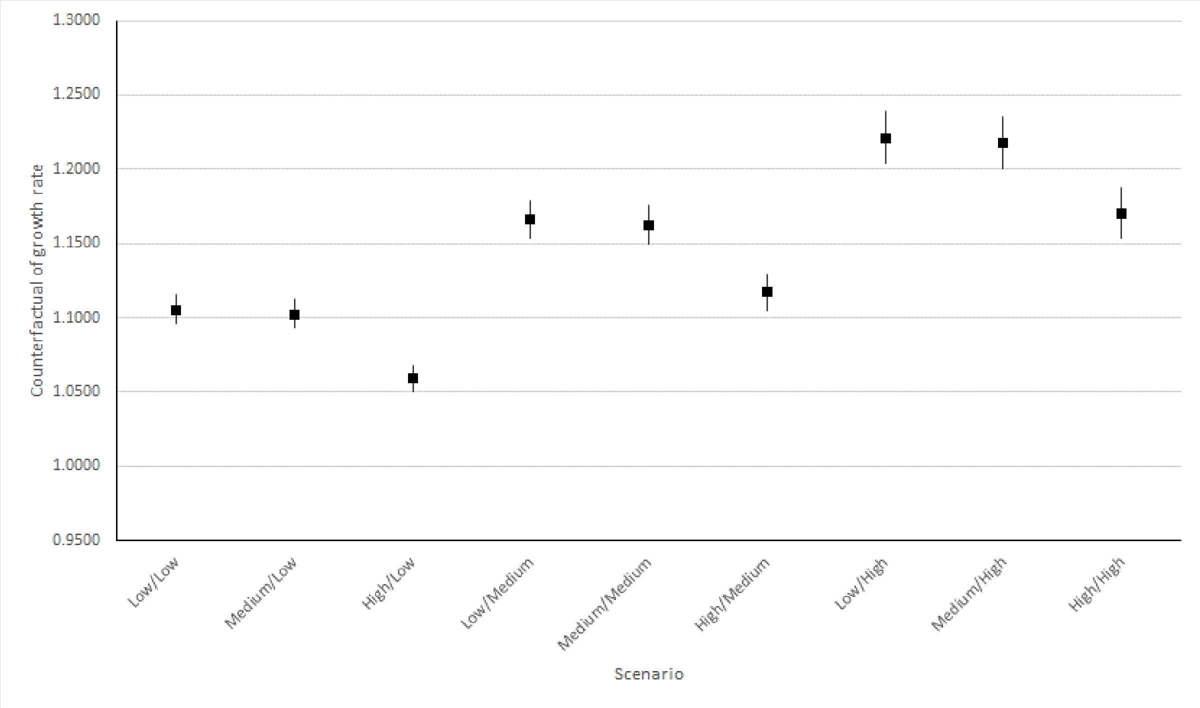



**Figure 62 Comparison of the mean CGR (and 95% confidence interval) between the tested scenarios. Each scenario is labelled as impact/compensations (e.g. the low impact with a high compensation scenario is low/high).**

These results suggest that low productivity is an important constraint on the lesser black-backed gull population at AOE SPA. Even low levels of compensation (a 30% increase in productivity from 0.33 to 0.43 chicks per pair) resulted in population growth, even for the high impact scenario (an increase of 33% to baseline adult mortality). It is apparent that predators, particularly foxes, are an important limitation to population growth at the AOE SPA. It is therefore not surprising that reducing this constraint would likely cause the population to grow, even if that growth was only through natal recruitment. It is likely that if predator control measures were successful there would also be growth in the population through immigration, which was not assessed here. There are several important caveats to consider when assessing the efficacy of the proposed compensation measures here. Firstly, the population would likely only grow to the limits of the area fenced within the nesting habitat, assuming no further predator control occurred (such as lethal control of foxes in the general area). The population model used here allows the population to grow unchecked, which is unrealistic. However, this growth to the limits of the protected habitat should allow an upper limit of population size to managed, assuming levels of fox predation and disturbance outside the fenced area remain as they are now. Secondly, the increases in productivity modelled here are only likely to be possible where there are sufficient food resources to support the colony growth and maintenance. However, lesser black-backed gulls have a broad diet, so supplementary feeding is likely to be a useful adaptive management action to ensure longer term population growth and then stability if food resources prove to be another limiting factor to productivity. The much higher levels of productivity of lesser black-backed gulls breeding in urban areas (Ross-Smith et al. 2014) shows that the combination of low predation risk to eggs and chicks combined with plentiful human food resources can occur, lending weight to the confidence in the success of these measures.

#### 8.7.1.1 *Compensation ratios*

The levels of impact that compensation measures would need to overcome were calculated for 1:3 and 1:6 ratios (Table 151). The 1:1 ratio impacts were tested above and all impact scenarios were considered sufficiently small to be compensated for. The high impact scenario at 1:3 and 1:6 was much larger than the high impact ratio at 1:1. These were close to or exceeded 100% of the current mortality level and so could not be compensated. The low and medium impact scenarios at both 1:3 and 1:6 ratios were all well below the high impact scenario at 1:1, so these scenarios could be compensated by the exclusion of foxes from the colony. Consequently, further PVA runs were unnecessary as the impacts were either well within the high impact scenario, or sufficiently larger than the high impact scenario that it is clear that the impact could not be compensated for.

**Table 151 Low, medium, and high impact scenarios at 1:3 and 1:6 ratios.**

| Impact scenario | Ratio | Low | Medium | High |
|---|---|---|---|---|
| Additional mortality (birds) | 1:3 | 14 | 42 | 453 |
| Additional mortality (rate) | | 3.0% | 9.1% | 98.5% |
| Additional mortality (birds) | 1:6 | 28 | 84 | 906 |
| Additional mortality (rate) | | 6.0% | 18.3% | 197.0% |

### 8.7.2 EAA 4

There are no other SPAs designated for their breeding lesser black-backed gull populations in EAA 4. Compensation beyond the AOE SPA but within the existing SPA network would therefore have to be elsewhere in the UK.

### 8.7.3 All other EAAs in the UK

There are nine further SPAs designated for their breeding lesser black-backed gull populations in the remaining EAAs in the UK. Five of these are on offshore islands where it would be reasonable to assume that fox predation is not exerting a pressure on the colonies. Of the remaining four SPAs, one is in Northern Ireland (Lough Neagh and Lough Beg) and the other three are in north-west England. Of these three there is ongoing work to protect the colony at South Walney (Morecambe Bay and Duddon Estuary). The colony at Bowland Fells SPA has decreased from 18,518 pairs during Seabird 2000 to 14,627 pairs in 2018[4]. This is an upland, mainland, colony of gulls and may be susceptible to fox predation. However, the surrounding areas are largely managed grouse moor, so fox populations may already be heavily managed. The colony in the Ribble and Alt Estuary SPA increased by 69% from 4,150 AON in 1998 to 7,022 in 2016, so predator exclusion may not be successful at increasing the population.

[4] https://jncc.gov.uk/our-work/lesser-black-backed-gull-larus-fuscus/

Should compensation at existing SPAs prove to be unlikely to be effective, several colonies that have been impacted by foxes, but not in SPAs, were identified by Davis et al. (2018). Fox exclosures may be suitable at one or more of these sites. Further research would be needed to confirm this.

### 8.8 Assessment of confidence

Using the methods outlined in Section 1.6 the confidence in the assessment of efficacy of the recommended compensation measure was undertaken for the control of predators compensation approach. The summary table for the predator control approach to compensation is given in Table 152 and the PVA assessment is shown in Table 153. The narrative describing and justifying the values given to the evidence and applicability metrics are described in Table 154 (predator control method) and Table 155 (PVA assessment).

**Table 152 Assessment of confidence in the predator control method of compensation.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Fox predation important at AOE SPA | n/a | Mavor et al. 2001 | Robust | Robust | Medium | Robust | ROBUST | HIGH | VERY HIGH |
| Predator proof fences exclude predators | n/a | Young et al. 2012, Furness et al. 2013, RSPB 2020, Xcluder 2020 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Excluding foxes can increase productivity of lesser black-backed colonies | n/a | Davis et al. 2018 | Robust | Robust | Robust | Robust | ROBUST | MEDIUM | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 153 Assessment of confidence in the PVA comparing impact scenarios to baseline conditions.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age at first breeding | 5 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Upper constraint on productivity | 3 | PVA app default | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Initial population size | 2,000 in 2015 | JNCC (2014) | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Productivity rate per pair | mean: 0.33, sd: 0.1756 | NE Draft Supplementary advice on conserving and restoring site features | Robust | n/a | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Adult survival rate | mean: 0.885, sd: 0.056 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Age class 0 to 1 | mean: 0.82, sd: 0.056 (SD assumed same as other age clases) | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 1 to 2 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 2 to 3 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Age class 3 to 4 | mean: 0.8283, sd: 0.077 | PVA app "National" default value | Robust | n/a | Robust | Robust | ROBUST | MEDIUM | HIGH |
| Impact on productivity rate | Low compensation = -0.29 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | Medium compensation = -0.56 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | High compensation = -1.07 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

**Table 154 Narratives justifying the evidence and applicability scores for metrics used in confidence in the predator control method of compensation.**

| Metric | Narrative |
|---|---|
| Fox predation important at AOE SPA | The evidence from Mavor et al. (2001) was considered robust evidence from national monitoring. Applicability was high, as the evidence was directly from the AOE SPA. As such overall confidence was very high. |
| Predator proof fences exclude predators | It is clear that there is robust evidence for the use of predator proof fencing to exclude the predators that affect lesser black-backed gulls, particularly foxes. None of the evidence was from the SPA of interest, so applicability was medium. Overall confidence was therefore high. |
| Excluding foxes can increase productivity of lesser black-backed colonies | Research from the lesser black-backed gull colony at South Walney by Davis et al. (2018) was assessed as robust but applicability was medium, as the evidence was not from AOE SPA. Confidence was high. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was very high or high, so an overall score of high was given. |

**Table 155 Narratives justifying the evidence and applicability scores for metrics used in the assessment of the potential compensation measures from the exclusion of predators.**

| Metric | Narrative |
|---|---|
| Age at first breeding | The age at first breeding of lesser black-backed gull is well established and not in question. It is not variable between populations and is directly applicable to the AOE SPA population. |
| Upper constraint on productivity | Lesser black-backed gulls have a modal clutch size of 3 (c. 75%) with clutches above this being very rare (Bolton 1991) so productivity is extremely likely to be below three chicks per pair. |
| Initial population size | The count used was based on a standardised and approved method. There is a long time series of counts from the colonies at AOE SPA with data reported in the SMP database. |
| Productivity rate per pair | The productivity rate was based on NE Draft Supplementary advice on conserving and restoring site features of the AOE SPA. These were robust data from the site and were assessed as having robust data and high applicability so were given a very high confidence score. |
| Adult survival rate | The adult survival rate evidence is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. A medium score was therefore given. With robust evidence and medium applicability, the confidence score was high. |
| Age specific survival rates from 0 to 4 years | The adult survival rate evidence is based on high quality analyses of robustly collected data. These data have been agreed for application in the Seabird PVA Tool. Since these data are robust for all evidence score the overall evidence score is robust. However, the applicability |

| Metric | Narrative |
|---|---|
| | of these data to the population being modelled is unknown. They are being used in the absence of colony specific data. A medium score was therefore given. With robust evidence and medium applicability, the confidence score was high. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the evidence was robust and the applicability was high or medium, so an overall score of high was given. |

### 8.8.1 Confidence in predator control

With an overall assessment of high in the compensation approach and in the PVA assessment method the assessment of confidence in the proposed predator control compensation against the three impact scenarios needs to be carefully considered. The population level assessment (8.7) showed that all impact scenarios could be compensated for by the low, medium, and high compensation scenarios. There was little difference between the low and medium impact scenarios for all three compensation scenarios and PVA metrics were all very positive for these combinations so, confidence in these was high (Table 156). While the PVA metrics for the high impact scenario for all three compensation scenarios were notably lower than the low or medium impact scenarios it was clear that all the compensation scenarios were sufficient. Consequently, the confidence in these was high as well (Table 156).

**Table 156 Assessment of confidence in the impact/compensation scenarios for lesser black-backed gulls from predator control measures increasing productivity.**

| | Low impact | Medium impact | High impact |
|---|---|---|---|
| Low compensation | HIGH | HIGH | HIGH |
| Medium compensation | HIGH | HIGH | HIGH |
| High compensation | HIGH | HIGH | HIGH |

## 8.9 Future monitoring and adaptive management

### 8.9.1 Fencing out terrestrial predators

While evidence supports foxes having important effects on the AOE SPA lesser black-backed gull population size, a decision on whether other predators may be affecting the colonies may be needed to determine a suitable control measure. Foxes alone appear to be readily controlled through suitable electric fencing and night-time patrols by wardens during the breeding season, though this can cause disturbance to the colony. If other predators, such as rats, are present then different methods of exclusion would be necessary. Thus, it is recommended that, unless a decision is taken to construct predator-proof fencing in the first instance, further monitoring of the presence of terrestrial mammal predators is undertaken to determine which species are present both within and around the colonies. This could use a variety of methods (camera traps, live traps, kill traps, DNA traps).

If permanent anti-predator fencing is used it will be necessary to remove any predators from within the fenced area and demonstrate that they are absent. This would best be achieved in the winter following completion of the fencing, when numbers should be at their lowest. Even after predators have been removed from fenced areas it will be important to maintain some form of monitoring to enable action to remove any predators that enter the exclosure.

Monitoring should include counts of total colony size, including any nests outside exclosures, and plot counts of productivity. Careful further monitoring of at least some nests may be helpful to determine whether other potential negative effects on productivity are occurring (e.g. food provisioning of chicks, disturbance of nests by other predators or human actions) to provide information for further potential management actions. For instance, if productivity doesn't increase, despite excluding terrestrial predators, it would be important to know the reasons for the constraint. Monitoring of the colony size and productivity should be annual, at least initially. If the compensation approach is proved to be successful from initial monitoring, then monitoring could be reduced in frequency or intensity. Monitoring should be continued throughout the period that compensation is required.

If the only predators that need to be excluded are foxes, then seasonal electric fencing and wardens undertaking regular night-time checks may suffice. Electric fencing typically needs checking on a daily basis to ensure that it is fully functional and has not been breached. Common problems with electric fencing include vegetation growth shorting out parts of fencing, wind damage and localised flooding causing shorts. Because of these issues fencing needs daily inspection and action to maintain its efficacy. These management actions all increase disturbance to the colony, so permanent fencing may be preferable.

For any type of exclosure fencing there will need continual monitoring of the condition of the fencing, at least immediately before and through the breeding season. There will need to be budget and suitable staff or contractors available to undertake both regular maintenance and emergency repairs throughout the compensation period.

#### 8.9.2 Future research

Recovery of the lesser black-backed gull colony at AOE SPA would require access to food resources to maintain the colony. In order to avoid future management issues between different SPA qualifying features future research on the resources needed to maintain a colony of 14,000 pairs of birds compared with estimate of current and future resource availability would be very useful.

### 8.10 Summary

The review considered each of the main forms of compensation recommended by Furness et al. (2013) that could be deployed for lesser black-backed gulls generally: mink eradication at lesser black-backed gull colonies, fencing out foxes from colonies, end culling of lesser black-backed gulls, closure of sandeel and sprat fisheries and eradicate rats at lesser black-backed gull colonies. The use of closure of sandeel and sprat fisheries was excluded because this species does not rely heavily on sandeels but takes a diverse diet with much food derived from terrestrial rather than marine sources. In addition, changes to the licensed culling of lesser black-backed gulls resulted in the conclusion that that this was no longer a suitable compensation measure. Since Furness et al. (2013) little new evidence was found on the benefits of the recommended compensation measures, though nothing was found that would contradict their findings.

PVA suggested that the AOE SPA population of lesser black-backed gulls should be decreasing and that this decrease would remain likely with all impact levels. The two colonies of lesser black-backed gulls at AOE SPA have undergone changes at different scales, but overall, the population size is much lower than in the past and much lower than the citation population size. Three levels of compensation measure were assessed against the three levels of impact. The potential increase in productivity from predator exclusion from colonies in the SPA were tested using the PVA. Results of the PVA suggested that population increase (from a decreasing baseline) was likely for all compensation levels combined with all impact levels tested.

Confidence in the PVA assessment process was high. Confidence was high for compensating all impact scenarios where predators were excluded from colonies on the basis that foxes are currently a key constraint on productivity at the AOE SPA, so successfully excluding them would with high certainty increase productivity, and the three levels of increased productivity modelled are entirely plausible results of such exclusion.

# 9 OUTER THAMES ESTUARY SPA – NON-BREEDING RED-THROATED DIVER

The Outer Thames Estuary SPA (OTE SPA) is in south-east England in the southern North Sea. The site is large and in three sections from the seas off east Norfolk in the north to the north coast of Kent. The habitat in the SPA includes shallow and deeper water, with high tidal current streams and a variety of mobile sediments. The presence of shallow sandbanks is particularly important for red-throated diver. A large proportion of the SPA is less than 20 m deep, with some areas 20 – 50 m deep, particularly towards the offshore boundary of the SPA.

## 9.1 Conservation status of red-throated diver

The biogeographic population was estimated at 300,000 individuals, of which 21,500 are estimated to winter in Great Britain (Woodward et al. 2020) and 2,000 individuals in all-Ireland (Crowe et al. 2008). The IUCN lists the global population of red-throated diver as "Least Concern".

The UK population was green listed in Birds of Conservation Concern (BOCC) 4 (Eaton et al. 2015) having been previously listed as 'amber' in BOCC3 and BOCC2.

It is protected under the Birds Directive as a migratory species. The SPAs in Great Britain were estimated to hold 44% of the Great Britain wintering population of red-throated divers present in 2008 (Stroud et al. 2016) an increase of 19% in SPAs estimated in the previous SPA review (Stroud et al. 2001).

## 9.2 Citation population size

The SPA was classified in 2010 based on an estimated 6,446 wintering individuals (1989 to 2006/07, peak mean estimate).

## 9.3 Conservation objectives

The site has conservation objectives, "to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site."

More detailed Supplementary Advice on Conservation Objectives (SACO) have since been added online, last updated 13 September 2019 (Natural England 2019). For red-throated diver at OTE SPA these are:

- Reduce the frequency, duration and/or intensity of disturbance affecting roosting, foraging, feeding, moulting and/or loafing birds so that they are not significantly disturbed;

- Maintain the size of the non-breeding population at a level which is at or above 18,079 individuals, whilst avoiding deterioration from its current level as indicated by the latest mean peak count or equivalent;
- Maintain concentrations and deposition of air pollutants at below the site-relevant Critical Load or Level values given for this feature of the site on the Air Pollution Information System (www.apis.ac.uk);
- Maintain the structure, function and supporting processes associated with the feature and its supporting habitat through management or other measures (whether within and/or outside the site boundary as appropriate) and ensure these measures are not being undermined or compromised;
- Maintain the extent, distribution and availability of suitable habitat (either within or outside the site boundary) which supports the feature for all necessary stages of the non-breeding/wintering period (moulting, roosting, loafing, feeding) at the following levels: Subtidal sand (220,295.55); Subtidal coarse sediment (73,606.64); Subtidal mixed sediments (62,100.63 ha); Subtidal mud (12,549.14 ha); Circalittoral rock (335.2 ha); and water column;
- Maintain the distribution, abundance and availability of key food and prey items (e.g. fish) at preferred sizes;
- Maintain the depth of inshore waters currently used as feeding or moulting sites;
- Reduce aqueous contaminants to levels equating to High Status according to Annex VIII and Good Status according to Annex X of the Water Framework Directive, avoiding deterioration from existing levels;
- Maintain the dissolved oxygen (DO) concentration at levels equating to High Ecological Status (specifically ≥ 5.7 mg per litre (at 35 salinity) for 95 % of the year), avoiding deterioration from existing levels;
- Maintain water quality at mean winter dissolved inorganic nitrogen levels where biological indicators of eutrophication (opportunistic macroalgal and phytoplankton blooms) do not affect the integrity of the site and features, avoiding deterioration from existing levels; and
- Maintain natural levels of turbidity (e.g. concentrations of suspended sediment, plankton and other material) across the habitat.

A Site Improvement Plan (SIP) for OTE SPA was published in May 2015 (Natural England 2015b). That identified commercial marine and estuarine fisheries as a threat to red-throated divers and identified introducing and enforcing appropriate management as necessary as a responsibility of Defra, Eastern Inshore Fisheries Conservation Authority (IFCA), Kent and Essex Inshore Fisheries Conservation Authority (IFCA), and the Marine Management Organisation (MMO). However, the issues and actions table includes the statement, "*Entanglement in static fishing nets is an important cause of death for red-throated divers in the UK waters. Netting is widespread across the sandbanks but is seasonal and occurs primarily when the Red-throated diver population is not at its peak. The scale of by-catch within the site has been assessed by the Kent & Essex IFCA and was not found to be problematic and so can be deemed to be low-risk.*"

#### 9.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of the qualifying features within the site.

There are two main sources of impact on red-throated divers from offshore wind farm development: displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The COs to maintain: the extent, distribution (and availability) of supporting habitat; the structure and function of the habitat, the supporting processes on which the habitats of the qualifying features rely and the distribution of the qualifying feature within the site could all be affected through the displacement of red-throated divers from the wind farm and its surroundings, if birds from the SPA used this area prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of red-throated divers will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival which could in turn impact the population of that qualifying feature.

The maintenance of the population of each of the qualifying features could be affected indirectly through impact to energy budgets from displacement and barrier effects.

### 9.4 Location of compensation

Furness et al. (2015) defined five BDMPS spatial areas for red-throated divers in UK waters in winter. The OTE SPA occurs within the “SW North Sea” area. Consequently, the hierarchy of the locations of compensation are:

1. OTE SPA;
2. SW North Sea; and
3. All other BDMPS spatial areas in the UK.

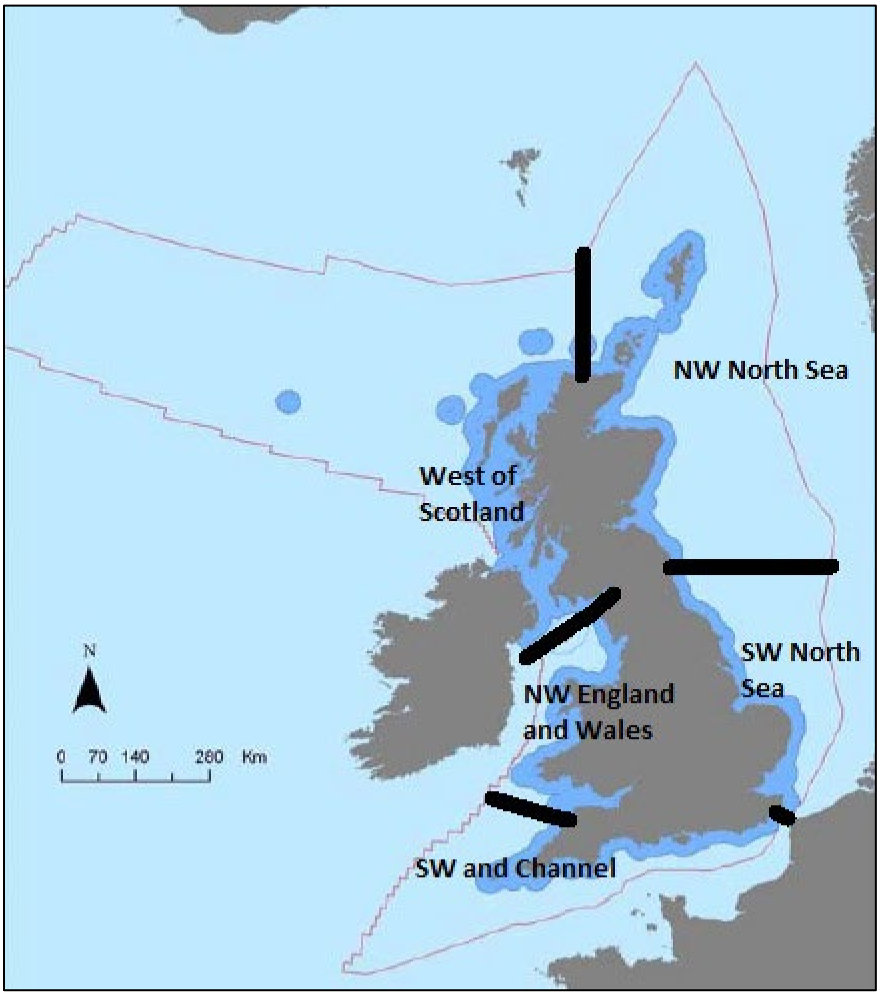



**Figure 63 Defined BDMPS spatial areas for red-throated divers in UK waters in winter from Furness et al. (2015). Limits of UK waters are shown by red line. BDMPS spatial areas extend from the UK coast to the red limit, bounded by the thick black lines marking the sides of each BDMPS area.**

## 9.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for wintering red-throated diver were developed based on the five potential compensation measures reviewed by Furness et al. (2013). The five potential measures listed were:

- Provision of nesting rafts at breeding lochs;
- Closure of sandeel and sprat fisheries close to wintering areas;
- Closure of sandeel and sprat fisheries close to breeding areas;
- Prevention of oil spills; and
- Reducing disturbance by vessel activity.

Of these five potential measures the only one that may be of direct value to compensating impacts on red-throated divers at OTE SPA was reducing disturbance by vessel activity. However, further review discovered that an existing project in the Netherlands created strict marine reserves within the Voordelta SPA in order to provide compensation measures. Red-throated diver is one of the features of the SPA.

### 9.5.1 Creating a marine reserve

The predicted impacts of many human activities that occur in or near the OTE SPA on the winter population of red-throated divers in the SPA have been found only to compromise the conservation objective to maintain "*the distribution of qualifying features within the site*". It appears that the population size within the SPA has been, at least, maintained since designation. Therefore, compensation measures need to focus primarily on enhancing the availability, extent, distribution and quality of habitat supporting red-throated divers within the OTE SPA and the wider UK SPA network, rather than increasing the size of the population within the OTE SPA or in the SPA network. As such, in addition to the measures identified by Furness et al. (2013) it was important to consider other measures to improve the quality of the habitat within the UK SPA network.

The negative effect of displacement from offshore wind farms and other disturbing activities e.g. those involving vessel movements on wintering red-throated diver is indirect loss of habitat that would otherwise be available. The preferred habitat of wintering red-throated diver is shallow sandbanks 10 – 30 m deep, which is also the habitat that offshore wind farms have been placed on in the OTE SPA. Therefore, a suitable compensation measure would be to improve the availability and/or quality of the remaining shallow sandbanks in the SPA for wintering red-throated divers. This could be achieved through a reduction in disturbance by vessel activity within those habitats and protection from damage from, for example, bottom trawled fishing gear or aggregate extraction.

A similar situation has previously occurred in the Voordelta SPA. Expansion of the Port of Rotterdam would remove 3,125 hectares of offshore sandbanks, among other habitats, from the SPA. The SPA was designated for, among other species, its population of 500 – 4000 non-breeding red-throated divers. The agreed compensation measure for the SPA was to create a marine reserve that restricted activities in the SPA, including closing the area to fisheries, closure of one third of the reserve to all disturbance from boats and flights, including military flights[5]. These measures aimed to increase the quality and the availability of habitat for the impacted features rather than to compensate for the loss of area of habitat. The area lost within the SPA was estimated to be 2,455 ha, while the area of the marine reserve was 24,550 ha. This compensation ratio of 1:10 was based on finding that restricting the usage of an area "can generate at least 10 % improvement in soil [*sic*] quality due to biomass growth". It is assumed that "soil quality" actually refers to benthic habitat and has been mistranslated from the original Dutch report.

### 9.5.2 Removal of existing wind farms

The wind farms which have been constructed within the OTE SPA since the start of the bird survey programme that informed the size and shape of the SPA are the biggest additional anthropogenic interventions in the SPA since the early 2000s. They appear to be responsible for a significant change to the distribution of red-throated divers within the SPA. Therefore, removal of these wind farms may remove the most significant contributory factor to the pressures on the distribution of

[5] https://www.legislation.gov.uk/eudn/2008/914

birds within the SPA. However, in order to determine whether removal of the wind farms would have the desired effect it is important to understand the effect the wind farms have had on the distribution of birds within the site. Comparison of red-throated diver distribution between the period prior to construction of the wind farms and after construction of the wind farms is therefore necessary.

1. Was there a statistically significant change in the distribution of red-throated divers from the period before wind farms were constructed and after wind farms were operational?
2. Can any change in distribution of red-throated divers be attributed to the wind farms?

The key biological questions for compensation measures for red-throated divers at OTE SPA are provided in Table 157.

**Table 157 Key Biological Questions in assessing the potential for compensation of non-breeding red-throated divers at OTE SPA.**

| No. | Key Biological question |
|---|---|
| 1 | Is there evidence that reducing shipping activity would increase habitat availability for non-breeding red-throated divers in: |
| 1.1 | OTE SPA? |
| 1.2 | SW North Sea BDMPS area? |
| 1.3 | All other BDMPS spatial areas in the UK? |
| 2 | Is there evidence that creating a strict marine reserve would improve habitat for red-throated divers in: |
| 2.1 | OTE SPA? |
| 2.2 | SW North Sea BDMPS area? |
| 2.3 | All other BDMPS spatial areas in the UK? |
| 1 | Is there evidence of displacement of red-throated divers from offshore wind farms in: |
| 1.1 | OTE SPA? |
| 1.2 | SW North Sea BDMPS area? |
| 1.3 | All other BDMPS spatial areas in the UK? |
| 2 | Is there evidence that removal of offshore wind farms would provide additional habitat to red-throated divers in: |
| 2.1 | OTE SPA? |
| 2.2 | SW North Sea BDMPS area? |
| 2.3 | All other BDMPS spatial areas in the UK? |

## 9.6 Review of potential compensation measures

### 9.6.1 Reducing disturbance by vessel activity

More recent research has better quantified the effect of ship traffic on non-breeding red-throated divers (APEM 2016; Burt et al 2017; Jarrett et al. 2018, Burger et al. 2019, Mendel et al. 2019). Management that could reduce vessel activity during winter in areas used by large numbers of non-breeding red-throated divers could reduce the disturbance to this species. That would most likely reduce energy expenditure as well as allowing birds to spend more time foraging, so would be highly likely to improve overwinter survival and body condition. While it may be difficult to reduce ship traffic, there may be options to limit that to smaller clearly defined shipping lanes, and to set speed limits as birds appear to be disturbed more by faster-moving vessels, and there may be some scope to shift ship traffic to times of year when red-throated divers are not aggregated in these areas (i.e. from winter to summer), or to move some forms of disturbance (such as recreational activity) from areas occupied by red-throated divers to other areas where these birds are not present in large numbers.

#### 9.6.1.1 *Answers to the key biological questions (9.5).*

The answers to the key biological question in relation to compensation through reducing disturbance by vessel activity are shown in Table 158.

**Table 158 Answers to Key Biological Questions in assessing the potential for compensation through reducing disturbance by vessel activity.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that reducing shipping activity would increase habitat availability for non-breeding red-throated divers in: | |
| 1.1 | OTE SPA? | Yes. There is evidence from digital aerial surveys and from modelling of red-throated diver spatial densities in the OTE SPA that they are avoiding shipping. |
| 1.2 | SW North Sea BDMPS area? | No. No evidence could be found from the only other SPA for non-breeding red-throated divers in the BDMPS area (Greater Wash SPA). However, it is likely that this occurs. |
| 1.3 | All other BDMPS spatial areas in the UK? | Yes. There is evidence from digital aerial surveys and from modelling of red-throated diver spatial densities in Liverpool Bay SPA that birds are avoiding shipping. |

### 9.6.2 Creating a marine reserve

The creation of the Voordelta marine reserve in the Netherlands strictly protected a large area of coastal sea to compensate for expansion of the Port of Rotterdam. In addition to creating this marine reserve with the compensation measures discussed in 9.5.1 three areas within the reserve with restricted access to vessels were created to provide undisturbed areas for common scoter. These areas totalled 5,173 ha.

Monitoring was undertaken on the benthos, birds, fish, physical processes, and human activities (van der Meulen 2016). At present monitoring results appear to be embargoed (e.g. see Borst et al. 2016) and reports are in Dutch. However, a complex integrated monitoring and management approach has been taken (Kinneging et al. 2017) and was proposed to continue for 30 years (Blake et al. 2020) so important results on the improvement in quality of the SPA to the qualifying features are likely to be available in the future.

#### 9.6.2.1 *Answers to the key biological questions (9.5).*

The answers to the key biological question in relation to compensation through creation of a marine reserve are shown in Table 159.

**Table 159 Answers to Key Biological Questions in assessing the potential for compensation through creation of a marine reserve.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2 | Is there evidence that creating a strict marine reserve would improve habitat for red-throated divers in: | |
| 2.1 | OTE SPA? | **Maybe.** There is no direct evidence of this for the OTE SPA, however there is direct evidence of the effects of shipping disturbance and offshore wind farms on the distribution of red-throated divers in this SPA. A marine reserve that protects suitable red-throated diver habitat from disturbance would reduce pressures on birds. |
| 2.2 | SW North Sea BDMPS area? | **Maybe.** There is no direct evidence of this for the Greater Wash SPA, however there is direct evidence of the effects of offshore wind farms on red-throated divers in this SPA. A marine reserve that protects suitable red-throated diver habitat from disturbance would reduce pressures on birds. |
| 2.3 | All other BDMPS spatial areas in the UK? | **Maybe.** There is no direct evidence of this for SPAs in other BDMPS areas, however there is direct evidence of the effects of shipping disturbance and offshore wind farms on red-throated divers in Liverpool Bay SPA. A marine reserve that protects suitable red-throated diver habitat from disturbance would reduce pressures on birds. |

### 9.6.3 Removal of existing wind farms

Removing wind farms from inside or close to the boundaries of SPAs would greatly reduce the observed or predicted impacts on the distribution of red-throated divers within these SPAs.

#### *9.6.3.1 Answers to the key biological questions (9.5).*

The answers to the key biological question in relation to compensation through removal of existing wind farms are shown in Table 160.

**Table 160 Answers to Key Biological Questions in assessing the potential for compensation through removal of existing wind farms.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence of displacement of red-throated divers from offshore wind farms in: | |
| 1.1 | OTE SPA? | **Yes.** There is strong evidence for displacement from offshore wind farms in the OTE SPA. Further evidence on the displacement distance and gradient of the effect around these wind farms within the OTE SPA is needed. |
| 1.2 | SW North Sea BDMPS area? | **Yes.** There is strong evidence for displacement from offshore wind farms in the Greater Wash SPA. Further evidence on the displacement distance and gradient of the effect around these wind farms within the Greater Wash SPA is needed. |
| 1.3 | All other BDMPS spatial areas in the UK? | **Yes.** There is strong evidence for displacement from offshore wind farms in Liverpool Bay SPA. Further evidence on the displacement distance and gradient of the effect around these wind farms within Liverpool Bay SPA is needed. Additionally, there is evidence of no effect of the Robin Rigg offshore wind farm on red-throated divers in the Solway Firth SPA. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2 | Is there evidence that removal of offshor wind farms would provide additional habitat to red-throated divers in: | |
| 2.1 | OTE SPA? | **No**. There is no evidence from OTE SPA, or anywhere else, that red-throated divers will return to the areas currently occupied by offshore wind farms following decommissioning. |
| 2.2 | SW North Sea BDMPS area? | **No**. See the answer to question 2.1 |
| 2.3 | All other BDMPS spatial areas in the UK? | **No**. See the answer to question 2.1 |

## 9.7 Population level assessment

### 9.7.1 OTE SPA

#### 9.7.1.1 *Reducing disturbance by vessel activity*

Red-throated divers are known to avoid ships (Mendel et al. 2019, Schwemmer et al. 2011) and the analysis for the East Anglia 1 North and East Anglia 2 windfarm impact assessment[6] also found avoidance of shipping in the OTE SPA. However, analysis of Automatic Identification System data by the MMO showed that the majority of the shipping in the Thames Estuary was within shipping lanes (Figure 64) and occurred in summer (MMO 2014); consequently, measures to move vessels in to shipping lanes or seasonal restrictions may be less effective in reducing disturbance than might be assumed.

[6] https://infrastructure.planninginspectorate.gov.uk/wp-content/ipc/uploads/projects/EN010078/EN010078-004480-ExA.AS-10.D8.V4%20EA1N&EA2%20Displacement%20of%20red-throated%20divers%20in%20the%20Outer%20Thames%20Estuary.pdf

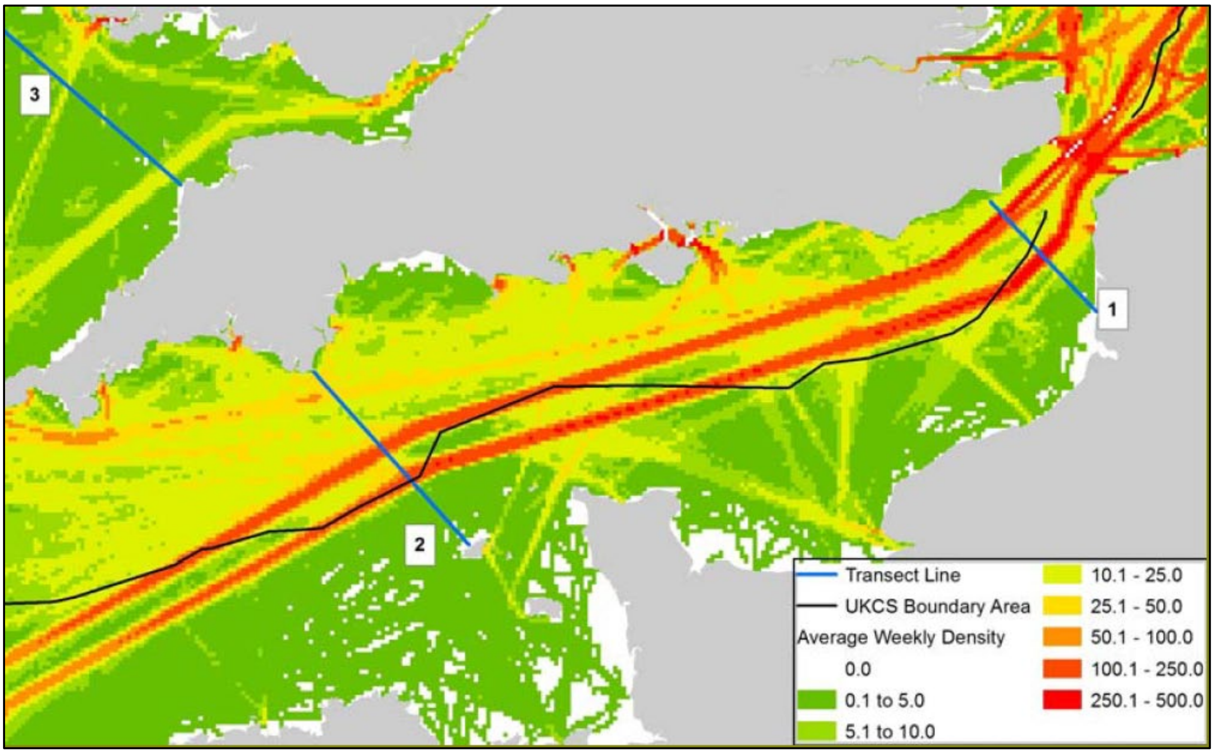



**Figure 64 South Coast vessel density and routes from AIS data in 2012 (from MMO 2014).**

The analysis of red-throated diver distributions in the OTE SPA for EA1N and EA2 offshore wind farm assessment showed there was strong avoidance of high levels of shipping density (Figure 65). This is also apparent in the analysis of red-throated diver distribution by Irwin et al. (2019).

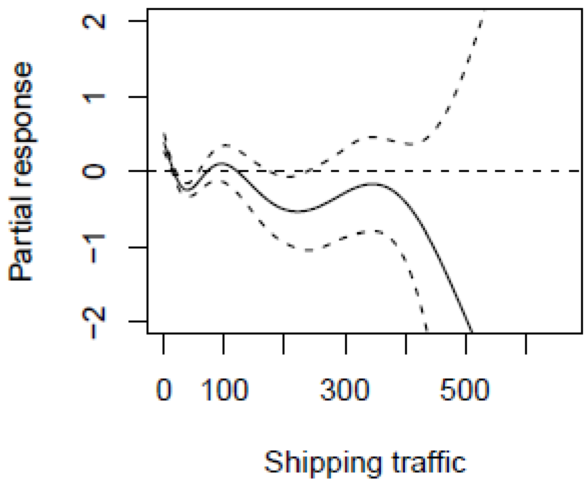
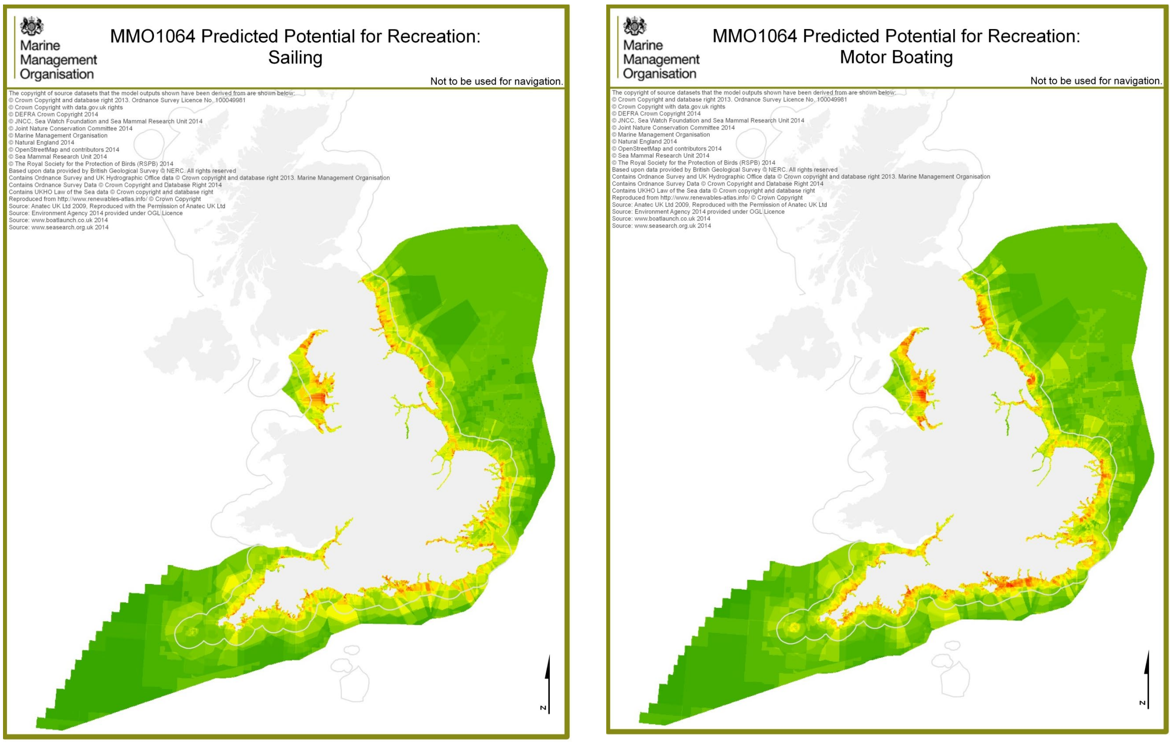



**Figure 65 Partial plots of smooth covariates of red-throated divers in the OTE SPA in relation to shipping (see EA1N/EA2 report cited above).**

However, AIS data are only collected from vessels more than 15 m in length. Recreational vessels are not required to used AIS transmitters, and most do not. The MMO have identified the potential for recreational sailing and motor boating in English waters, including the outer Thames estuary

(Figure 66). This indicates high levels of potential for recreational sailing and motor-boating close to the coast of the OTE SPA but low potential across large areas of the SPA away from the coast which appears to relate to the shallower sandbanks.




Recreational sailing | Recreational motor boating

**Figure 66 Sailing (left) and motor boating (right) activity map from MMO (2014).**

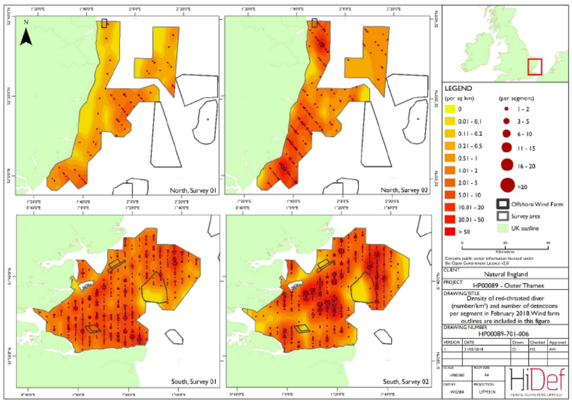
It is apparent that there are offshore areas within the OTE SPA that are not predicted to be suitable for recreational boating, though presumably these are across the areas of shallow sandbanks. It is apparent that there is reluctance among many recreational boaters to use much of the area across the OTE SPA for this reason. However, it is apparent that there are coastal areas within the SPA with a high potential for both recreational sailing and motor boating. Despite this, recreational boating is highly seasonal with a large majority of activity occurring in the summer months (MMO 2014), while red-throated divers occur mostly from October to March. Therefore, reducing recreational boating in the OTE SPA in winter is unlikely to result in important changes in disturbance. However, a combination of strict no boating policy within protected areas combined with stakeholder engagement and education could be a beneficial element to a marine reserve approach to compensation.

#### 9.7.1.2 *Creating a marine reserve*

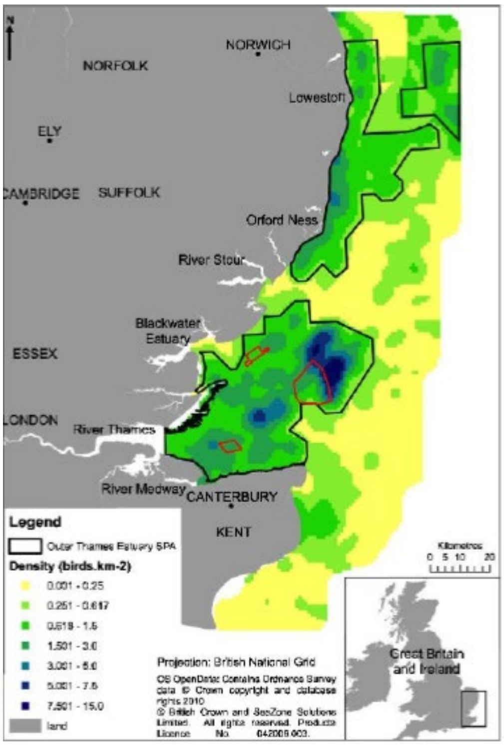
Since the impacts on the red-throated diver population in the OTE SPA are apparently not affecting the population size within the SPA, the assessment of compensation measures in maintaining coherence of the UK SPA network is focused on improving the availability/quality of the habitat within the network.

Recent analyses of the spatial distribution of red-throated divers in the OTE SPA has been undertaken for the assessment of East Anglia One North and East Anglia 2 offshore wind farms

(PINS document reference ExA.AS-4.D3.V1). The analysis found a preference for water depths below 20 m, which was similar to that found in the German Bight of 10 – 30 m (Vilela, et al. 2020). These analyses and subsequent commentary in PINS with Natural England have raised the question as to whether the level of displacement that may be occurring in the OTE SPA is different to that found in the German Bight, as it was found to be between 7 km (see EA1N/EA2 report cited above) and 12 km (APEM 2021) in the OTE SPA. However, comparison of the predicted distribution of red-throated divers in the OTE SPA between 2001/02 to 2006/07, prior to construction of wind farms in the SPA (O'Brien et al. 2012) and 2018 (Irwin et al. 2019) does strongly suggest that the pattern of change in distribution is response to wind farms is not simple (Figure 67). The most recent analyses by APEM (2021) has provided evidence consistent with there being a direct displacement effect following the construction of London Array OWF within the OTE SPA and that this influence may extend up to 12km from its boundary. However, this effect is not apparent equally in all directions around the windfarm footprint and further analyses of these and similar data from other windfarms may be merited to more accurately quantify the magnitude and spatial extent of the effect and to understand variation in the displacement effect around individual windfarms and between them.




From Irwin et al. (2019)




From O'Brien et al. (2012)

**Figure 67 Distribution of red-throated divers in the OTE SPA in 2018 (left) and 2001/02 to 2006/07 in relation to the locations of offshore wind farms.**

The primary habitats of value to wintering red-throated divers in the OTE SPA are shallow sandbanks up to 20 m deep. Therefore, additional protection of these habitats would potentially result in increased usage of those areas by the existing population of birds and potentially better overwinter survival of the current red-throated diver population, thus maintaining it. In addition, the aim of this measure would also be to ensure population growth within the remaining, protected, habitat within the SPA.

The area of habitat within the OTE SPA is estimated across three levels of buffer zone around the wind farm clusters in the SPA (Table 161). These levels were based upon current predicted displacement levels around offshore wind farms (Heinänen et al. 2020), who showed strong displacement at 5 km from wind farm in the German Bight and statistically significant difference in red-throated diver abundance as much as 12 km from offshore wind farms (a distance supported by the analyses reported by APEM 2021). However, the responses of red-throated divers to offshore wind farms are not simple and displacement does not occur in isolation from other pressures and preferences. Different responses to impact should be expected between areas where habitat is abundant and carrying capacity is high, compared with areas where habitat is more limited and carrying capacity is much lower. It is also important to consider the pattern of preferred habitat and location of offshore wind farms when considering displacement distances (Figure 68). If birds have a preference for shallow sand banks and are displaced from them by offshore wind farms they may move to the next available preferred habitat. The distance that this habitat is from the wind farm may appear as a displacement distance, but it may be that birds would not move as far if preferred habitat was closer. Consequently, the distances used in assessing potential habitat loss here should be considered precautionary and future monitoring and research of displacement of red-throated divers is needed to resolve the level and pattern of displacement that occurs within the OTE SPA.

**Table 161 Potential habitat loss from OTE SPA from existing offshore wind farms within the SPA and three possible buffer distances.**

| Buffer size | 0 km | 8 km | 10 km | 12 km |
|---|---|---|---|---|
| Area (km²/%) of potential habitat loss | 158.7/4.0% | 1209.0/30.8% | 1500.9/38.3% | 1788.9/45.6% |
| Remaining available habitat inside OTE SPA (km²/%) | 3761.0/96.0% | 2710.7/69.2% | 2418.8/61.7% | 2130.8/54.4% |
| Difference in habitat lost to remaining | 3602.3 | 1501.7 | 917.8 | 341.9 |
| Ratio of habitat loss to remaining | 1:23.7 | 1:2.2 | 1:1.6 | 1:1.2 |

It is apparent from the bathymetry showing preferred habitat depths that compensatory measures should be focussed on the nearshore waters of the southern and northern sections of the SPA. The north-east section is mostly in water deeper than 20 m and so, while it may be used by red-throated divers, it is likely to be less than optimal habitat.

The ratio of habitat loss to habitat strictly protected for the Voordelta marine reserve was 1:10. It is apparent that the remaining habitat inside the OTE SPA is insufficient to reach a ratio of much more than 1:2 for the 8 km and 10 km buffers, and it is not possible to reach a 1:1 ratio for the 12 km buffer. Compensation through improvement in habitat would therefore need to include habitat improvements in other SPAs for wintering red-throated divers. Other SPAs in the UK designated for their wintering red-throated diver populations include the Liverpool Bay SPA, Greater Wash SPA, Firth of Forth SPA, Outer Firth of Forth and St Andrews Bay Complex SPA and Solway Firth SPA. Most of these other SPAs are also subject to displacement pressures from offshore wind farms, or other pressures and impacts. Consequently, the application of a marine reserve approach to compensation may be better considered at a strategic level and considered across all of the SPAs for wintering red-throated diver subject to pressures from offshore wind farm developments, including Liverpool Bay SPA (Section 10).

Given the combined issues of uncertain displacement distance, the apparently complex effect of offshore wind farms on the spatial distribution of red-throated divers in the OTE SPA and the remaining available areas within the OTE SPA where a reserve may be created, a further option may be to provide marine reserves in appropriate locations within the SPA and then add further marine reserves within the footprints of the wind farms inside the SPA as these are decommissioned. Assuming that further consents to re-power or extend the life of the existing wind farms are not granted, this approach could assure the integrity of winter red-throated divers in UK SPA and correct the negative effects of the existing projects on the distribution of the species within the OTE SPA.

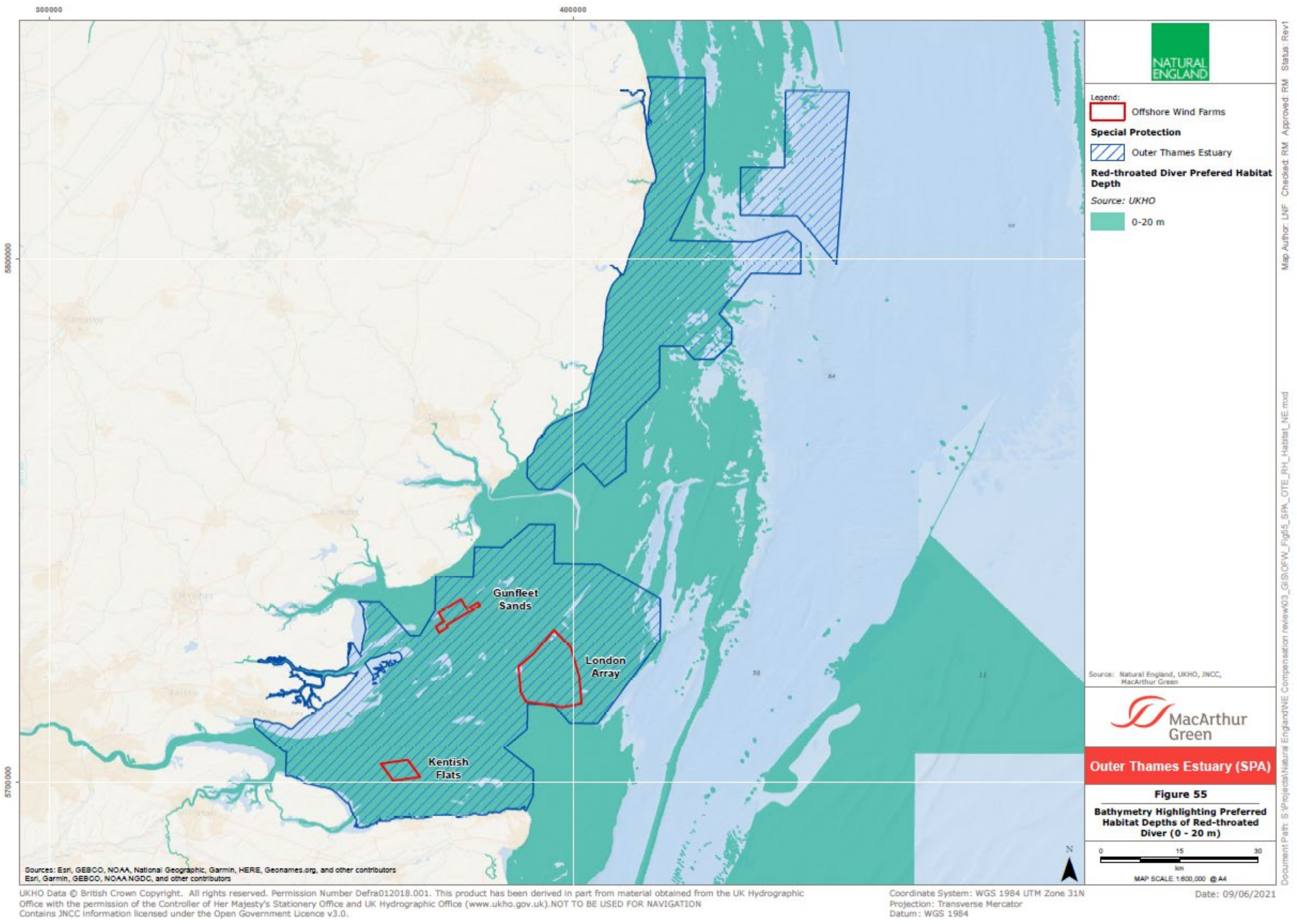



**Figure 68 Bathymetry of the OTE SPA highlighting the preferred depth of red-throated diver (0 - 20 m).**

#### 9.7.1.3 *Removal of existing wind farms*

The footprint of the projects within the SPA is 159 km$^2$ with a combined area of 1,948 km$^2$ within the OTE SPA of a 12 km buffer around the wind farms (this takes account of the overlap between the 12 km buffers around the Gunfleet Sands wind farms and the London Array wind farm). Thus, a large proportion of the OTE SPA would have this pressure removed if these wind farms were removed. It is also important to note that the wind farms are not a permanent feature and will be decommissioned in the future. Assuming that consents for the projects inside the OTE SPA were for 25 years the existing projects will begin to be decommissioned between 2030 to 2038 (Table 162), though the decommissioning process is likely to take more than one year. Once removed it is unknown how long it might take for the area to be used by red-throated divers again.

**Table 162 Commissioning and decommissioning years of offshore wind farms within the OTE SPA.**

| Project | Commissioning year | Decommissioning year |
|---|---|---|
| Kentish Flats | 2005 | 2030 |
| Gunfleet Sands 1 & 2 | 2010 | 2035 |
| London Array | 2013 | 2038 |
| Gunfleet Sands 3 | 2013 | 2038 |

Analysis to determine whether there was a statistically significant change in the distribution of red-throated divers following wind farm construction was beyond the scope of this study. However, existing modelling for EA1N and EA2 offshore wind farms has shown that displacement likely occurs out to 7 km from the wind farms, but that there may have been little change in red-throated diver distribution between the pre-construction and post-construction phases of the wind farms within the OTE SPA (see Figure 55). However, APEM's (2021) analysis of pre-, during and post-construction monitoring of red throated diver distribution around London Array OWF showed areas of statistically significant declines in diver density between pre- and post-construction centred around the windfarm footprint and areas of significant increase in more distant areas, entirely consistent with the hypothesis that there has been a direct displacement effect.

#### 9.7.1.4 *Compensation ratios*

As discussed above, it is suggested that the solution to an adverse effect on site integrity being concluded on the basis of changes to the distribution of birds within the SPA, rather than an impact on population size, is that strict marine reserves are used to create better quality/availability of habitat for birds. This follows the approach undertaken at Voordelta SPA where habitat loss from port expansion was compensated for through marine reserves at a 1:10 ratio for the area lost to the area protected. Since it is apparent that this could not be achieved within the OTE SPA with a 12 km buffer around existing wind farms, it may need to be applied across multiple SPAs. In this circumstance the other SPAs should be in the BDMPS spatial areas identified in Section 9.4.

9.7.2 SW North Sea

Within the SW North Sea BDMPS spatial area there is one other SPA designated for its non-breeding red-throated diver population: the Greater Wash SPA. This is a huge SPA along the coastal waters of Yorkshire south to Suffolk covering about 3,536 km$^2$.

9.7.2.1 *Reducing disturbance by vessel activity*

There is likely to be areas of significant shipping activity within the Greater Wash SPA, based on the MMO analysis of shipping density in 2012 (Figure 69). These data showed particularly high densities of shipping off the mouth of the Humber estuary and off the north Norfolk and Lincolnshire coast, some of which is likely to be boat traffic to and from existing offshore wind farms. Much of this usage appears to be within shipping lanes in and out of existing ports and harbours, such as the Port of Hull. However, it appears that much of the high-density traffic in the SPA occurs in the summer (Figure 70). Scope for compensation through reducing or concentrating commercial vessel activity within the SPA may be limited.

Amongst this commercial shipping, there would appear to be a lot of potential of recreational vessel activity (Figure 66). However, this activity is typically concentrated in the summer months, when red-throated divers are absent from the SPA. Compensation measure may be limited here too.

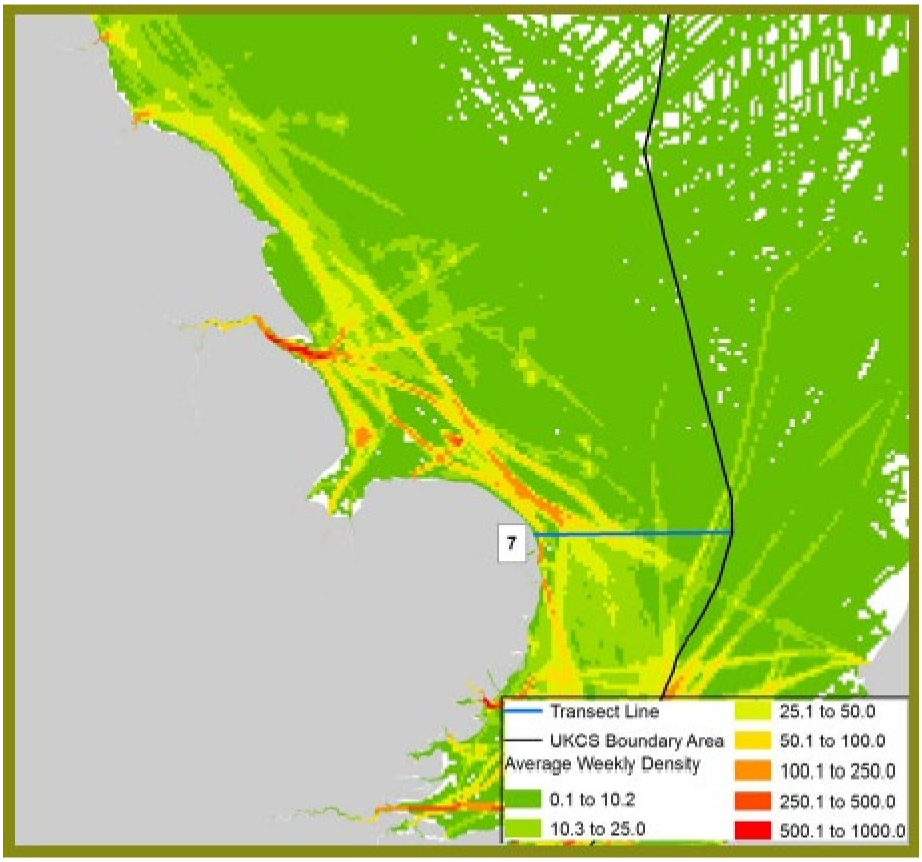



**Figure 69 East Coast vessel density and routes 2012 (from MMO 2014).**

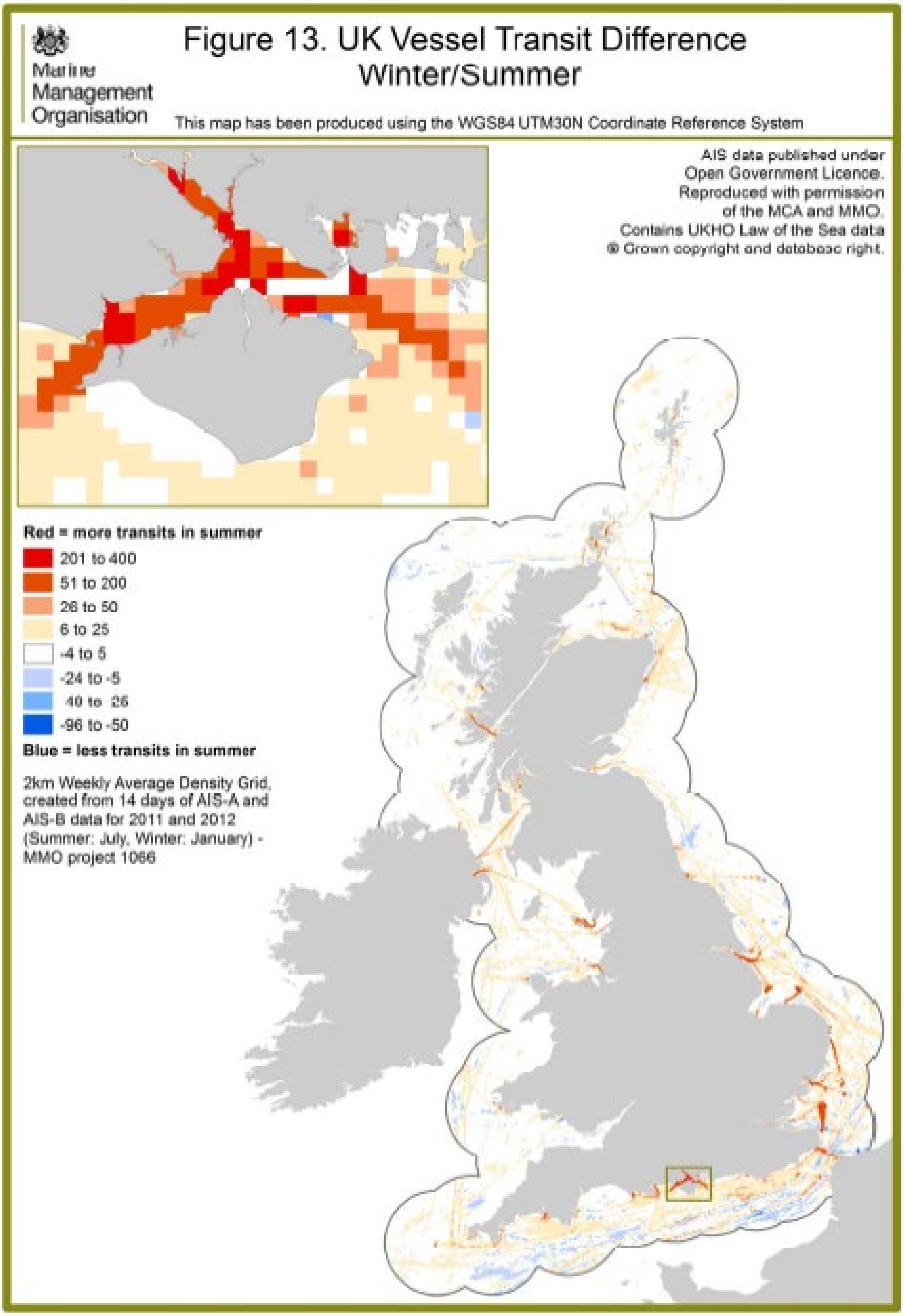



**Figure 70 UK vessel transit difference winter/summer (from MMO 2014).**

#### 9.7.2.2 *Creating a marine reserve*

If compensation to habitat loss from OTE SPA was applied in the Greater Wash SPA there would need to be between 159 km$^2$ and 1,789 km$^2$ of the area within the SPA managed as a strict marine reserve. This is between 4.5% and 50.6% of the total area within the SPA and only for compensating at a 1:1 ratio. As discussed above a much larger compensation ratio was applied in the Voordelta case (1:10). At this ratio, between 45% and 506% of the SPA area would need to be in strict marine reserve to provide compensation. Clearly, the high level of uncertainty in the level of habitat loss through displacement in OTE SPA is very important in deciding the potential for compensation in the Greater Wash SPA. It is also important to take account of the existing pressures on red-throated divers in the Greater Wash SPA, particularly from existing offshore wind farms (Webb et al. 2017) and likely also from shipping.

#### 9.7.2.3 *Removal of existing wind farms*

The footprint of the projects within the SPA is 71 km$^2$. This excludes the LID and Lincs wind farms that are technically not in the SPA, but are surrounded by the SPA, so functionally the impacts on red-throated diver are no different than it being inside the SPA. Including the LID and Lincs wind farms results in a wind farm footprint of 123 km$^2$ in the SPA, without including buffers. This is similar to the area within the OTE SPA (159 km$^2$). It is also important to note that the wind farms are not a permanent feature and will be decommissioned in the future. Assuming that consents for the projects inside the Greater Wash SPA were for 25 years the existing projects will begin to be decommissioned between 2029 to 2040 (Table 163), though the decommissioning process is likely to take more than one year. Once removed it is unknown how long it might take for the area to be used by red-throated divers again.

**Table 163 Commissioning and decommissioning years of offshore wind farms within the OTE SPA.**

| Project | Commissioning year | Decommissioning year |
|---|---|---|
| Westermost Rough | 2015 | 2040 |
| Humber Gateway | 2015 | 2040 |
| LID* | 2009 | 2034 |
| Lincs* | 2013 | 2038 |
| Scroby Sands | 2004 | 2029 |
| * Not within the SPA footprint | | |

### 9.7.3 All other BDMPS spatial areas in the UK

Within the remaining BDMPS spatial areas there are four other SPAs designated for non-breeding red-throated diver populations: Moray Firth SPA, Outer Firth of Forth and St Andrews Bay Complex, Solway Firth and Liverpool Bay / Bae Lerpwl. This covers a total area of 8,368 km$^2$ (Table 164Table 173).

**Table 164 SPAs in all other BDMPS spatial areas in the UK.**

| SPA | Area ($km^2$) |
|---|---|
| Moray Firth SPA | 1,762 |
| Outer Firth of Forth and St Andrews Bay Complex | 2,721 |
| Solway Firth | 1,357 |
| Liverpool Bay / Bae Lerpwl | 2,528 |
| Total area | 8,368 |

*9.7.3.1 Reducing disturbance by vessel activity*

There is generally less shipping activity within the remaining SPAs than in either the OTE SPA or the Greater Wash SPA, based on the MMO analysis of shipping density in 2012 (Figure 71 UK vessel density grid 2012 (From MMO 2014).). Among the remaining SPAs there appears to be higher traffic levels in the Liverpool Bay SPA than the other sites, with much lower levels in the Solway Firth SPA. Much of this shipping density appears to be within shipping lanes in and out of existing ports and harbours, such as the Port of Liverpool, Leith and Inverness. However, it appears that much of the high-density traffic in the SPA occurs in the summer (Figure 70). Scope for compensation through reducing or concentrating commercial vessel activity within these SPAs may be limited.

Amongst this commercial shipping, there would appear to be a lot of potential of recreational vessel activity (Figure 66). However, this activity is typically concentrated in the summer months, when red-throated divers are absent from the SPAs. Compensation measures may be limited here too.

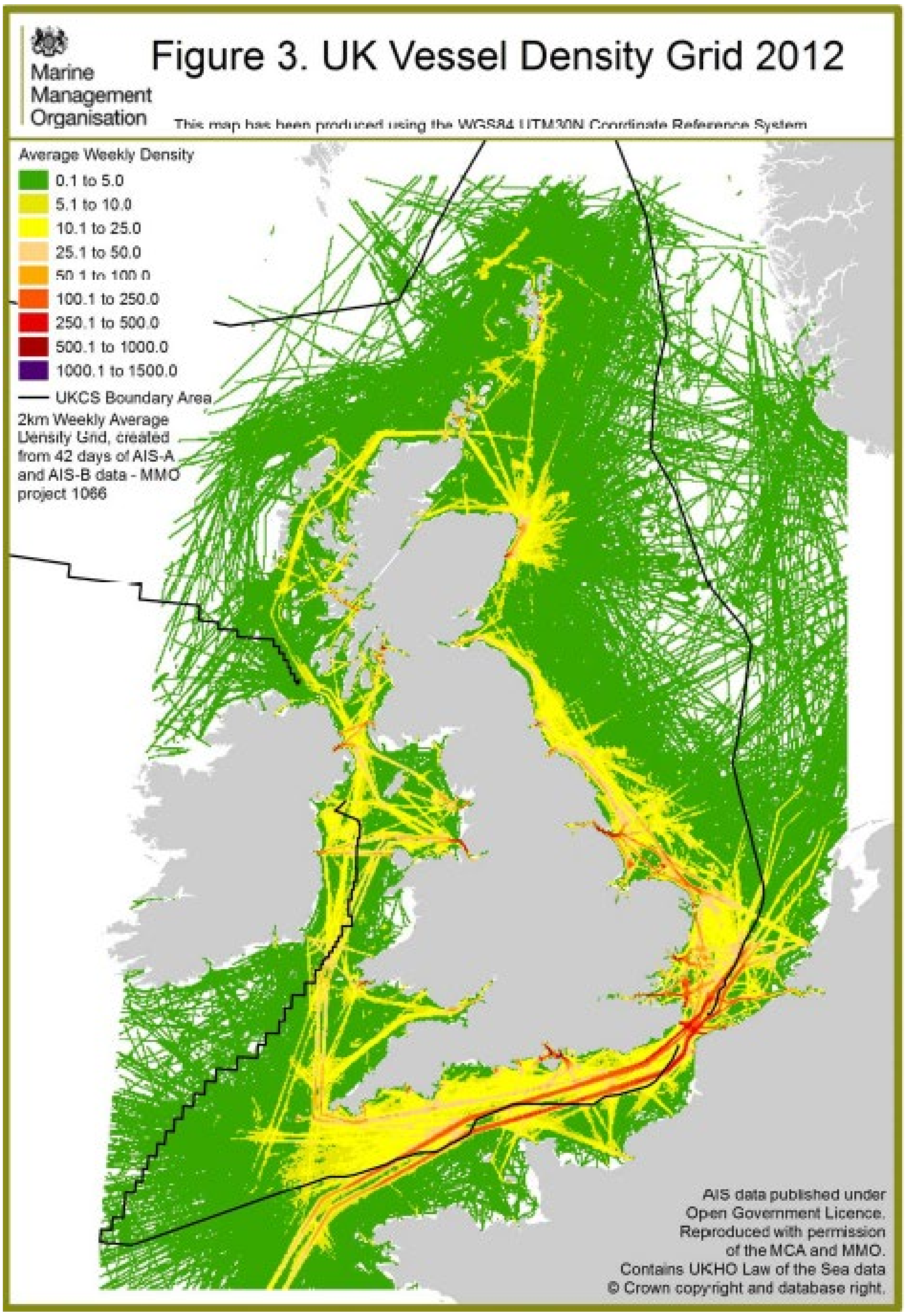



**Figure 71 UK vessel density grid 2012 (From MMO 2014).**

### 9.7.3.2 *Creating a marine reserve*

If compensation to habitat loss from OTE SPA was applied in the remaining SPAs there would need to be between 159 km$^2$ and 1,789 km$^2$ of the area within the SPAs managed as a strict marine reserve. This is between 1.9% and 21.4% of the total area within the SPAs and only for compensating at a 1:1 ratio. As discussed above a much larger compensation ratio was applied in the Voordelta case (1:10). At this ratio, between 19% and 214% of the SPAs area would need to be in strict marine reserve to provide compensation. Clearly, as with the Greater Wash, the high level of uncertainty in the level of habitat loss through displacement in OTE SPA is very important in deciding the potential for compensation in the remaining SPAs. It is also important to take account of the existing pressures on red-throated divers in the remaining SPAs, particularly from existing offshore wind farms and likely also from shipping. Only two of the SPAs have existing wind farms within them: Liverpool Bay (see Section 10.7.1.2) and the Solway Firth. While compensation in Liverpool Bay is discussed below, it seems that there is little pressure on red-throated divers in the Solway Firth from Robin Rigg offshore wind farm, as the site was not used by important numbers of red-throated divers before construction (Walls et al. 2013).

#### *9.7.3.3 Removal of existing wind farms*

As described above, only two SPAs in all other BDMPS spatial areas have offshore wind farms within them: Liverpool Bay SPA and Solway Firth SPA. The effects of removal of offshore wind farms within Liverpool Bay SPA are described in Section 10.7.1.3 for English waters. Since the Robin Rigg wind farm did not exert a pressure on the red-throated diver population in the Solway Firth SPA removal of this wind farm would not provide any compensation.

## 9.8 Assessment of confidence

The assessment showed that red-throated divers were sensitive to disturbance from vessels, and that there were relatively high levels of shipping activity with the OTE SPA, and several other SPAs where compensation could be applied. However, shipping was concentrated in narrow shipping lanes, which are typically deeper than the surrounding seabed. Assessment of confidence in the metrics for measure was summarised in Table 165 and the narrative provided in Table 167.

The assessment also showed that the area that would need to be included in a marine reserve was larger than the area available within the OTE SPA, depending on the size of the buffer applied around the existing wind farms inside the SPA and the compensation ratio applied. Confidence that the impact on red-throated divers has not been underestimated increases with the size of the buffer zone over which the impact is assumed to occur, but confidence in the ability to apply the measure decreases with increasing buffer size. Two studies were available that modelled the displacement based on empirical data and each had relatively similar results. The most recent analyses by APEM (2021) have provided further evidence consistent with there being a direct displacement effect following the construction of London Array OWF and that this influence may extend up to 12km from its boundary. However, this effect is not apparent equally in all directions around the windfarm footprint and further analyses of these and similar data from other windfarms may be merited to more accurately quantify the magnitude and spatial extent of the effect and to understand variation in the displacement effect around individual windfarms and between them. Confidence in the metrics for the marine reserve measure was summarised in Table 166and the narrative provided in Table 168.

Unlike the other assessments of confidence, the principal aim of the compensation measures studied here was not to increase the population size (or demographic rates) of the population in the UK SPA network. The aim was to determine potential methods to address the adverse effect being caused by impacts on the conservation objective to maintain the distribution of the species within the site. The integrity of the network could be maintained through creation and management of marine reserves within the OTE SPA and likely other SPAs. Confidence would be ensured through the combination of relatively large compensation ratios (such as the 1:10 ratio used in the Voordelta case) and monitoring with adaptive management. Given the wind farms within the OTE SPA will be reaching the end of their consented lifespan over the next 10 to 20 years, it may be advantageous to have these added to the marine reserve areas over time, thus reducing the need for compensation measures being needed in other areas.

**Table 165 Assessment of confidence in reducing vessel disturbance as a compensation method.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Effect of shipping on red-throated diver distribution | n/a | Mendel et al. 2019, Schwemmer et al. 2011, Irwin et al. 2019, EA1n & EA2 assessment | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Presence of high density shipping in OTE SPA | n/a | MMO 2014 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Potential for recreational boating in OTE SPA | n/a | MMO 2014 | Robust | Robust | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | VERY HIGH |

**Table 166 Assessment of confidence in creating a marine reserve as a compensation method.**

| Metric | Value | Source | Quality of evidence | Consistency of evidence | Amount of evidence | Type of evidence | Overall evidence score | Applicability | CONFIDENCE |
|---|---|---|---|---|---|---|---|---|---|
| Preference for specific water depths | 10 – 30 m | Vilela, et al. 2020, EA1n & EA2 assessment | Robust | Medium | Robust | Robust | ROBUST | HIGH | VERY HIGH |
| Displacement distance | 12 km & 7 km | APEM 2021, Mendel et al. 2020, EA1n & EA2 assessment | Robust | Medium | Medium | Robust | MEDIUM | HIGH | HIGH |
| Area of effective habitat loss in the SPA | 159 – 2,324 km2 | As calculated | Robust | Low | Medium | Robust | MEDIUM | HIGH | HIGH |
| | | | | | | | | **OVERALL CONFIDENCE SCORE** | HIGH |

MacArthur Green

**Table 167 Narrative justifying the evidence and applicability scores for metrics used in reducing vessel disturbance as a compensation method.**

| Metric | Narrative |
|---|---|
| Effect of shipping on red-throated diver distribution | Several studies have reported robust evidence that red-throated divers are strongly affected by shipping. Two studies were from the OTE SPA (Irwin et al. 2019). So, the applicability was considered high. These combined to give a confidence value of very high. |
| Presence of high-density shipping in OTE SPA | The study from the MMO (2014) provided robust evidence that there was high density shipping in the OTE SPA (high applicability). Confidence was therefore very high. |
| Potential for recreational boating in OTE SPA | The study from the MMO (2014) provided robust evidence that there was high potential for recreational boating in the OTE SPA (high applicability). Confidence was therefore very high. |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the effects of shipping on red-throated divers was very high. However, it is difficult to assess how to apply reduction from vessel activity to compensation. Available evidence suggests vessels already use existing routes, in deeper water. **Without more study confidence in this approach to compensation should be low**. |

**Table 168 Narrative justifying the evidence and applicability scores for metrics used for creating a marine reserve as a compensation method.**

| Metric | Narrative |
|---|---|
| Preference for specific water depths | Two studies have reported preferences for water depth from modelling results. Vilela et al. (2020) was a robust study in a peer reviewed journal. This study was from Germany so the applicability would be medium. The study from the EA1n & EA2 assessment was not in a peer reviewed journal, so should not be given as high an evidence score as the Vilela et al. (2020) study. However, the study was of the OTE SPA, so the applicability was high. **As such the very high confidence in this finding should probably be reduced to high**. |
| Displacement distance | The two studies were based on sound and robust digital aerial survey data. Both studies used similar modelling approaches and were undertaken by highly competent researchers. The data were all quantitative and matched to the sites being assessed. The results were broadly similar but found slightly different results (though in the context of their application these differences can be very important). The Mendel et al. (2020) study was published in a peer reviewed journal, while the EA1n & EA2 assessment was not. Both studies found slightly different, but important, displacement distances, so the consistency of evidence was medium. Amount of evidence was assessed as medium as this only provides two results. The combination of these factors resulting in an evidence score of medium. Applicability was assessed as high as two studies were from the OTE SPA. Overall confidence was high as a result. |
| Area of effective habitat loss in the SPA | This metric was based on two studies (described above) in displacement distance. The resulting range of areas of effective habitat loss was so large that the consistency was assessed as Low. This gave an overall evidence score of medium. Applicability was high as this was directly applied to the OTE SPA, giving a confidence of high. **However, due to the very large range of possible areas of** |

| Metric | Narrative |
|---|---|
| | **effective habitat loss, this should probably be expressed as a range of confidence from low (for the smallest area) to high (for the largest area).** |
| **OVERALL CONFIDENCE SCORE** | Overall, the confidence in the assessment of the marine reserves approach was assessed as high. **However, due to the range of potential area this should be amended to a range of confidence from low (for the smallest area) to high (for the largest area).** |

While there was a very high overall confidence in the effects of vessel disturbance on red-throated divers, it is difficult to assess how to apply reduction from vessel activity to compensation. From the available evidence it appears that vessels already largely use existing shipping routes, which are generally in deeper water, which is not as favoured by red-throated divers. Without much more detailed study than was possible here confidence in this approach to compensation should perhaps be considered low.

A range of confidence in the marine reserve approach, from low to high, was given due to the uncertainty in the displacement distance from the wind farms. However, further, and much more detailed, analyses would be needed in order to begin the next steps towards the implementation of this compensation measure. The Voordelta compensation plan was a large multidisciplinary approach that required multiple experts from multiple agencies to develop into the final plan, and it is likely a similar approach would be needed here.

### 9.9 Future monitoring and adaptive management

It is apparent that further analysis is needed to determine the patterns of displacement of red-throated divers in the OTE SPA. The data exists to statistically compare the abundance of birds in a grid across the SPA before and after construction of the wind farm. This could be a more effective approach that the concentric rings that have been used elsewhere (e.g. Mendel et al. 2019) as that approach assumes an equal effect in all directions from the wind farm, which is apparently not occurring in the OTE SPA (APEM 2021). This would be important in determining whether future compensation measures are effective in improving the distribution of birds within the site. It is assumed that the aim would be to return the use of the site to the distribution seen within the SPA prior to construction of the wind farms, so future monitoring would need to collect spatial data on the distribution of divers across the SPA and these would need to be compared with the visual aerial survey data collected prior to the construction of the wind farms.

As wind farms are removed (either as part of a compensation measure or due to decommissioning) further digital aerial surveys and spatial analysis would be needed to show whether birds return to these areas.

It is apparent from the monitoring programme for the Voordelta SPA that a marine reserves approach to compensation would benefit for a similar approach to monitoring focused on the improvement in quality and/or availability of habitat for red-throated divers. This may therefore require better understanding of the habitat requirements of red-throated divers in winter in the OTE SPA, so may require forage fish and benthic sampling within areas already preferred and avoided by divers in the SPA. Adding the footprints of existing wind farms inside the SPA to these marine reserves as they are decommissioned should provide adequate compensation without the need for a strategic level assessment.

#### 9.9.1 Future research

There are clearly important gaps in the knowledge base around the scale of the displacement effect of red-throated divers. It appears that there may be a different displacement distance occurring in the OTE SPA than in the German Bight. Further modelling of the effects of offshore wind farms across multiple sites may be necessary to resolve this issue. However, it is clear that data exist across three locations in the UK: OTE SPA, Greater Wash SPA and Liverpool Bay SPA.

Further research on the use of SPAs by non-breeding red-throated divers in the UK in relation to shipping would also be useful, with evidence from large scale digital aerial surveys for multiple sites now possible including OTE SPA, Liverpool Bay SPA and Moray Firth SPA. The key to applying compensation measures to these sites may be to understand what shipping could be concentrated into existing shipping lanes, reducing pressure on birds elsewhere in the SPAs. There is also little information on the distribution and seasonality of small vessels, particularly recreational vessels, that do not carry VMS or AIS equipment. Results on the closure of areas of suitable habitat to all vessels (e.g. from the Voordelta compensation scheme) on red-throated diver distribution would also be helpful.

### 9.10 Summary

The review found that the only compensation measure recommended by Furness et al. (2013) which was still potentially relevant for red-throated divers at OTE SPA was reducing disturbance by vessel activity. However, the seasonal pattern of use of the SPA by shipping suggests that this measure could have relatively little benefit for wintering red-throated divers. Since Furness et al. (2013) new evidence was found on an existing compensation measure for a similar SPA in the Netherlands. The Voordelta SPA was impacted through habitat loss caused by the expansion of the Port of Rotterdam. The SPA was designated for multiple species, including wintering red-throated diver. The compensation measures suitable for divers included creation of marine reserves within the SPA. Within these reserves seasonal exclusion of all vessels and low flying aircraft reduced disturbance. Exclusion of bottom trawling fisheries improved habitat quality. Finally, since the presence of operational wind farms from within the OTE SPA causes the adverse effect on site integrity, their removal would represent adequate compensation. Whether this is practical is beyond the remit of this report, but its effectiveness is highly likely to be successful.

The assessment of the area within the OTE SPA showed that there was insufficient area beyond the buffers around the existing wind farms to provide marine reserves at a ratio of 1:10. This was the ratio applied in the Voordelta case. This and the presence of existing pressures from other offshore wind farms in UK waters resulted in the recommendation for further work to undertake a strategic level assessment of the available suitable habitat for marine reserves within all SPAs that include wintering red-throated divers as a feature.

Confidence in the marine reserves method for compensation of red-throated divers was assessed as low to high, depending on the size of the area, the assessment of other compensation measures was difficult to apply using the approach applied to other SPA features in this study. Confidence in the marine reserves approach would be achieved through a very high compensation ratio (1:10) combined with adaptive management. Removal of existing wind farms should have complete confidence, but only if the areas currently occupied by offshore wind farms was previously used by red-throated divers. It was recommended that further analysis is completed to determine whether these sites were important for the OTE red-throated diver prior to construction of the wind farms.

# 10 LIVERPOOL BAY / BAE LERPWL SPA – NON-BREEDING RED-THROATED DIVER

The Liverpool Bay / Bae Lerpwl SPA (hereafter Liverpool Bay SPA) is in north-west England and north Wales in the eastern Irish Sea. The site is very large (2,528 km$^2$) and covers the coastal seas from the east coast of Anglesey to Morecambe Bay. The habitat in the SPA is mostly sandy substrates and the sea has a large tidal range (6 – 8 m) but relatively low tidal currents.

## 10.1 Conservation status of red-throated diver

The biogeographic population was estimated at 300,000 individuals, of which 21,500 are estimated to winter in Great Britain (Woodward et al. 2020) and 2,000 individuals in all-Ireland (Crowe et al. 2008). The IUCN lists the global population of red-throated diver as "Least Concern".

The UK population was green listed in Birds of Conservation Concern (BOCC) 4 (Eaton et al. 2015) having been previously listed as 'amber' in BOCC3 and BOCC2.

It is protected under the Birds Directive as a migratory species. The SPAs in Great Britain were estimated to hold 44% of the Great Britain wintering population of red-throated divers present in 2008 (Stroud et al. 2016) an increase of 19% in SPAs estimated in the previous SPA review (Stroud et al. 2001).

## 10.2 Citation population size

The SPA was classified in August 2010 based on an estimated 1,171 wintering individuals (2004/05 – 2010/11, peak mean estimate).

## 10.3 Conservation objectives

The site has conservation objectives, "to ensure that, subject to natural change, the integrity of the site is maintained or restored as appropriate, and that the site contributes to achieving the aims of the Wild Birds Directive, by maintaining or restoring:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of qualifying features within the site."

More detailed Supplementary Advice on Conservation Objectives (SACO) have not been added for this SPA.

A Site Improvement Plan (SIP) was published in March 2015 (Natural England 2015c). That identified six pressures as a threat to red-throated divers:

- Commercial marine and estuarine fisheries;
- Transportation and service corridors;
- Recreational marine and estuarine fisheries;

- Extraction of non-living resources (i.e. aggregates);
- Siltation; and
- Water Pollution.

Introducing and enforcing appropriate management as necessary was identified as a responsibility of a variety of organisations, depending on the issue, including Natural England, Natural Resources Wales, North Western Inshore Fisheries Conservation Authority (IFCA) and the Marine Management Organisation (MMO).

The SIP stated that the commercial marine and estuarine fisheries issue was only in English waters and was related to dredges, benthic trawls, and seine nets being used over stony reef and bedrock communities. Since these are not preferred habitat types of red-throated divers they are unlikely to be key in providing future compensation measures.

The transportation and service corridors issue was concerned with research in to the effects of current vessel disturbance on qualifying features only in English waters. This is highly relevant to red-throated diver conservation as they are known to be sensitive to disturbance from shipping (Mendel et al. 2019) and the issues and actions table stated that, "Proposals for shipping and transport routes outside of the established corridors would have high potential for disturbance to SPA birds". The recreational fisheries issue was also only a research issue, due to a potential threat of disturbance to SPA birds from recreational vessels being poorly understood. Aggregate extraction in English waters was also identified as needing further research as the effects on SPA birds were not fully understood.

The effect of moving the dredge disposal site for the Mersey Narrows from the SPA to within the Mersey estuary also required research. This was to determine if moving the site had reduced siltation of habitats within the SPA that could otherwise support SPA features.

Finally, the water pollution issue was only a review of oil spill contingency plans at appropriate intervals to ensure it was still relevant to the risks.

#### 10.3.1 How each Conservation Objective might not be achieved

Among the conservation objectives the following objectives are relevant to the assessment of impacts from offshore wind farm, the remainder are not:

- the extent and distribution of the habitats of the qualifying features;
- the structure and function of the habitats of the qualifying features;
- the supporting processes on which the habitats of the qualifying features rely;
- the populations of each of the qualifying features; and
- the distribution of the qualifying features within the site.

There are two main sources of impact on red-throated divers from offshore wind farm development: displacement from the wind farm and barrier effects resulting in increased energy expenditure.

The CO to maintain or enhance the red-throated diver population and its supporting habitats in favourable condition could be affected through the displacement of red-throated divers from the wind farm and its surroundings, if birds from the SPA used this area prior to the construction of the wind farm. In the absence of empirical evidence, it is assumed that a proportion of red-throated divers will be displaced from the wind farm and that this will influence their ability to gain energy, with a subsequent impact on survival which could in turn impact the population of that qualifying feature.

The maintenance of the population of each of the qualifying features could be affected indirectly through impact to energy budgets from displacement and barrier effects.

### 10.4 Location of compensation

Furness et al. (2015) defined five BDMPS spatial areas for red-throated divers in UK waters in winter. The Liverpool Bay SPA occurs within the "NW England and Wales" area. Consequently, the hierarchy of the locations of compensation are:

1. Liverpool Bay SPA;
2. NW England and Wales; and
3. All other BDMPS spatial areas in the UK.

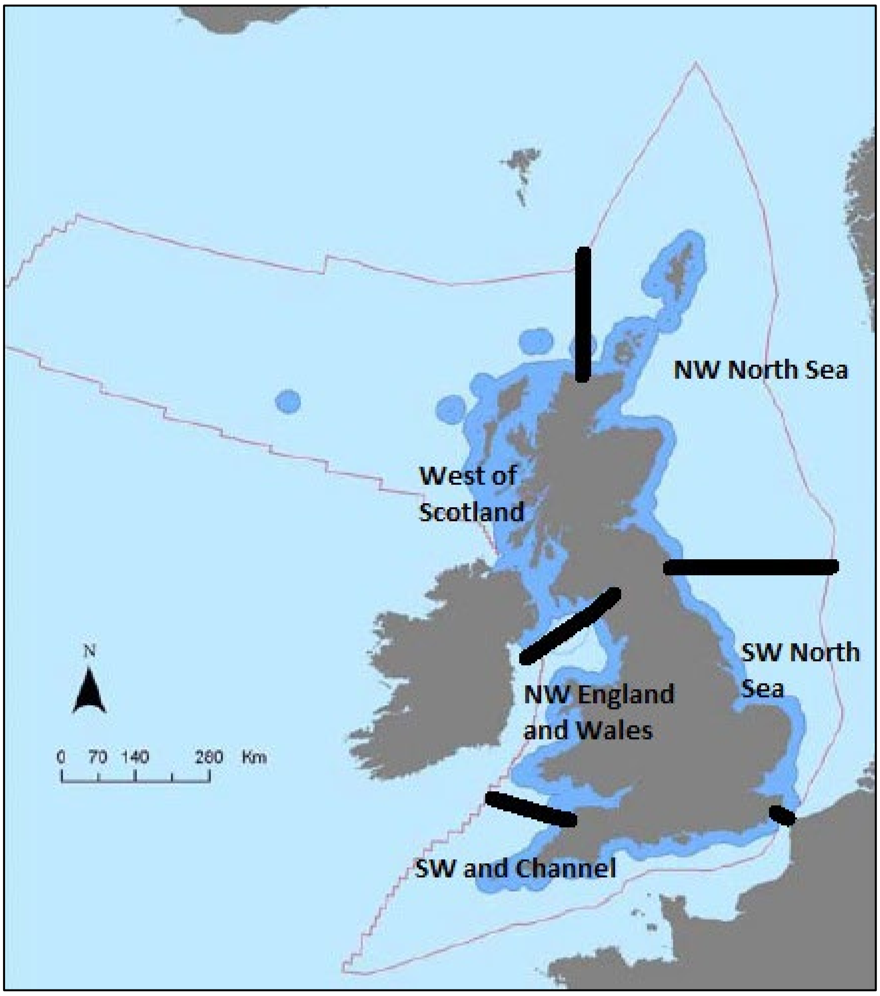



**Figure 72 Defined BDMPS spatial areas for red-throated divers in UK waters in winter from Furness et al. (2015). Limits of UK waters are shown by red line. BDMPS spatial areas extend from the UK coast to the red limit, bounded by the thick black lines marking the sides of each BDMPS area.**

## 10.5 Key biological questions

The key biological questions that need to be addressed in order to assess the potential benefits of compensation measures for the UK SPA network for wintering red-throated diver were developed for OTE SPA. The questions posed for Liverpool Bay SPA were identical (see Section 9.5 and Table 157).

## 10.6 Review of potential compensation measures

### 10.6.1 Reducing disturbance by vessel activity

More recent research has better quantified effect of ship traffic on non-breeding red-throated divers (APEM 2016; Burt et al 2017; Jarrett et al. 2018, Burger et al. 2019, Mendel et al. 2019). Management that could reduce vessel activity during winter in areas used by large numbers of non-breeding red-throated divers could reduce the disturbance to this species. That would most likely reduce energy expenditure as well as allowing birds to spend more time foraging, so would be highly likely to improve overwinter survival and body condition. While it may be difficult to reduce ship traffic, there may be options to limit that to smaller clearly defined shipping lanes, and to set speed limits as birds appear to be disturbed more by faster-moving vessels, and there may be some scope to shift ship traffic to times of year when red-throated divers are not aggregated in these areas (i.e. from winter to summer), or to move some forms of disturbance (such as recreational activity) from areas occupied by red-throated divers to other areas where these birds are not present in large numbers.

#### *10.6.1.1 Answers to the key biological questions (10.5).*

The answers to the key biological question in relation to compensation through reducing disturbance by vessel activity are shown in Table 169.

**Table 169 Answers to Key Biological Questions in assessing the potential for compensation through reducing disturbance by vessel activity.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that non-breeding red-throated divers are displaced by shipping activity in: | |
| 1.1 | Liverpool Bay SPA? | Yes. There is evidence from digital aerial surveys and from modelling of red-throated diver spatial densities in Liverpool Bay SPA that birds are avoiding shipping. |
| 1.2 | NW England and Wales BDMPS area? | No. The only SPA in the NW England and Wales BDMPS is the Liverpool Bay SPA. |
| 1.3 | All other BDMPS spatial areas in the UK? | Yes. There is evidence from digital aerial surveys and from modelling of red-throated diver spatial densities in the OTE SPA that they are avoiding shipping. |

### 10.6.2 Creating a marine reserve

The creation of the Voordelta marine reserve in the Netherlands strictly protected a large area of coastal sea to compensate for expansion of the Port of Rotterdam. In addition to creating this marine reserve with the compensation measures discussed in 9.5.1 three areas within the reserve with restricted access to vessels were created to provide undisturbed areas for common scoter. These areas totalled 5,173 ha.

Monitoring was undertaken on the benthos, birds, fish, physical processes, and human activities (van der Meulen 2016). At present monitoring results appear to be embargoed (e.g. see Borst et al. 2016) and reports are in Dutch. However, a complex integrated monitoring and management approach has been taken (Kinneging et al. 2017) and was proposed to continue for 30 years (Blake et al. 2020) so important results on the improvement in quality of the SPA to the qualifying features are likely to be available in the future.

#### *10.6.2.1 Answers to the key biological questions (10.5).*

The answers to the key biological question in relation to compensation through creation of a marine reserve are shown in Table 170.

**Table 170 Answers to Key Biological Questions in assessing the potential for compensation through creation of a marine reserve.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence that creating a strict marine reserve would improve habitat for red-throated divers in: | |
| 1.1 | Liverpool Bay SPA? | Maybe. There is no direct evidence of this for the Liverpool Bay SPA, however there is direct evidence of the effects of shipping disturbance and offshore wind farms on red-throated divers in this SPA. A marine reserve that protects suitable red-throated diver habitat from disturbance would reduce pressures on birds. |
| 1.2 | NW England and Wales BDMPS area? | No. The only SPA in the NW England and Wales BDMPS is the Liverpool Bay SPA. |
| 1.3 | All other BDMPS spatial areas in the UK? | Maybe. There is no direct evidence of this for the OTE SPA, however there is direct evidence of the effects of shipping disturbance and offshore wind farms on red-throated divers in this SPA. A marine reserve that protects suitable red-throated diver habitat from disturbance would reduce pressures on birds. |

### 10.6.3 Removal of existing wind farms

Removing the three wind farms (Barrow, Burbo Bank and Burbo Bank extension) from inside the English waters within the SPA would greatly reduce the predicted impacts on the distribution of red-throated divers within the SPA. In addition, there are two wind farms beyond the SPA boundary and in English waters, but sufficiently close that they could have an influence within the SPA (Walney projects and Ormonde). Note that there are also existing wind farms within the SPA in Welsh waters, but these were not assessed here.

#### *10.6.3.1 Answers to the key biological questions (9.5).*

The answers to the key biological question in relation to compensation through removal of existing wind farms are shown in Table 171.

**Table 171 Answers to Key Biological Questions in assessing the potential for compensation through removal of existing wind farms.**

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 1 | Is there evidence of displacement of red-throated divers from offshore wind farms in: | |
| 1.1 | Liverpool Bay SPA? | Yes. There is strong evidence for displacement from offshore wind farms in Liverpool Bay SPA. Further evidence on the displacement distance from these wind farms within Liverpool Bay SPA is needed. Additionally, there is evidence of no effect of the Robin Rigg offshore wind farm on red-throated divers in the Solway Firth SPA. |
| 1.2 | NW England and Wales BDMPS area? | No. The only SPA in the NW England and Wales BDMPS is the Liverpool Bay SPA. |
| 1.3 | All other BDMPS spatial areas in the UK? | Yes. There is strong evidence for displacement from offshore wind farms in the OTE SPA and Greater Wash SPA. Further evidence on the displacement distance from these wind farms within these SPAs is needed. |

| No. | Key Biological question | Answers to Key Biological Questions |
|---|---|---|
| 2 | Is there evidence that removal of offshore wind farms would provide additional habitat to red-throated divers in: | |
| 2.1 | Liverpool Bay SPA? | No. There is no evidence from Liverpool Bay SPA, or anywhere else, that red-throated divers will return to the areas currently occupied by offshore wind farms following decommissioning. |
| 2.2 | NW England and Wales BDMPS area? | No. See the answer to question 2.1. |
| 2.3 | All other BDMPS spatial areas in the UK? | No. See the answer to question 2.1. |

## 10.7 Population level assessment

### 10.7.1 Liverpool Bay SPA

#### *10.7.1.1 Reducing disturbance by vessel activity*

Red-throated divers are known to avoid ships (Mendel et al. 2019, Schwemmer et al. 2011). Analysis of Automatic Identification System data by the MMO showed that the majority of the shipping in Liverpool Bay was within shipping lanes (Figure 73) and more transits occurring in summer within the main shipping routes (MMO 2014).

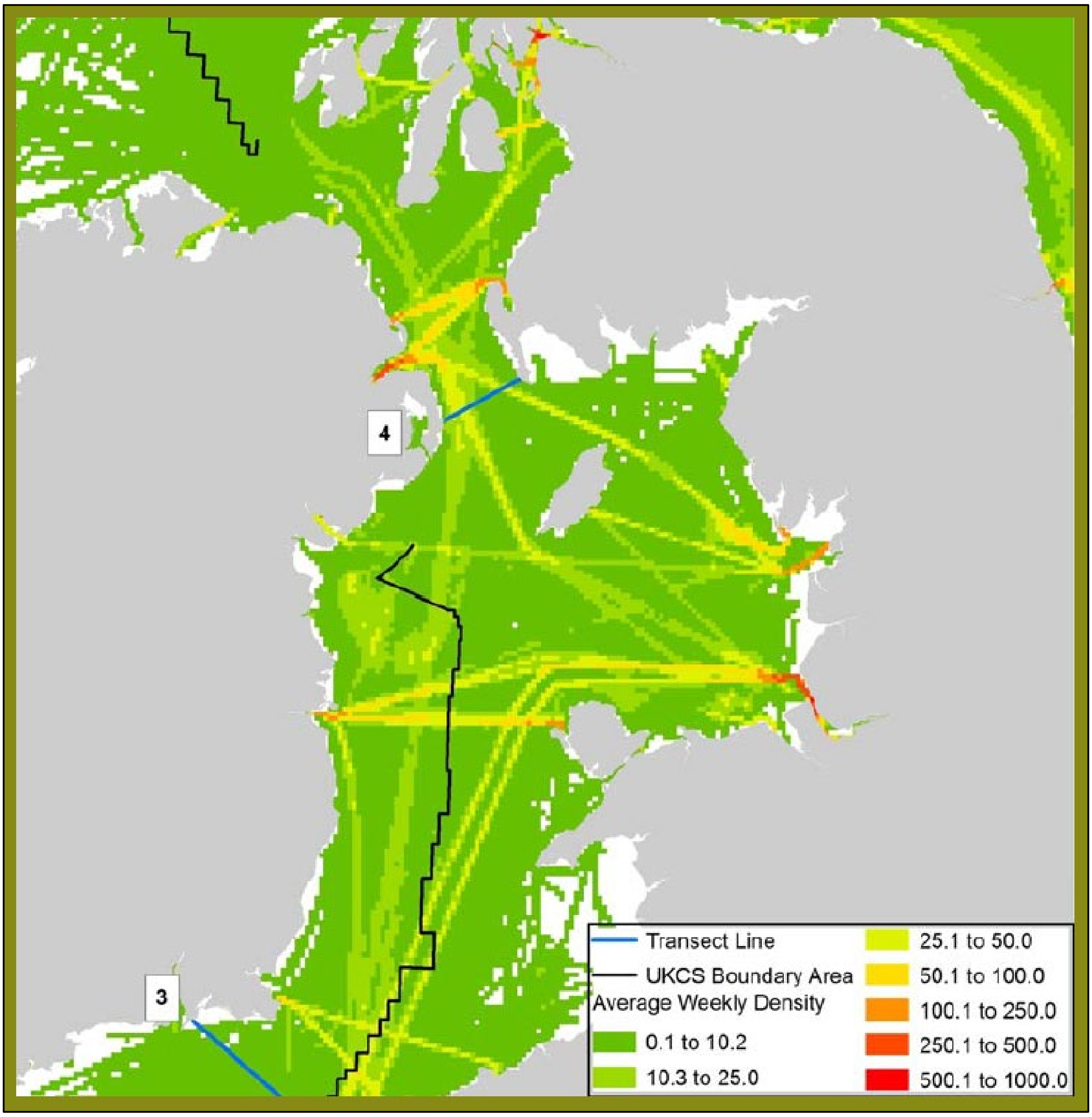
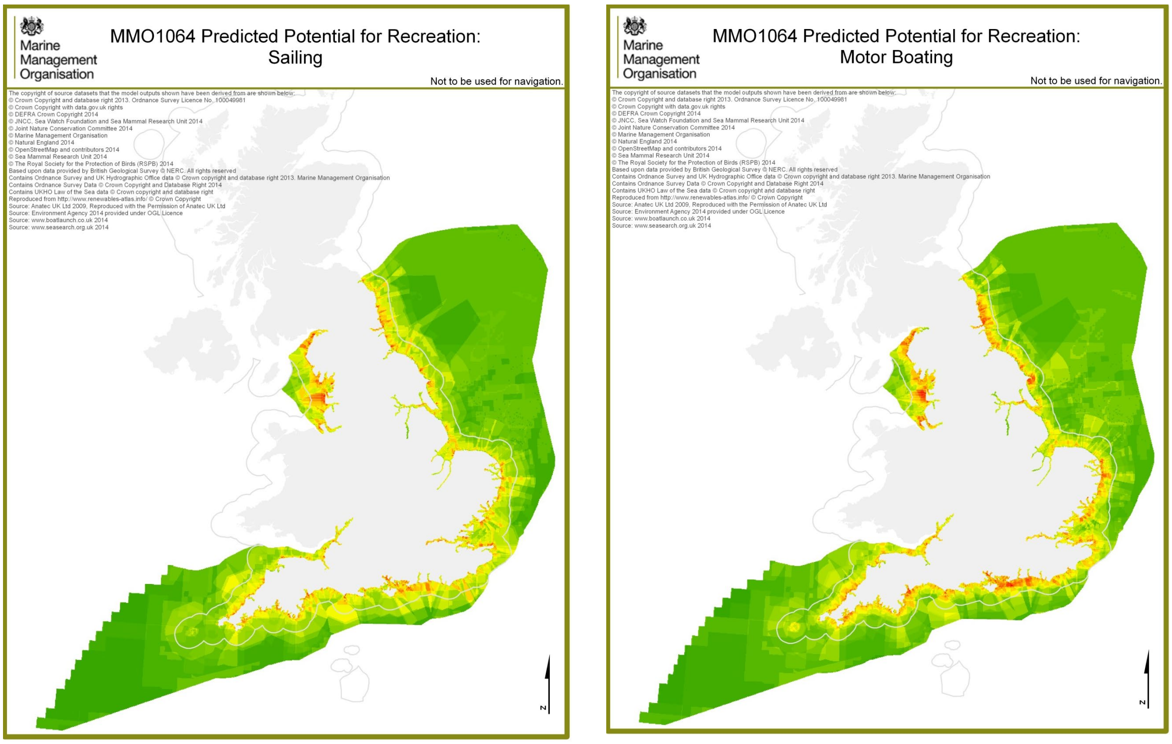



**Figure 73 Irish Sea vessel density and routes from AIS data in 2012 (from MMO 2014).**

However, AIS data are only collected from vessels more than 15 m in length. Recreational vessels are not required to use AIS transmitters, and most do not. The MMO have identified the potential for recreational sailing and motor boating in English waters, including the Liverpool Bay area (Figure 74). This indicates high levels of potential use within the SPA, particularly off the Blackpool area. The analysis was restricted to English waters, so it is unclear if there may be high potential areas off the north Wales coast.




Recreational sailing | Recreational motor boating

**Figure 74 Sailing (left) and motor boating (right) activity map from MMO (2014).**

Most of the area within the SPA appears to show a high potential, with low potential for recreational boating occurring offshore, between England the Isle of Man. Despite this, recreational boating is highly seasonal with a large majority of activity occurring in the summer months (MMO 2014), while red-throated divers occur mostly from October to March. Therefore, reducing recreational boating in the Liverpool Bay SPA in winter is unlikely to result in important changes in disturbance. However, a combination of strict no boating policy within protected areas combined with stakeholder engagement and education could provide a beneficial element to a marine reserve approach to compensation.

*10.7.1.2 Creating a marine reserve*

Monitoring of the Liverpool Bay SPA has been undertaken recently as part of the Burbo Bank Extension offshore wind farm post-consent monitoring. As well as surveying the wind farm and a 4 km buffer the whole of the SPA was also surveyed. The data were analysed using MRSea (Scott-Hayward et al. 2013)

The analysis of the surveys of the Liverpool Bay SPA by Orsted for the post-construction monitoring of the Burbo Bank extension wind farm (Humphries et al. 2018) found that the population of red-throated divers had declined from 2011 to 2014/15 but then remained very similar in 2017/18 (Figure 75).

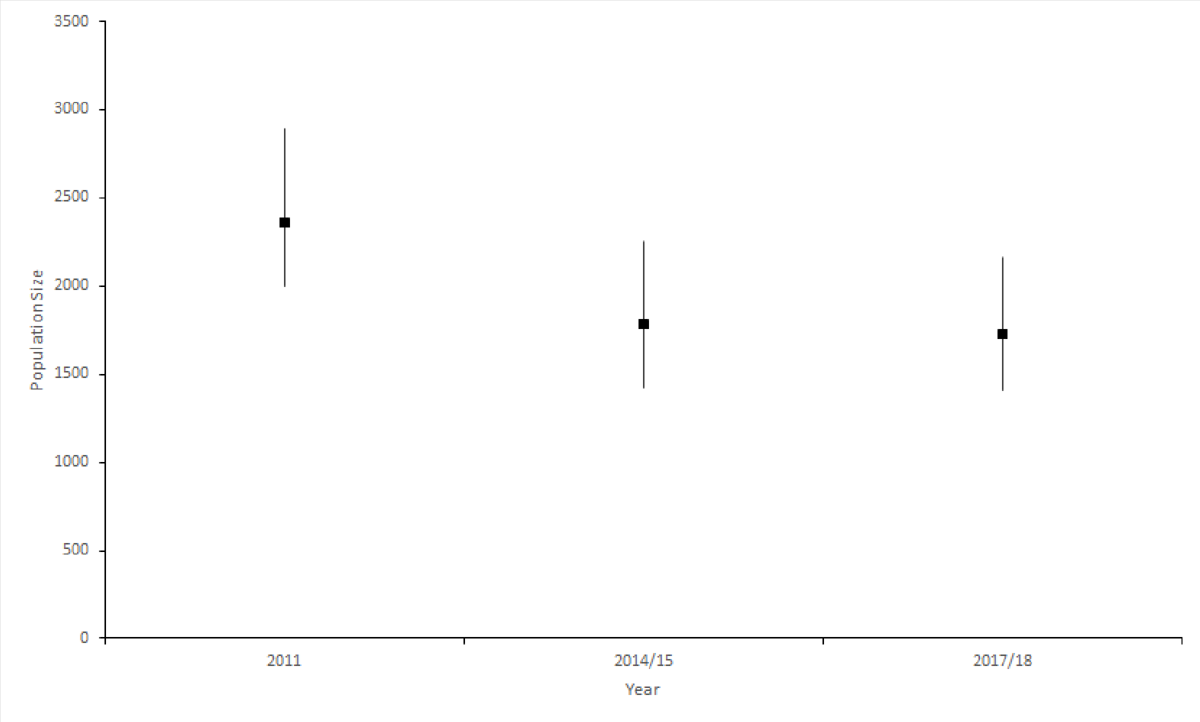
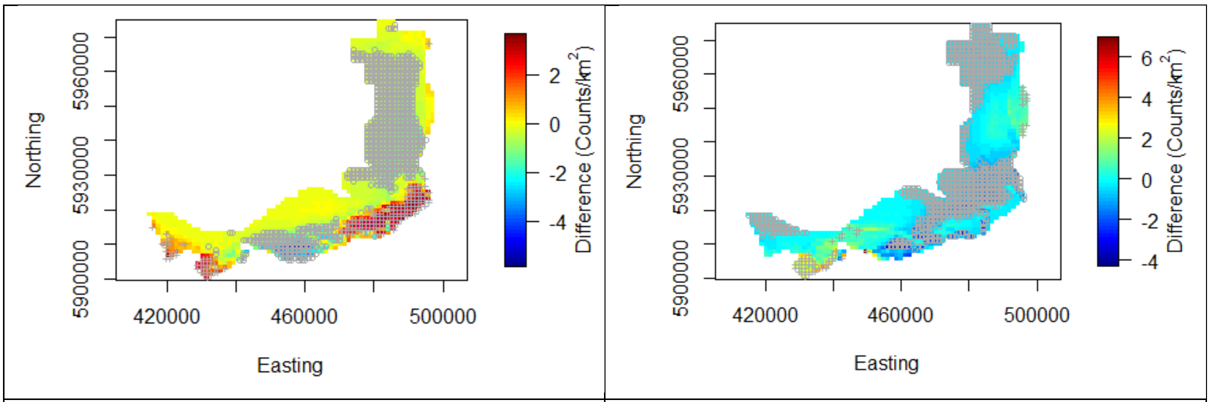



**Figure 75 Change in population size of red-throated divers in Liverpool Bay SPA from 2011 to 2018.**

Density surface maps of the difference in abundance of red-throated divers between 2011 and 2015 and 2018 showed that distributions were likely dynamic from year to year (Figure 76). In 2015 there were statistically significantly fewer birds in two areas of the SPA when compared to 2011: the areas in the north of the SPA, and the area off the north Wales coast from Prestatyn to Llandudno. In addition, there were statistically significantly more red-throated divers in two areas: the very southwest of the SPA (east coast of Anglesey), and the very southeast of the SPA (off the Dee and Mersey estuaries). However, the difference between 2011 and 2018 was a little more complex. Statistically significantly fewer birds occurred across the north, southeast and far west of the SPA and more birds in Menai Straight and Great Orme, and coastal waters of the Ribble & Alt estuary.




| February 2011 – February 2015 | February 2011 – February 2018 |
|---|---|

**Figure 76 Geo-referenced estimated differences in average numbers between February 2011 and February 2015 (left) and February 2018 (right). The values represent averaged predicted differences for February for a 1km x 1km grid cell and positive significant changes are represented using "+" while significant negative change is represented using "O" (from Humphries et al. 2018).**

Humphries et al. (2018) reported that the abundance of red-throated divers within the 4 km buffer around the Burbo Bank extension was 14% of the SPA total in February 2011 and this was 18% in 2015, but only 6% in 2018. The report concludes that there was a displacement effect away from the Burbo Bank extension apparent in the 2017/18 data. However, these results also suggest that the changes in spatial distributions and responses to offshore wind farms are more complex than simple displacement effects in all directions from wind farms.

The primary habitats of value to wintering red-throated divers in the Liverpool Bay SPA are shallow sandbanks up to 20 m deep. Therefore, additional protection of these habitats would potentially result in better overwinter survival of the current red-throated diver population, thus maintaining it. In addition, the aim of this measure would also be to ensure population growth within the remaining protected habitat within the SPA.

The area of habitat around the three windfarms within the SPA and in English waters was estimated across three levels of buffer zone around each of the wind farm clusters (Table 172). These levels were based upon current predicted displacement levels around offshore wind farms. Heinänen et al. (2020) showed strong displacement at 5 km from wind farm in the German Bight and statistically significant difference in red-throated diver abundance as much as 12 km from offshore wind farms. However, responses of red-throated divers to offshore wind farms are not simple and displacement does not occur in isolation from other pressures and preferences. Different responses to impact should be expected between areas where habitat is abundant and carrying capacity is high, compared with areas where habitat is more limited and carry capacity is much lower. It is also important to consider the pattern of preferred habitat and location of offshore wind farms when considering displacement distances. If birds have a preference for shallow sand banks and are displaced from them by offshore wind farms they may move to the next available preferred habitat. The distance that this habitat is from the wind farm may appear as a displacement distance, but it may be that birds would not move as far if preferred habitat was closer. Consequently, the distances used in assessing potential habitat loss here should be considered precautionary and future monitoring and research of displacement of red-throated divers is needed to resolve the level and pattern of displacement that actually occurs within the Liverpool Bay SPA.

**Table 172 Potential habitat loss from Liverpool SPA from existing offshore wind farms within the SPA and three possible buffer distances.**

| Buffer size | 0 km | 8 km | 10 km | 12 km |
|---|---|---|---|---|
| Area ($km^2$/%) of potential habitat loss | 68.9/2.7% | 696.6/27.6% | 893.8/35.5% | 1089.3/43.2% |
| Remaining available habitat inside OTE SPA ($km^2$/%) | 2452.2/62.6% | 1824.5/27.6% | 1627.3/35.5% | 1431.9/43.2% |
| Ratio of habitat loss to remaining | 1:35.6 | 1:2.6 | 1:1.8 | 1:1.3 |

It is apparent from the bathymetry showing preferred habitat depths (Figure 77) that preferred water depths are not limiting within the SPA, at least within English waters. The only bathymetry data available for this study were those within English waters.

The ratio of habitat loss to habitat strictly protected for the Voordelta marine reserve was 1:10. It is apparent that the remaining habitat inside the Liverpool SPA is insufficient to reach a ratio of much more than 1:2 for the 8 km buffer, and a 1:1 ratio for 10 and 12 km buffers. Compensation through improvement in habitat would therefore need to include habitat improvements in other SPAs for wintering red-throated divers. Other SPAs in the UK designated for their wintering red-throated diver populations include the OTE SPA, Greater Wash SPA, Firth of Forth SPA, Outer Firth of Forth and St Andrews Bay Complex SPA and Solway Firth SPA. Most of these other SPAs are also subject to displacement pressures from offshore wind farms, or other pressures and impacts. Consequently, the application of a marine reserve approach to compensation may be better considered at a strategic level and considered across all of the SPAs for wintering red-throated diver subject to pressures from offshore wind farm developments, including the OTE SPA (Section 9).

Given the combined issues of uncertainty in the displacement distance, the potential for complex effects of offshore wind farms on the spatial distribution of red-throated divers in the Liverpool Bay SPA and the remaining available areas within the SPA where a reserve may be created a further option may be to provide marine reserves in appropriate locations within the SPA and then add further marine reserves within the footprints of the wind farms inside the SPA as these are decommissioned. Assuming that further consents to re-power or extend the life of the existing wind farms are not granted, this approach could assure the integrity of winter red-throated divers in UK SPA and correct the negative effects of the existing projects on the distribution of the species within the Liverpool Bay SPA.

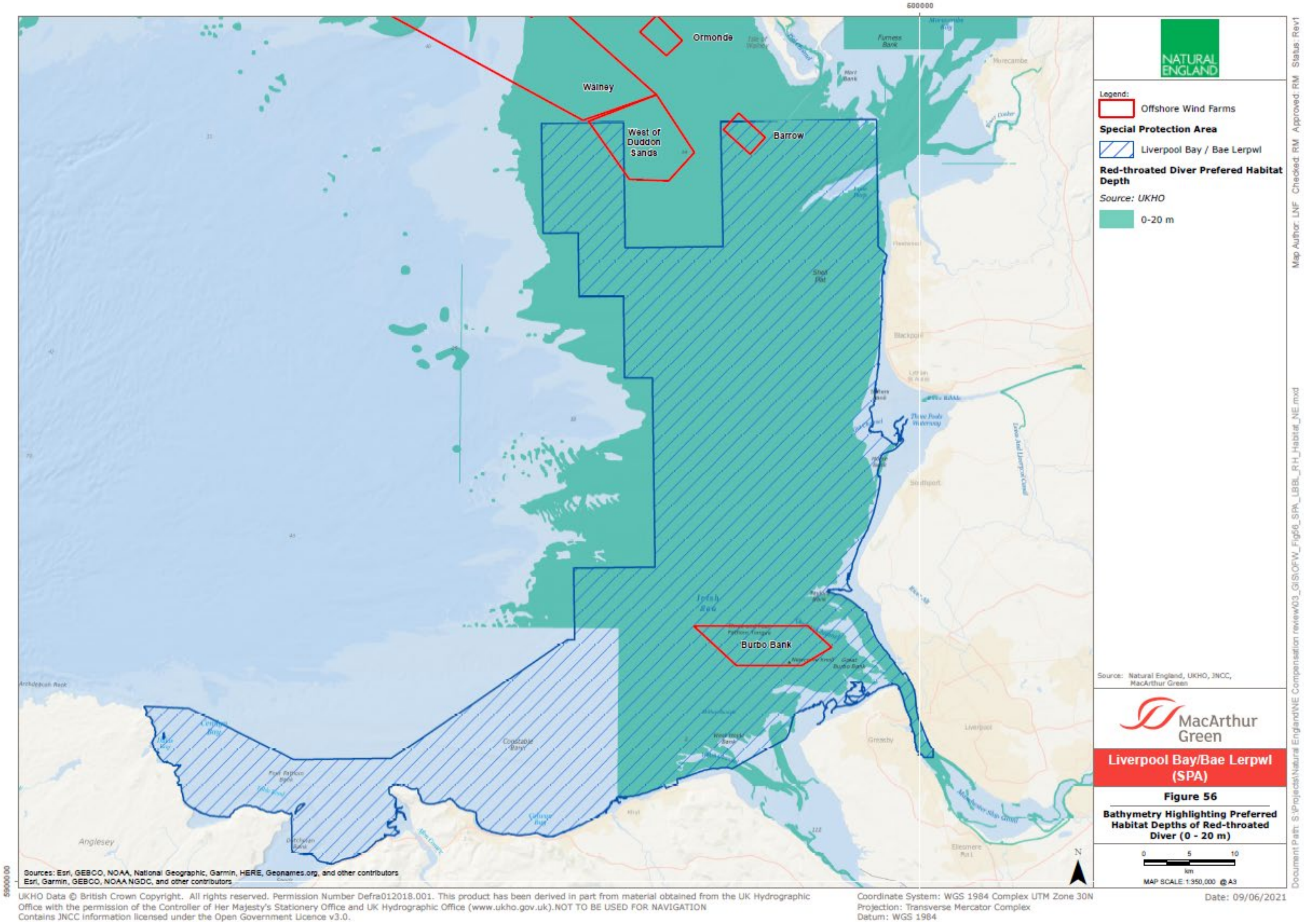



**Figure 77 Bathymetry of the Liverpool Bay SPA highlighting the preferred depth of red-throated diver (0 – 20 m).**

#### *10.7.1.3 Removal of existing wind farms*

The footprint of the projects within the English part of the SPA is 69 km² with a combined area of 1,089 km² within the Liverpool Bay SPA of a 12 km buffer around the wind farms (this takes account of the overlap between the 12 km buffers around the Walney wind farms, the Barrow wind farm and Ormonde wind farm). Thus, a large proportion of the Liverpool Bay SPA would have this pressure removed if these wind farms were removed. It is also important to note that the wind farms are not a permanent feature and will be decommissioned in the future. Assuming that consents for the projects inside the Liverpool Bay SPA were for 25 years the existing projects will begin to be decommissioned between 2031 to 2043 (Table 173), though the decommissioning process is likely to take more than one year. Once removed it is unknown how long it might take for the area to be used by red-throated divers again.

**Table 173 Commissioning and decommissioning years of offshore wind farms within the Liverpool Bay SPA.**

| Project | Commissioning year | Decommissioning year |
|---|---|---|
| Barrow | 2006 | 2031 |
| Burbo Bank | 2007 | 2032 |
| Walney | 2010 | 2035 |
| Ormonde | 2012 | 2037 |
| Burbo Bank extension | 2017 | 2042 |
| Walney extension | 2018 | 2043 |

Analysis to determine whether there was a statistically significant change in the distribution of red-throated divers following wind farm construction was beyond the scope of this study and would be important in determining the use of the areas within these wind farms and footprints prior to recommending this as a compensation measure.

#### *10.7.1.4 Compensation ratios*

As with the OTE SPA (see section 9.7.1.4), it would not be possible to create marine reserves within the Liverpool Bay SPA based on a 12 km displacement buffer and a 1:10 compensation ratio. It would therefore be necessary to include marine reserves across multiple SPAs.

### 10.7.2 NW England and Wales

The only SPA in the NW England and Wales BDMPS is the Liverpool Bay SPA, so further compensation is not possible in this spatial area.

### 10.7.3 All other BDMPS spatial areas in the UK

Within the remaining BDMPS spatial areas there are five other SPAs designated for non-breeding red-throated diver populations: Moray Firth SPA, Outer Firth of Forth and St Andrews Bay Complex, Solway Firth, Greater Wash and Outer Thames Estuary. This covers a total area of 13,301 km² (Table 174).

**Table 174 SPAs in all other BDMPS spatial areas in the UK.**

| SPA | Area ($km^2$) |
|---|---|
| Moray Firth SPA | 1,762 |
| Outer Firth of Forth and St Andrews Bay Complex | 2,721 |
| Solway Firth | 1,357 |
| Greater Wash | 3,536 |
| Outer Thames Estuary | 3,925 |
| Total area | 13,301 |

*10.7.3.1 Reducing disturbance by vessel activity*

There is generally similar shipping activity within the remaining SPAs in all other BDMPS spatial areas in the UK than in the Liverpool Bay SPA, based on the MMO analysis of shipping density in 2012 (Figure 73). Among the remaining SPAs there appears to be higher traffic levels in the OTE SPA and Greater Wash SPA than the other sites, with much lower levels in the Solway Firth SPA. Much of this shipping density appears to be within shipping lanes in and out of existing ports and harbours, such as the Port of London, Leith, Hull and Inverness. However, it appears that much of the high-density traffic in these SPAs occurs in the summer (Figure 70). Scope for compensation through reducing or concentrating commercial vessel activity within these other SPAs may be limited.

Amongst this commercial shipping, there would appear to be a lot of potential of recreational vessel activity (Figure 66). However, this activity is typically concentrated in the summer months, when red-throated divers are absent from the SPAs. Compensation measures may be limited here too.

*10.7.3.2 Creating a marine reserve*

If compensation to habitat loss from Liverpool Bay SPA was applied in the remaining SPAs there would need to be between 69 $km^2$ and 1,089 $km^2$ of the area within the SPAs managed as a strict marine reserve. This is between 0.5% and 8.2% of the total area within the SPAs and only for compensating at a 1:1 ratio. As discussed above a much larger compensation ratio was applied in the Voordelta case (1:10). At this ratio, between 5% and 82% of the SPAs area would need to be in strict marine reserve to provide compensation. Clearly, as with the OTE SPA, the high level of uncertainty in the level of habitat loss through displacement in the Liverpool Bay SPA is very important in deciding the potential for compensation in the remaining SPAs. It is also important to take account of the existing pressures on red-throated divers in the remaining SPAs, particularly from existing offshore wind farms and likely also from shipping. Three of the SPAs have existing wind farms within them: OTE SPA (see Section 9.7.1.2), Greater Wash (see Section 9.7.2.2) and the Solway Firth. As discussed above, it seems that there is little pressure on red-throated divers in the Solway Firth from Robin Rigg offshore wind farm, as the site was not used by important numbers of red-throated divers before construction (Walls et al. 2013). Compensation measures for impacts within Liverpool Bay SPA are also unlikely to be applicable within the OTE SPA while that SPA is also subject to sufficient impacts for an existing adverse effect on site integrity to have been concluded.

#### 10.7.3.3 *Removal of existing wind farms*

As described above, only three SPAs in all other BDMPS spatial areas have offshore wind farms within them: OTE SPA, Greater Wash SPA and Solway Firth SPA. The effects of removal of offshore wind farms within OTE SPA are described in Section 9.7.1.3 and within the Greater Wash SPA in Section 9.7.2.3. Since the Robin Rigg wind farm did not exert a pressure on the red-throated diver population in the Solway Firth SPA removal of this wind farm would not provide any compensation.

## 10.8 Assessment of confidence

The assessment of confidence in the metrics used to determine potential compensation measures for Liverpool Bay SPA were identical to those of the OTE SPA (see Section 9.8).

## 10.9 Future monitoring and adaptive management

It is apparent that further analysis is needed to determine the patterns of displacement of red-throated divers in the Liverpool Bay SPA. The data exists to statistically compare the abundance of birds in a grid across the SPA before and after construction of the wind farms. This could be a more effective approach than the concentric rings that have been used elsewhere (e.g., Mendel et al. 2020) as that approach assumes an equal effect in all directions from the wind farm. This would be important in determining whether future compensation measures are effective in improving the distribution of birds within the site. It is assumed that the aim would be to return the use of the site to the distribution seen within the SPA prior to construction of the wind farms, so future monitoring would need to collect spatial data on the distribution of divers across the SPA and these would need to be compared with the aerial survey data collected prior to the construction of the wind farms.

As wind farms are removed (either as part of a compensation measure or due to decommissioning) further digital aerial surveys and spatial analysis would be needed to show whether birds return to these areas.

It is apparent from the monitoring programme for the Voordelta SPA that a marine reserves approach to compensation would benefit from a similar approach to monitoring focused on the improvement in quality of habitat for red-throated divers. This may therefore require better understanding of the habitat requirements of red-throated divers in winter in the Liverpool Bay SPA, so may require forage fish and benthic sampling within areas already preferred and avoided by divers in the SPA. Adding the footprints of existing wind farms inside and adjacent to the SPA to these marine reserves as they are decommissioned could provide adequate compensation without the need to a strategic level assessment.

### 10.9.1 Future research

The needs for future research to inform compensation in Liverpool Bay SPA were identical to those in the OTE SPA (see Section 9.9.1).

## 10.10 Summary

The review found that the only compensation measure recommended by Furness et al. (2013) which was still potentially relevant for red-throated divers at Liverpool Bay SPA was reducing disturbance by vessel activity. However, the seasonal pattern of use of the SPA by shipping suggests that this measure could have relatively little benefit for wintering red-throated divers. Since Furness et al. (2013) new evidence was found on an existing compensation measure for a similar SPA in the Netherlands. The Voordelta SPA was impacted through habitat loss caused by the expansion of the Port of Rotterdam. The SPA was designated for multiple species, including wintering red-throated diver. The compensation measures suitable for divers included creation of marine reserves within the SPA. Within these reserves seasonal exclusion of all vessels and low flying aircraft reduced disturbance. Exclusion of bottom trawling fisheries improved habitat quality. Finally, since the presence of operational wind farms within the Liverpool Bay SPA causes the adverse effect on site integrity, their removal would represent adequate compensation. Whether this is practical is beyond the remit of this report, but its effectiveness is highly likely to be successful.

The assessment of the area within the Liverpool Bay SPA showed that there was insufficient area beyond the buffers around the existing wind farms to provide marine reserves at a ratio of 1:10. This was the ratio applied in the Voordelta case. This and the presence of existing pressures from other offshore wind farms in UK waters resulted in the recommendation for further work to undertake a strategic level assessment of the available suitable habitat for marine reserves within all SPAs that include wintering red-throated divers as a feature.

Confidence in the marine reserves method for compensation of red-throated divers was assessed as low to high, depending on the size of the area, the assessment of other compensation measures was difficult to apply using the approach applied to other SPA features in this study. Confidence in the marine reserves approach would be achieved through a very high compensation ratio (1:10) combined with adaptive management. Removal of existing wind farms should have complete confidence, but only if the areas currently occupied by offshore wind farms was previously used by red-throated divers. It was recommended that further analysis is completed to determine whether these sites were important for the Liverpool Bay SPA red-throated diver prior to construction of the wind farms.

# 11 SUMMARY OF COMPENSATION APPROACHES

The overall aim of compensating impacts to the population of a qualifying feature of an SPA is to ensure the integrity of the UK SPA network is maintained. While it is not an absolute requirement that the impacted SPA is directly compensated, previous compensation measures, for example those involving habitat creation through managed realignment schemes, have tended to be applied in areas directly adjacent to the impacted site so as to be of direct benefit to it.

For each SPA qualifying feature this assessment has tested whether measures across a range of levels (low, medium and high) could compensate for three predicted impact scenarios based on low (1% of baseline mortality), medium and high levels of impact. The medium and high impact levels were derived on a pro-rata basis from in-combination impact assessments linked to the capacity of currently installed, consented and in planning developments i.e. 26GW. The medium impacts equated to mortality levels estimated on the basis of the planned capacity from Round 4 (7GW). The high impacts equated to mortality levels estimated on the basis of an additional 74Gw required to meet the ambition of 100GW by 2050.

For each SPA qualifying feature summaries of their general conservation status, citation population size and conservation objectives have highlighted a range of conservation status from those with significant population declines (e.g. breeding lesser black-backed gull at Alde-Ore Estuary SPA) to those with strongly increasing populations (e.g. gannet at FFC SPA). Thus, predicted impacts, and compensation for these, have varying levels of population consequence.

For each SPA qualifying feature, the key biological questions that would need to be answered in order to assess the potential application of compensation to SPA populations were formulated based on the combination of the nature of the impact on the population and nature of the effect of the compensation measure on population demographics.

For each SPA qualifying feature available literature was reviewed to assess whether there was important additional information available since the review of compensation by Furness et al. (2013). For each of the key biological questions asked, answers were provided from the reviewed literature, where possible. For many of the features there was additional supporting published information to help inform the assessment of compensation from predicted impacts from offshore wind farm developments. For some species there was little or no additional information found. However, no new published information was found to contradict the findings of Furness et al. (2013).

Population level assessment for each SPA qualifying feature was completed using the Seabird PVA Tool for low, medium and high impact scenarios, wherever this was possible. All populations were assumed to be closed (i.e. no immigration or emigration), which is not true. So, it is important that this assumption is carefully considered when interpreting the effects of compensation on the specific SPA qualifying features assessed in this report. For each feature, where possible, low, medium and high compensation scenarios were quantified and comparisons made between population projections from combined impact and compensation scenarios with baseline (no impact, no compensation) conditions. Comparisons were made using counterfactuals of population size and growth rate, though emphasis was based on counterfactuals of growth rate as the models were all density independent.

For compensation methods using invasive terrestrial mammal eradication from offshore islands PVA was not used. For each SPA qualifying feature, where this was a recommended compensation method, islands with these species present (either in SPA or not) and invasive mammals present were identified. Comparisons were made between the predicted impact scenarios and the change in population size on the island between either the citation population or the Seabird 2000 count and the most recent count.

An assessment of the confidence in both the process and recommended compensation measure(s) was completed for each SPA qualifying feature. For each combination of impact scenario and compensation scenario that was assessed using the Seabird PVA Tool, confidence in the success of the proposed the compensation measure was scored as low, medium or high (Table 175).

**Table 175 Summary of confidence in recommended compensation methods for each SPA qualifying feature.**

| Qualifying feature | SPA | Compensation method | Compensation level | Impact level | | |
|---|---|---|---|---|---|---|
| | | | | Low | Medium | High |
| Kittiwake | FFC | Sandeel closure | Low | MEDIUM | MEDIUM | MEDIUM |
| | | | Medium | HIGH | HIGH | HIGH |
| | | | High | HIGH | HIGH | HIGH |
| | | Artificial colonies | Very low | MEDIUM | LOW | LOW |
| | | | Low | HIGH | MEDIUM | MEDIUM |
| | | | Medium | HIGH | MEDIUM | MEDIUM |
| | | | High | HIGH | MEDIUM | MEDIUM |
| Gannet | FFC | Fisheries bycatch reduction | Low | MEDIUM | MEDIUM | LOW |
| | | | Medium | MEDIUM | MEDIUM | LOW |
| | | | High | HIGH | MEDIUM | LOW |
| | | End chick harvest | Low | HIGH | HIGH | LOW |
| | | | Medium | HIGH | HIGH | LOW |
| | | | High | HIGH | HIGH | LOW |
| Guillemot | FFC | Sandeel or sprat fisheries closure | Low | HIGH | HIGH | LOW |
| | | | Medium | HIGH | HIGH | LOW |
| | | | High | HIGH | HIGH | LOW |
| | | Eradication of rats and other invasive mammal predators | Low | LOW | LOW | LOW |
| | | | Medium | LOW | LOW | LOW |
| | | | High | LOW | LOW | LOW |
| Razorbill | FFC | Sandeel or sprat fisheries closure | Low | LOW | LOW | LOW |
| | | | Medium | LOW | LOW | LOW |
| | | | High | LOW | LOW | LOW |
| | | | Low | MEDIUM | MEDIUM | LOW |

MacArthur Green

| Qualifying feature | SPA | Compensation method | Compensation level | Impact level | | |
|---|---|---|---|---|---|---|
| | | | | Low | Medium | High |
| | | Eradication of rats and other invasive mammal predators | Medium | MEDIUM | MEDIUM | LOW |
| | | | High | MEDIUM | MEDIUM | LOW |
| Puffin | FFC | Sandeel or sprat fisheries closure improving productivity | Low | HIGH | MEDIUM | LOW |
| | | | Medium | HIGH | HIGH | LOW |
| | | | High | HIGH | HIGH | LOW |
| | | Sandeel or sprat fisheries closure improving adult survival | Low | HIGH | HIGH | LOW |
| | | | Medium | HIGH | HIGH | MEDIUM |
| | | | High | HIGH | HIGH | HIGH |
| | | Eradication of rats and other invasive mammal predators | Low | MEDIUM | MEDIUM | LOW |
| | | | Medium | MEDIUM | MEDIUM | LOW |
| | | | High | MEDIUM | MEDIUM | LOW |
| Sandwich tern | NNC | Predator control measures | Low | HIGH | HIGH | MEDIUM |
| | | | Medium | HIGH | HIGH | MEDIUM |
| | | | High | HIGH | HIGH | MEDIUM |
| | | Sandeel or sprat fisheries closure increasing productivity | Low | HIGH | HIGH | MEDIUM |
| | | | Medium | HIGH | HIGH | MEDIUM |
| | | | High | HIGH | HIGH | HIGH |
| Lesser black-backed gull | AOE | Predator control measures | Low | HIGH | HIGH | HIGH |
| | | | Medium | HIGH | HIGH | HIGH |
| | | | High | HIGH | HIGH | HIGH |
| Red-throated diver | OTE | Marine reserve | n/a | n/a | n/a | n/a |
| | | Vessel movements | n/a | LOW | LOW | LOW |
| | | Wind farm removal | n/a | HIGH | HIGH | HIGH |
| | Liverpool Bay | Marine reserve | n/a | n/a | n/a | n/a |
| | | Vessel movements | n/a | LOW | LOW | LOW |
| | | Wind farm removal | n/a | HIGH | HIGH | HIGH |

There were some species-specific differences in degrees of confidence in compensation measures, some of which were due to differing availability of evidence. For example, confidence in sandeel closure was higher for guillemot and puffin than razorbill. This was largely due to the differing availability of evidence to support the effects of this measure on each species, rather than an expectation that populations of the different species would respond differently to this measure. There were also species differences across the same measure that were due to differences in their ecology. For example, rat eradication had a much lower confidence for guillemot than for razorbill or puffin. This was partly due to available evidence (one study showed guillemot responding favourably to rat eradication, but another did not) but also due to the different nesting habitats for these species. Guillemots nest on open cliff ledges that are largely inaccessible to rats, while razorbills and puffins nest in boulders or burrows, which are much more accessible to rats. For red-throated divers the recommended compensation measures were to provide marine reserves to improve habitat quality. The confidence in this approach is dependent on a suitably high compensation ratio and appropriate adaptive management, so the approach to confidence assessment applied to other SPA features was not appropriate for red-throated divers.

Overall, compensation ratios had a large effect on the predicted efficacy of recommended compensation measures. The scale of this effect varied considerably with the ratio applied, or with the impact scenario considered. The scaling of impact scenarios, from low to medium to high, was not linear. High impact scenarios were considerably larger than low or medium impact scenarios (in most cases). Thus, applying compensation ratios to the high impact scenario had a larger effect than to the low or medium impact scenarios and mostly showed that these impacts were far larger than measures could compensate for.

For each recommended compensation measure, recommended future monitoring and adaptive management approaches were provided. Monitoring was recommended to determine whether the compensation measure was directly successful (e.g sandeel stocks increase, or rats were eradicated from an offshore island) and whether it was successful in affecting demographic process and, ultimately, SPA population size or distribution within the site for red-throated diver. Where monitoring proves less successful than required, adaptive management processes were suggested. In general, monitoring methods were very similar to existing productivity and population size monitoring. For some SPA qualifying features, it was also suggested that either return rates (as a proxy for adult survival) or survival modelling from ringing recoveries or resightings were monitored. It was recommended that monitoring should be annual initially but potentially reduced in frequency and/or intensity if monitoring can show that compensation has been successful. Finally, it was recommended that monitoring would need to occur, at least periodically, across the period that compensation was required.

This work has found that various levels of impact on seabird SPAs that might arise as a consequence of the construction and operation of offshore wind farms could be compensated for, with varying degrees of confidence. In general, confidence in compensation measures was highest for low impact scenarios and lowest for high impact scenarios. It is acknowledged that the medium and high impact scenarios considered are extreme (plus 7GW and 74GW respectively) and probably pessimistic, being derived on a pro-rata basis from current predicted impact levels per GW. This takes no account of potential reductions in per GW impacts that might result from future increases in turbine height and deployment in deeper water further from shore and seabird colonies. These scenarios have been included largely to raise awareness of: i) the potential scale of cumulative impact on these species/SPAs (or seabirds more generally) if energy policy is successful in progressing towards delivery of current targets and ii) the difficulty there could be in finding ways to compensate for these potentially large impacts on (some of) our protected seabird populations.

At the higher levels of impact considered here there were few compensation measures that were potentially effective. For several species it is clear that the most effective compensation measure would be to close UK waters to sandeel, and sprat, fisheries. This would appear to be more likely than other measures to be able to compensate for relatively high levels of impact, although with varying degrees of confidence. It is important to note that this study only looked at the benefits of compensation to the focal species and SPAs individually. Closure of these fisheries would be highly likely to benefit multiple species at many sites, including other SPAs. The overall level of compensation achieved at the SPA network scale through this measure has the greatest chance of being sufficient to provide strategic level compensation for the potential impacts on seabirds from the offshore wind industry now and in the future.

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