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Report to Crown Estate Scotland and SOWEC: HRA Derogation Scope B - Review of seabird strategic compensation options

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CONTENTS

1 SUMMARY....1
2 INTRODUCTION....3
3 METHODS....6
4 SPECIES ACCOUNTS....8
  4.1 Great northern diver *Gavia immer*....8
  4.2 Red-throated diver *Gavia stellata*....13
  4.3 Fulmar *Fulmarus glacialis*....18
  4.4 Manx shearwater *Puffinus puffinus*....23
  4.5 European storm-petrel *Hydrobates pelagicus*....28
  4.6 Leach's petrel *Oceanodroma leucorhoa* (*Hydrobates leucorhous* in North America)....33
  4.7 Gannet *Morus bassanus*....36
  4.8 Arctic skua *Stercorarius parasiticus*....42
  4.9 Great skua *Stercorarius skua*....47
  4.10 Lesser black-backed gull *Larus fuscus*....52
  4.11 Herring gull *Larus argentatus*....55
  4.12 Great black-backed gull *Larus marinus*....58
  4.13 Kittiwake *Rissa tridactyla*....61
  4.14 Sandwich tern *Sterna sandvicensis*....76
  4.15 Common tern *Sterna hirundo*....80
  4.16 Arctic tern *Sterna paradisaea*....84
  4.17 Common guillemot *Uria aalge*....88
  4.18 Razorbill *Alca torda*....95
  4.19 Puffin *Fratercula arctica*....99
5 STRATEGIC COMPENSATION APPROACHES....104
  5.1 No-take zones for forage fish....104
  5.2 Eradication of invasive mammal predators....107
  5.3 Artificial colonies for kittiwakes....108
6 RECOMMENDATIONS....109
7 REFERENCES....110
8 ANNEX 1. STATUS OF SPA POPULATIONS WITH SEABIRDS AS BREEDING FEATURES....129
  8.1 Great northern diver....129
  8.2 Red-throated diver....129

8.3 Fulmar 130
8.4 Manx shearwater 133
8.5 European storm-petrel 134
8.6 Leach’s petrel 136
8.7 Gannet 136
8.8 Arctic skua 137
8.9 Great skua 138
8.10 Lesser black-backed gull 139
8.11 Herring gull 140
8.12 Great black-backed gull 141
8.13 Kittiwake 142
8.14 Sandwich tern 145
8.15 Common tern 146
8.16 Arctic tern 149
8.17 Common guillemot 151
8.18 Razorbill 156
8.19 Puffin 159

1 SUMMARY

This evidence review was commissioned by Crown Estate Scotland on behalf of the Scottish Offshore Wind Energy Council (SOWEC). We reviewed possible options for compensation under The Birds and Habitats Directives for seabird populations, with emphasis on Scotland. Seabird species included in this review are: great northern diver, red-throated diver, fulmar, Manx shearwater, European storm-petrel, Leach's petrel, gannet, Arctic skua, great skua, lesser black-backed gull, herring gull, great black-backed gull, kittiwake, Sandwich tern, common tern, Arctic tern, common guillemot, razorbill, and puffin (scientific names head each species account). The review considers each species in turn, outlining the conservation status of the species nationally and at each SPA where the species is a feature (including where the species is a named component of the seabird assemblage), outlines the demographic values for the species available in the literature, briefly outlines the species' ecology and threats to population, then considers potential compensation measures (starting from information in the Defra review (Furness et al. 2013) as a baseline, updated where relevant by more recent literature and evidence). Particular emphasis is put on kittiwake because additional in-combination impacts on that species are highly likely to be considered to affect integrity. The review then considers the extent to which a strategic approach to compensation measures may be most appropriate and most effective in terms of conservation gain. Annex 1 contains details, for each species, of each SPA in the UK where the species is a breeding species feature, with details of its most recent published breeding population census and its most recently assessed conservation status (up to December 2020).

There is strong evidence that seabird species are highly dependent for breeding success and for survival on healthy stocks of forage fish, especially sandeels *Ammodytes marinus*. Therefore, several seabird species would benefit from measures that allow stocks of forage fish (sandeels and sprats *Sprattus sprattus*) to recover from depletion caused by fishing on those stocks. Modelling sandeel stock dynamics indicates that reducing fishing mortality on those stocks would lead to some recovery of stock biomass. Developing forage fish no-take zones would most effectively be done by UK Government action. Because many Scottish seabirds migrate after breeding, and some move to the southern North Sea, management actions in the southern North Sea that enhance overwinter survival of seabirds may be appropriate as compensation for impacts on some Scottish seabird populations, in addition to measures that assist recovery of forage fish stocks in Scottish waters.

There is strong evidence that many seabird species are adversely affected by invasive non-native mammals that have colonised seabird islands. There is also strong evidence that eradicating these mammals leads to rapid recovery of affected seabird populations. This therefore represents a highly effective means of compensation that would be relevant for a particular (but limited) set of seabird species.

Even where a potential compensation measure might be effective only for a single species, that measure may best be delivered by strategic action co-ordinated by Scottish Government in some cases, rather than by scramble competition among developers to achieve limited compensation as required for individual developments. For example, constructing artificial nest sites for kittiwakes might be suitable as a compensation mechanism, at least in parts of England, but a strategic approach could ensure that coherent and optimal measures were taken to achieve this, in a way that may be difficult for any individual developer to achieve. The measures discussed may also be

useful to consider in terms of the developing UK Seabird Conservation Strategy and the Scottish Government's Seabird Conservation Strategy, and if proposed as compensation, should be fully integrated with those strategies, once they are published.

Finally, we make six recommendations:

- Given the scale and nature of the compensation likely to be required, we recommend a strategic approach to compensation for seabirds coordinated by Government, in order to optimize the compensation and to align closely with UK Government's Seabird Conservation Strategy, and with Scottish Government's Seabird Conservation Strategy and National Marine Plan, with developer contributions made in line with this approach.
- Where compensation may be project-based rather than strategic, we advise close alignment with UK Government's and Scottish Government's Seabird Conservation Strategies, and consideration of effects of climate change over the time scale of the measure, as well as longer term consideration on predicted changes in seabird distributions as a result of climate change as that may influence which compensation measures would be most appropriate.
- We recommend that closure of UK waters to directed fishing for sandeels should be a strategic approach to compensation, noting also that over the longer term, such an approach would benefit UK fishermen in terms of increasing condition of predatory fish such as cod, haddock and whiting.
- We recommend eradicating invasive mammal populations from prioritized seabird islands should be a strategic approach to compensation, relevant for a particular (but limited) set of seabird species, and that this approach should align with objectives of the UK Government's and Scottish Government's Seabird Conservation Strategies, but note also that sustained long-term support for biosecurity measures would be essential to prevent recolonisation by invasive mammals.
- We recommend that wider impacts of strategic compensation options should be considered before they are endorsed, ideally via a strategic framework overseen by Government.
- Given that significant conservation gains could be achieved for individual seabird species through a variety of species-specific management measures outlined in this report, we recommend that in cases where these may not be required as compensation, they should be considered as potential management under the UK Government's and Scottish Government's Seabird Conservation Strategies to enhance the conservation status of those seabirds.

2 INTRODUCTION

Where the competent authority cannot conclude that there is no adverse effect on site integrity (AESI) from a plan or project, alone or in-combination with other reasonably foreseeable plans or projects, the plan or project can only proceed where there is an imperative reason of overriding public interest (IROPI), there must be no feasible alternative solutions to the plan or project which are less damaging to the affected European site(s). Finally, it is necessary for compensation to the Natura 2000 network to be provided to maintain the favourable conservation status of the feature(s) being potentially damaged. The key objective of this review is to outline the evidence underpinning strategic level forms of compensation for impacts to breeding seabirds in SPAs that may arise from potential offshore wind farm projects in Scottish waters at ScotWind sites. Through application of the HRA processes, seven of the fifteen Plan Options (POs) in the Sectoral Marine Plan (SMP) are subject to additional ornithological constraints, including some of the few sites that are suitable for fixed-foundation development. Some of these ornithological constraints are caused by predicted in-combination impacts on populations of widespread seabirds, such as kittiwake *Rissa tridactyla* and gannet *Morus bassanus*. An increase in estimated in-combination impacts arises with increasing development, so there is a high risk of this becoming a constraint. However, some new ornithological constraints may also arise because some ScotWind areas are in locations where there has not previously been offshore wind development, and where there are novel seabird issues. The need to be precautionary where there is a lack of evidence regarding novel issues (such as displacement of shearwaters, behavioural responses of storm-petrels, or collision risk for skuas) may result in these species becoming constraints in areas with connectivity to SPAs.

In the southern North Sea, and in SMP areas NE2-7, E1-3, it is in-combination impacts of multiple offshore wind farm developments on SPA populations that represents the greatest consenting risk to further development in those areas. In-combination impacts of collision on populations of abundant seabirds (such as kittiwakes, gannets, herring gulls *Larus argentatus*, lesser black-backed gulls *Larus fuscus* and great black-backed gulls *Larus marinus*), and in-combination impacts of displacement on populations of abundant seabirds (such as gannets, common guillemots *Uria aalge*, razorbills *Alca torda*, Atlantic puffins *Fratercula arctica*) generate particular consenting risk.

In SMP areas W1, N1-4, and NE1, in-combination impacts are not yet likely to be the main constraint during the early development of the industry in those new areas. Instead, it is possible that uncertain impacts on novel species (for this industry) will be a concern because of a lack of evidence and a lack of experience in considering these novel species' interactions. The seabird species that seem most likely to represent novel concerns are Manx shearwater *Puffinus puffinus*, European storm-petrel *Hydrobates pelagicus*, Leach's petrel *Oceanodroma leucorhoa*, Arctic skua *Stercorarius parasiticus*, great skua *Stercorarius skua*, and possibly great northern diver *Gavia immer*. In Scotland, in contrast to all the other species that are features of Special Protection Areas (SPAs) mainly or only for their breeding populations, great northern diver is a feature of SPAs only for the nonbreeding population. In this case, "nonbreeding" birds are birds that breed overseas but spend the autumn, winter and spring in UK waters. However, "nonbreeding" may also be used as a term for seabirds that are features of SPAs designated for marine areas away from seabird colonies where there are aggregations of seabirds at sea. In the latter case, those aggregations may be in winter (in which case the birds are clearly "nonbreeding") or may be in summer (in which case the birds may include a mixture of immature birds that are "nonbreeding" and also some adults that are breeding at colonies distant from the marine aggregation site. This causes some confusion in the term "nonbreeding", and both uses of the term may appear in this report depending on the site type.

Although red-throated diver *Gavia stellata* has been a major focus of impact assessments of some offshore wind developments in parts of English waters where this species overwinters in large numbers and shows strong displacement by offshore wind farms and by vessel traffic, this species has been less prominent in impact assessments in Scottish waters (where overwinter densities of red-throated divers tend to be lower, but there is greater connectivity to red-throated diver breeding areas).

By agreement with CES and SOWEC, some species that are features of SPAs in the UK were not included in the scope of this project because they were considered unlikely to be constraints on offshore wind development in Scottish waters, at least within the current ScotWind leasing process.

The measures discussed here in the context of possible compensation for impacts of renewables developments may also be useful to consider in terms of the developing UK Seabird Conservation Strategy and the Scottish Government's Seabird Conservation Strategy. Both of those strategies are apparently still at the early development stage, but little information on these strategies is in the public domain at present. The Scottish Government has expressed a desire to develop a Scottish Seabird Conservation Strategy with particular reference to adaptation to impacts of climate change (Scottish Government 2019). However, two years on from that announcement, details of that strategy have yet to be published. The UK Government has also announced development of a Seabird Conservation Strategy for England. The UK Environment Minister, Rebecca Pow, announced on 16/01/2020 "This year we will be publishing a comprehensive Seabird Conservation Strategy for England" (UK Government 2020). However, no publication has been produced. On 19/01/2021 the same minister said "I have commissioned a comprehensive review of the different pressures our seabirds are facing" (UK Government 2021). Not only has the government missed the target of publishing the strategy during 2020, but apparently they are only now in 2021 starting to gather evidence that will eventually allow them to consider the background information on pressures and threats that will need to be taken into account when developing a strategy. However, if a strategy is published, any actions proposed as compensation for impacts of offshore wind on seabirds should be fully integrated with that strategy.

The aims of this review are therefore:

a) To concisely, but systematically review the conservation status and demography of each focal seabird species and SPA population;

b) To review potential compensation measures that may be suitable ways to improve conservation status through improvements to key demographic parameters that limit population growth;

c) To consider how suitable each of these possible measures may be for strategic approaches to compensation for the seabird species regionally and nationally in Scotland, and in the UK as a whole;

d) To present the case for particular measures for each seabird species, with particular emphasis on the potential for development of strategic measures.

3 METHODS

In some cases it can be difficult to determine if a particular seabird species is included as a feature of a particular SPA, because the 'Seabird Assemblage' listed in citations and Natura 2000 standard data forms does not always list all of the species comprising the assemblage. We have used the NatureScot SiteLink v3 list of seabird features of Scottish SPAs as the most accessible comprehensive source for this information in Scotland. For SPA features in England, Wales and Northern Ireland we have accepted the classifications in Stroud et al. (2016) as a comprehensive list up until the mid 2010s, accepting that there might be some uncertainties for a few species and sites where documentation is inconsistent or unclear. We have added relevant sites that have been designated since 2016 and appreciate inputs on this from NE staff.

Conservation status of each focal seabird species was extracted from NatureScot SiteLink v3, from the JNCC web pages listing protected site details, from the Republic of Ireland online SPA features Excel spreadsheet, and from the 3rd review of UK SPAs (Stroud et al. 2016). The 3rd review had a terrestrial focus and did not consider sufficiency in the marine environment. Since then, however, several further marine sites have been designated. We have not used the 3rd review to consider sufficiency, but only to indicate the current conservation status of the focal seabird populations. In each species account, Site Condition Monitoring (SCM) is recorded as 'Favourable Maintained' FM, 'Favourable Declining' FD, 'Unfavourable Declining' UD, 'Unfavourable no change' UNc, or 'Unfavourable Recovering' UR. Favourable Maintained is traffic light coded green, Favourable Declining as amber, Unfavourable as red. This coding is used throughout the species accounts. It is important to point out that SCM is not designed as a tool for determining if compensation measures are required or not, or that if a feature is in favourable conservation status at a site then compensation measures are not required. SCM is a blunt tool in this context because sites will be designated "Favourable" if population size remains no more than 25% less than at designation (and habitat remains favourable). Nevertheless, there is a strong correlation between the SCM classifications and the change in numbers between designation and the most recent count provided in the JNCC SMP database, and that indicates a broad utility of this approach to assessing the status of a seabird species across the Natura 2000 network. That approach will, hopefully, be superseded very soon by the publication of the latest national census of seabirds breeding in Britain and Ireland.

The latest national census of seabirds breeding in Britain and Ireland is currently being completed (following those in 1969-70, 1985-86 and 1998-2002). Analysis of results from the current census may well shed new light on seabird population trends, regional differences in population trend, and likely drivers of change that are creating these regional patterns. However, many of the counts contributing to the latest national census are already listed in the JNCC Seabird Monitoring Programme online database, and so have been incorporated into this review by extracting the most up to date counts from that database for each SPA population (JNCC 2020).

Demographic parameters of UK seabirds were reviewed and tabulated by Horswill and Robinson (2015) and those have been used in this review. Additional data since publication of that review have been extracted by searching for more recent publications using seabird species and demographic parameter terms as search terms, but also by searching for publications that cite the ones listed in Horswill and Robinson (2015) as those are the most likely publications to contain any updated measurements of demography.

Literature review to identify and assess efficacy of potential compensation measures has been carried out primarily using Web of Knowledge database, as that has very extensive coverage of primary scientific literature and excellent search facilities. SCOPUS and Google Scholar were also used. Google Scholar has the best coverage of books, conference proceedings, ‘grey’ literature and reports, so is a useful addition to Web of Knowledge literature but is less easily searched. ‘Grey’ literature reports such as SNH Commissioned Research Reports, JNCC Reports, Marine Scotland Science Reports, SOSS Reports, COWRIE reports, ORJIP reports, and BTO Research Reports that are available online were also searched.

4 SPECIES ACCOUNTS

4.1 Great northern diver *Gavia immer*

4.1.1 Conservation status and populations

IUCN Red List classification "Least Concern". Birds of Conservation Concern (BOCC) 2, 3, and 4 all classified great northern diver as Amber. Birds Directive Annex 1 and migratory species.

The great northern diver (known as common loon in North America) does not breed in Scotland but spends winter here and migrates to breed in Iceland, Greenland and eastern Canada. The west and north of Scotland are the most important wintering areas for this species in Europe, holding up to 30% of the Western Palearctic winter population (Forrester et al. 2007). However, knowledge of the numbers of great northern divers in Scottish (or UK) waters is limited. The birds are not easy to count. Evidence suggests that the same identifiable individuals turn up each autumn in the same place and stay all winter. In North America, Paruk et al. (2015) estimated winter site fidelity as 85% among adult birds. When large numbers were killed by oil spill at Shetland the numbers there remained depleted for many years, also suggesting very low mobility among sites in Scotland. Woodward et al. (2020) suggest that there are about 4,400 individuals wintering in UK waters. Weir et al. (1996) estimated from biometrics of specimens in the National Museums of Scotland that about 45% of the birds wintering in Scotland originate from Iceland, 45% from Greenland and Baffin Island, and 10% from mainland eastern Canada. Despite differences in biometrics among great northern diver breeding populations (Weir et al. 1996), studies of mtDNA indicate low genetic diversity and no geographical pattern in mtDNA, consistent with a small genetic population size and mixing (Bartolome et al. 2011).

No sites had been classified for nonbreeding great northern divers when the 3rd UKSPA review was carried out in 2016 (Stroud et al. 2016). Since then, sites have been designated. SPAs for nonbreeding (wintering) great northern divers in Scotland are:

East Mainland Coast Shetland; This site was classified on 03/12/2020 to protect foraging grounds of wintering great northern divers at sea;

West Coast of the Outer Hebrides; This site was classified on 03/12/2020 to protect foraging grounds of wintering great northern divers at sea;

Moray Firth; This site was classified on 03/12/2020 to protect foraging grounds of wintering great northern divers at sea;

Coll & Tiree; This site was classified on 03/12/2020 to protect foraging grounds of wintering great northern divers at sea; and

Sound of Gigha; This site was classified on 03/12/2020 to protect foraging grounds of wintering great northern divers at sea.

In addition to these designated sites, it is likely that an area that includes Scapa Flow (Jackson 2018) and waters off the north coast of Orkney Mainland (Upton et al. 2018) will also be designated for wintering birds including great northern divers in the near future. Scapa Flow proposed SPA and North Orkney Waters proposed SPA were put forward by Marine Scotland and have qualifying numbers of great northern divers. It is custom and practice to treat pSPAs as if designated.

In England, Falmouth to St Austell Bay SPA was designated for nonbreeding great northern divers (and also black throated divers and Slavonian grebe).

Although these SPAs are not overlapping with any ScotWind lease areas, the vulnerability of great northern divers to vessel traffic means that vessel traffic relating to offshore wind could impact on this qualifying feature.

4.1.2 Demography

Adult survival (4 years old and older) 0.87 (s.e. 0.078), age of first breeding 6 years, incidence of missed breeding relatively high at about 30.7%, immature survival 0.77 (s.e. 0.02), breeding success generally around 0.54 chicks per pair, natal dispersal 0.069, and adult dispersal 0.09 (Horswill and Robinson 2015). Paruk et al. (2015) estimated adult survival as 0.77 (0.48-0.93) which is an estimate not included in Horswill and Robinson (2015) but is based on return of marked individuals to wintering areas in the USA, and may underestimate survival if some birds are not site-faithful between winters.

4.1.3 Ecology and threats to population

Great northern divers breed at freshwater lakes in USA, Canada, Greenland and Iceland, and some migrate to overwinter at sea off the coasts of Europe, especially off west and north Scotland. During winter they feed on small fish and crabs caught by foot-propelled diving, usually to the seabed, in shallow and relatively sheltered areas, especially where the seabed is sandy.

On their breeding grounds, great northern divers are under pressure from lake acidification, which can reduce forage fish populations and causes increases in methyl-mercury in the fish they eat while breeding (Evers et al. 2011, Schoch et al. 2019), and from recreational disturbance (Field et al. 2015, Buxton et al. 2019). They have been under pressure from hunting on some of their breeding grounds or migration routes, and an estimated 34% of the birds reaching Scotland carry lead shot fragments (Heubeck et al. 1993, Weir et al. 1996). They can also ingest fishing lead weights at freshwater breeding lakes (Specht et al. 2019), and so toxic effects of lead are a major concern, with lead toxicosis considered to be responsible for about half of the deaths occurring in some regions (Grade et al. 2018). Great northern divers are highly vulnerable to oil spills (Heubeck et al. 1993, Paruk et al. 2015, Evers et al. 2019). They may also accumulate high levels of polycyclic aromatic hydrocarbons (PAHs) on some of their North American wintering grounds; these compounds are highly toxic, but the fitness costs of this for great northern divers are unknown (Paruk et al. 2018). Great northern divers are at risk of bycatch in fishing gear, especially in set nets. In UK waters, Miles et al. (2020) estimated that bycatch mortality of great northern divers in UK waters exceeded 1% of total adult mortality and that removal of that bycatch mortality may increase great northern diver population size by more than 1% over a 25-year period. However, data on great northern diver bycatch in UK waters are scarce (Northridge et al. 2020) and so this estimate is uncertain. Great northern divers tend to show avoidance of shipping/ferries/fishing boats (Jarrett et al. 2018), although there has been very little research to quantify this. Anecdotal observations suggest that some individuals can tolerate human activity. Their behavioural responses to offshore wind farm structures are not yet known but may be similar to those shown by red-throated divers. If so, they are likely to avoid areas with offshore turbines, so could be affected by barrier effects and displacement, although offshore wind developments around Scotland (and the rest of the UK) are likely to be in areas that are more exposed and further offshore than habitat most preferred by great northern divers.

4.1.4 Potential compensation measures and suitability for this species

Great northern diver was not included in the review commissioned by Defra (Furness et al. 2013). However, five potential compensation measures can be identified that would be anticipated to result in a significant improvement of the conservation status of this species. These are deployment of nest platforms at breeding lakes where birds are subject to human disturbance, action to reduce use of lead weights in sport fishing in Canada, action to reduce use of lead ammunition in areas where great northern divers are shot, action to reduce hunting of great northern divers, and action to reduce vessel traffic through areas with high numbers of wintering great northern divers. Only the last of these measures could be carried out in Scotland/UK. A sixth possible measure would be bycatch reduction in UK fisheries. However, although that could be carried out in the UK, bycatch of great northern divers would be difficult to reduce because it involves small numbers of individuals (but a relatively high percent of the population), and there is uncertainty as to how this might effectively be reduced further. These measures are only outlined briefly below, since it seems unlikely that offshore wind will lead to a need for compensation in relation to great northern divers. However, the species is included here for completeness.

Nest platforms at breeding lakes:

There is strong evidence available that providing floating nesting platforms increases productivity considerably, especially on lakes where water levels fluctuate, predators are present, or human disturbance occurs (Piper et al. 2002, de Sorbo et al. 2007). This also appears to be a management option that would be generally supported by the public, and where volunteer help and matched funding may be available.

In North America, the loon preservation committee puts out about 40 rafts each year for common loons (great northern divers) in New Hampshire, and strongly advocates the benefits of these rafts for increasing productivity in that state, protecting the birds from fluctuating water levels and reducing impacts of human recreational disturbance and nest predation by raccoons (Loon Org 2020). The Big Mantrap Lake Association also deploys rafts for nesting loons on Big Mantrap Lake Minnesota (BMLA 2020). Their rafts are rather unattractive, with a metal mesh roof, whereas most diver rafts are designed to look like natural islets, but their design presumably helps to reduce predation risk, and they state that it achieves high productivity and is well used by their population of great northern divers. Because diver breeding lakes in North America freeze over in winter, they deploy rafts each spring when ice thaws, and remove them in the autumn. What appears to be an updated version of this raft is described in detail by De Sorbo et al. (2008) who also estimate the cost of materials to construct their design at around US$100 per raft for a raft that will require replacement about every 10 years. Maine Audubon Society provide guidance on construction and siting of diver nesting rafts (MAS 2020).

Deploying rafts on lakes in eastern Canada, or in Greenland or Iceland at sites where breeding success is known to be low because of high levels of human activity could be anticipated to increase breeding success significantly, so would be a suitable measure as compensation for assessed losses from this population. Feasibility is high, as the method is tried and tested and shown to be effective, although apparently not in use in Greenland or Iceland. However, whereas human disturbance is known to affect productivity of great northern divers in the United States, there is less evidence that this is the case in Iceland, Greenland, or mainland Canada, so there would be a need to determine whether there are suitable sites for such work in those locations. However, the genetic evidence indicates that the birds in the United States are the same genetic population and so compensation at sites in the United States would appear to be appropriate.

Following an oil spill that killed large numbers of great northern divers off Rhode Island, and court action to achieve compensation, Evers et al. (2019) describe in detail the process of calculating how many great northern diver nests were needed to achieve productivity gain to compensate for this loss. That paper would be a very useful model for any compensation programme. The authors point out that any future compensation should incorporate site-specific productivity data, that ranking lake habitat quality optimizes restoration (compensation) effectiveness, and that great northern diver breeding success has the potential to be highest on larger (24-81 ha) lakes.

Action to reduce use of lead weights in sport fishing in eastern USA and Canada:

Since lead toxicosis is a major cause of mortality among great northern divers, and some of their exposure to lead is from ingestion of fishing weights (especially where great northern divers breed on lakes where recreational fishing is common), the banning of use of lead for fishing weights in North America would be highly effective in reducing this mortality, at least in those parts of the breeding range of great northern divers where fishing occurs on their breeding lakes.

Action to reduce use of lead ammunition in areas where great northern divers are hunted:

Since lead toxicosis is a major cause of mortality among great northern divers, and much of their exposure to lead is from lead ammunition, the banning of use of lead ammunition in Greenland, Baffin Island and Iceland would contribute to reducing this mortality. To be effective this would require an initial study to determine where such hunting occurs, and what scope there may be to require use of lead-free ammunition. It seems likely that hunting is in Greenland, and possibly in Baffin Island and possibly in Iceland.

Action to reduce hunting of great northern divers:

It is unclear how many great northern divers are killed by hunters each year, but since about one-third of the birds in Scottish waters in winter appear to carry some fragments of lead embedded in their tissues (Weir et al. 1996), this number would appear to be large. If action could be taken to restrain hunting of great northern divers, the reduction in mortality would most likely be significant. To be effective this would require an initial study to determine where such hunting occurs, and what scope there may be to reduce that. It seems likely that hunting is in Greenland, and possibly in Baffin Island and possibly in Iceland. Impacts of hunting will be included in the estimated adult survival rate for this species. Although adult survival appears to be moderately high (0.87 according to Horswill and Robinson 2015) this is not an especially high survival rate for such a large marine bird with a late age of first breeding, so enhancement of survival by reducing mortality due to lead toxicosis seems very likely to occur if appropriate measures can be put in place.

Action to reduce vessel traffic during winter through areas with high numbers of wintering great northern divers:

This is the only measure suggested that could be carried out within the UK. Since great northern divers show avoidance of vessels at sea, a reduction in the amount of vessel traffic during winter through marine areas with high winter densities of great northern divers would reduce that disturbance. That could be achieved either by seasonal constraints on vessel movements where feasible, or the mandatory use of defined vessel lanes, and perhaps the setting of speed limits if birds are more disturbed by faster-moving vessels. Not all vessel traffic is currently licenced, and so this might require by-laws to constrain activities of unlicenced vessel types if those represent a significant component of total vessel activity in the area. However, the extent to which great northern divers are displaced by vessel activity is uncertain because it has not been subject to much study. More evidence would be needed to inform a policy of reducing disturbance by vessels. It is unclear whether vessel disturbance of great northern divers has any population-level effect through reductions in survival of disturbed birds. It is likely that disturbance may increase energy expenditure or reduce food intake by great northern divers. If so, it might reduce body reserves and that could lead to an increase in mortality. However, this is speculation based on understanding of bird ecology and physiology but not supported (or refuted) by any empirical evidence from studies of great northern divers (or other similar species). Research into energy budgets of red-throated divers (Duckworth et al. 2020) and of common guillemots during winter is being carried out at present, and results from that may help to inform the understanding of the likely costs, if any, of displacement of nonbreeding seabirds. If offshore wind construction and/or operation results in increased vessel traffic through areas with great northern divers (and especially those now designated or likely to be designated for nonbreeding great northern divers), an approach to compensation could be to reduce the total level, or spatial spread, or movement speed, or diurnal timing of vessel movements in the area during winter to ensure that offshore wind vessels were not increasing any impact of disturbance/displacement to great northern divers from existing baseline levels.

Action to reduce bycatch of great northern divers in UK waters:

Based on evidence summarised in Northridge et al. (2020), Miles et al. (2020) estimated that great northern diver population size in UK waters could increase by >1% over 25 years if bycatch could be eliminated. This suggests that this measure may have a smaller impact on population than the previously outlined measures, but it is a measure that could be introduced in UK waters, and seems likely to be effective if bycatch could be reduced. Modelling would be required to assess how effective such measures may be, but a possible approach could be to have seasonal closures of set net fisheries within areas frequented by high densities of great northern divers in UK waters during autumn to spring.

4.2 Red-throated diver *Gavia stellata*

4.2.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified red-throated diver as Amber, Amber and Green respectively. Birds Directive Annex 1 and migratory species.

The biogeographic population (Europe) is estimated at 27,000 pairs (BirdLife International 2004), of which 1,300 pairs breed in Great Britain (Dillon et al. 2009). Population trend, if any, is uncertain at the biogeographic level but also at the national level. However, UK waters are also very important for non-breeding red-throated divers, with particularly high concentrations along the coast from the Wash to the Thames, and in Liverpool Bay.

The SPA suite with breeding red-throated diver as a designated feature has ten sites in Great Britain that protect breeding lochs, four sites in Great Britain that protect foraging areas at sea close to breeding aggregations. All these sites are in Scotland (Caithness and Sutherland Peatlands SPA; Foula SPA; Hermaness, Saxa Vord and Valla Field SPA; Hoy SPA; Lewis Peatlands SPA; Mointeach Scadabhaig SPA; Orkney Mainland Moors SPA; Otterswick and Graveland SPA; Ronas Hill – North Roe and Tingon SPA; Rum SPA for nesting areas and Bluemull & Colgrave Sounds SPA; East Mainland Coast Shetland SPA; West Coast of the Outer Hebrides SPA; and Rum SPA for foraging marine areas close to breeding aggregations). The breeding site SPAs in Great Britain were estimated to hold 30.5% of the Great Britain breeding population of red-throated divers present in 2000 (Stroud et al. 2016).

Table 1. Summary of red-throated diver breeding SPA feature conservation status in the UK. Site Condition Monitoring (SCM) is recorded as 'Favourable Maintained' FM, 'Favourable Declining' FD, Unfavourable Declining' UD, Unfavourable no change' UNc, or 'Unfavourable Recovering' UR. Favourable Maintained is traffic light coded green, Favourable Declining as amber, Unfavourable as red. This coding is used throughout the subsequent species accounts.

SPA	Pairs in citation	SCM
Hermaness, Saxa Vord & Valla	26	UD
Otterswick and Graveland	26	UD
Ronas Hill – North Roe	56	FD
Foula	11	FM
Orkney Mainland Moors	18	FM
Hoy	58	FM
Caithness & Sutherland Peatlands	46	FM
Lewis Peatlands	80	UD
Mointeach Scadabhaig	48	FM
Rum	10	FM

Seven out of ten sites are in Favourable condition, suggesting that the Natura suite for breeding red-throated divers in the UK can be considered to be at Favourable conservation status.

On 03/12/2020 breeding season marine SPAs were designated with breeding red-throated diver foraging areas protected:

Rum SPA. Red-throated diver was added to the features of the existing marine extension to Rum SPA;

West Coast of the Outer Hebrides SPA. This site was designated in part to protect the foraging area at sea used by red-throated divers breeding in adjacent areas;

Bluemull and Colgrave Sounds SPA. This site was designated exclusively to protect the foraging area at sea used by red-throated divers breeding in adjacent areas; and

East Mainland Shetland Coast SPA. This site was designated in part to protect the foraging area at sea used by red-throated divers breeding in adjacent areas.

In addition, pSPAs for red-throated divers are being considered in Orkney Waters, but have not yet completed the designation process.

Stroud et al. (2016) identified three sites where non-breeding red-throated diver qualifies to be a designated feature. These are Firth of Forth SPA, where nonbreeding red-throated diver is included as a feature based on presence of 88 individuals the 1990s (SPA citation 30 October 2001), Liverpool Bay, where at that time the site was not designated but was known to hold 922 individuals in 2001/02 to 2006/07, and Outer Thames Estuary, where at that time the site was not designated but was known to hold 6,466 individuals in 1989 to 2006/07 (Stroud et al. 2016). Since then, further areas have been designated as SPAs with non-breeding red-throated diver as a feature.

Outer Thames Estuary SPA standard data form (compiled August 2010, updated February 2011) lists a “peak mean” (probably this should be a mean of peak counts) of 6,466 nonbreeding red-throated diver individuals over the period 1989-2006/07. However, more recent surveys by digital aerial photography have found much larger numbers of nonbreeding red-throated divers in this SPA, with peak population estimates of 14,161 individuals in Jan-Feb 2013, and 22,280 individuals in early 2018 (Scott et al. 2019).

Liverpool Bay SPA standard data form (compiled August 2010) lists a “peak mean” (probably this should be a mean of peak counts) of 922 nonbreeding red-throated diver individuals over the period 2001/02 to 2006/07 excluding 2003/04.

Greater Wash SPA citation (compiled, and site classified March 2018) lists a mean peak count of 1,407 nonbreeding red-throated diver individuals (2002/03 to 2005/06).

Solway Firth SPA was designated on 03/12/2020 with nonbreeding red-throated diver as a nonbreeding season feature.

Outer Firth of Forth and St Andrews Bay Complex SPA was designated on 03/12/2020 with nonbreeding red-throated diver as a feature. Pressures and threats to red-throated divers at this site were reviewed by Woodward et al. (2015).

Moray Firth SPA was designated on 03/12/2020 with nonbreeding red-throated diver as a feature.

4.2.2 Demography

Adult (3 years and older) survival 0.84 (s.e. 0.074), age of first breeding 3 years, incidence of missed breeding uncertain, juvenile survival 0.60, immature (1-2 year) survival 0.62, breeding success generally around 0.57 chicks per pair, natal dispersal uncertain, adult dispersal 0.25 (Horswill and Robinson 2015). Schmutz (2014) estimated annual survival of adults as 0.92, although based on a small sample and if two uncertain data points were included this could have been as low as 0.84.

4.2.3 Ecology and threats to population

UK birds breed at freshwater pools close to the sea. Adults commute to the sea to catch small fish, mostly in shallow areas close to shore, and mostly within 5 km of their nest site, and almost always within 10 km (Black et al. 2015). However, birds presumed to be immatures seeking nest sites to occupy will fly long distances in search of sites, including between islands such as Foula and Shetland mainland. After breeding, birds move to the sea to spend winter at sea. UK birds winter mostly in UK waters, particularly in Scotland. Birds from breeding areas in Fennoscandia and Russia migrate to winter mainly in the southern North Sea (Outer Thames Estuary and Greater Wash), moving to the German Bight in spring before migrating back to breeding areas.

Breeding success of red-throated divers is strongly influenced by human disturbance, fluctuating water levels in breeding lochs, predation of eggs and chicks, scarcity of forage fish (Forrester et al. 2007). Survival of red-throated divers is probably influenced by forage fish availability, oil pollution, risk of drowning in fishing nets, winter weather (Forrester et al. 2007, Woodward et al. 2015), and possibly by disturbance and displacement by shipping and by offshore wind farms (Woodward et al. 2015, Dierschke et al. 2017, Burger et al. 2019, Mendel et al. 2019, Allen et al. 2020, Heinanen et al. 2020). Distribution during the winter may alter in relation to regional abundance of pelagic forage fish such as sprats and juvenile herring *Clupea harengus* and possibly other forage fish species (Morkune et al. 2016). Large aggregations of red-throated divers occurred in the Moray Firth in the 1970s and 1980s but declined after the 1990s as sprat abundance declined there.

4.2.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified four potential measures that were likely to improve the conservation status of red-throated divers.

Provision of nesting rafts at breeding lochs:

Provision of nesting rafts at breeding lochs was considered highly likely to be effective, with high confidence based on evidence. There is no evidence from the literature since 2013 to suggest any change of opinion regarding the high suitability of nesting rafts. However, a recent paper describing how nest raft enhancement for great northern divers (common loons) in USA was taken forward as compensation for oil spill mortality (Evers et al. 2019) provides a useful case study with regard to this form of compensation. While this form of compensation could be carried out by individual wind farm developers if required due to HRA assessment, this is an example where a strategic approach would be highly desirable. It is likely that if compensation was required it would be due to in-combination impacts rather than the impact of a single development, and for that reason a strategic approach would be most suitable in order to ensure that the most effective selection of sites could be achieved. It is important to note that impacts on red-throated divers in the southern North Sea could not be effectively compensated by nest rafts at Scottish breeding lochs, since the Scottish red-throated diver population appears not to winter in the southern North Sea. The birds wintering in the southern North Sea mainly originate from Russia and Fennoscandia. However, impacts on SPA red-throated diver populations in Scottish waters could be compensated by nest rafts at Scottish breeding lochs, as the evidence suggests that most red-throated divers in Scottish waters are from the local breeding population.

Closure of sandeel and sprat fisheries close to wintering areas:

Diets of nonbreeding red-throated divers are not well known in Scotland or England. It is therefore difficult to conclude whether closure of sandeel and sprat fisheries in red-throated diver wintering areas would increase survival rates of the birds, but it is likely that would be the case. However, there is new literature since 2013 that adds significantly to our understanding of red-throated diver diet in the non-breeding season. Kleinschmidt et al. (2019) used DNA metabarcoding to identify fish prey species in faecal samples obtained from red-throated divers caught at sea in winter in the German Bight. Clupeids were the most frequent prey (presumably sprats and juvenile herring) but 13 families of fish were represented in the samples from 34 birds, including most frequently sandeels, mackerel, gadoids and flatfish as well as clupeids. These data support the conclusion of Morkune et al. (2016) based on stable isotope analysis that wintering red-throated divers feed mainly on pelagic forage fish rather than benthic fish but will take a wide range of fish species, including some benthic ones. This adds further to the strong evidence that non-breeding red-throated divers feed mainly on forage fish such as sprats, juvenile herring and sandeels, and therefore that their over-winter survival and body condition are likely to be influenced by the availability of this lipid-rich food supply. To the extent that reducing fishing mortality on these forage fish stocks will result in an increase in their mean abundance, red-throated divers can be expected to benefit in terms of their winter survival and body condition.

Closure of sandeel and sprat fisheries close to breeding areas:

Red-throated divers breeding in Shetland and Orkney mainly feed sandeels to their chicks. The lack of sandeels at Shetland in the late 1980s led to a decrease in breeding numbers of red-throated divers (Pennington et al. 2004). Measures that resulted in an increase in sandeel abundance would therefore be highly likely to result in an increase in breeding numbers of red-throated divers. However, sandeel fishing ended at Shetland around 2002, and there appears to have been a slight recent recovery in sandeels in the area only very recently (although with no monitoring of sandeel abundance this can only be inferred from observations of breeding success and diet of sandeel-dependent seabirds such as kittiwake and Arctic tern). Recent scenario modelling (Lindegren et al. 2018) concluded that reducing fishing on the sandeel stock at Dogger Bank would lead to a large increase in sandeel stock biomass, but that recovery of the stock may be slow and incomplete because of adverse environmental conditions. It is likely that red-throated divers catch sprats when these are available. There is fishing directed at sprat in Scottish and English waters, but fishing effort is highly variable depending on sprat abundance and distribution (ICES 2020). In 2019, sprats were taken from many areas along the North Sea coast of Scotland, as far north as Shetland (ICES 2020). Most fishing is not by UK vessels, but Scottish boats sometimes fish for sprats. For example, in 2011, Scottish boats caught 2,800 tonnes (ICES 2020). Such fishing is likely to target small areas with high local concentrations of sprats, as sometimes present in the Moray Firth or Firth of Forth. Reducing fishing mortality on sandeels and sprats in areas near to red-throated diver breeding sites is therefore very likely to result in some increase in abundance of these forage fish, and that can be expected to lead to higher breeding success of red-throated divers. Strategic reduction of fishing impact on sandeel and sprat stocks for the benefit of seabirds would most likely enhance foraging for red-throated divers as well as for several other seabird species.

Prevention of oil spills:

Red-throated divers spend a very high proportion of their time on the sea surface during the non-breeding period, and a moderately high proportion of their time on the sea surface during the breeding season. The species is recognised to be highly vulnerable to oil spills. However, the trend in oil spill risk has been a long-term decline over decades since the mid-20th century, and so this risk is lower now than in the past. Reducing oil spill incidences would reduce mortality of red-throated divers, but it is unclear how large an effect that might be. It may be less influential than measures to improve forage fish abundance. It is also difficult to see how oil spill risk could be reduced further in practical terms, given that strong efforts are already made to minimize that risk.

Reducing disturbance by vessel activity:

Recent research suggests a further possibility, based on the now much better quantified effect of ship traffic on non-breeding red-throated divers (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.

4.3 Fulmar *Fulmarus glacialis*

4.3.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 all classified fulmar as Amber. Birds Directive migratory species.

The biogeographic population (North Atlantic) was estimated at 2,700,000 to 4,100,000 pairs, of which 500,000 pairs breed in Great Britain and 39,000 pairs breed on the island of Ireland (Mitchell et al. 2004). Before the mid-18th Century, St Kilda was the only breeding site for fulmar in Britain and Ireland. Breeding range and numbers expanded dramatically from colonisation of Foula, Shetland, in 1878, followed by expansion throughout the coast of Britain and Ireland. Monitoring data suggest a population trend for the UK of a very small increase from 1986 to 1996, followed by a slow decease from 1996 to 2018, with the population index reaching 140% of the 1986 baseline in 1996 but falling to about 62% of the baseline in 2018 JNCC (2020). The index for Scotland is almost identical to that for the UK, since Seabird 2000 found that the Scottish fulmar population was about 97% of the UK total (JNCC 2020).

The SPA suite with breeding fulmar as a designated feature has 24 sites (if Forth Islands SPA is included) in Great Britain, and one in Northern Ireland, according to Stroud et al. (2016). However, fulmar does not appear in the citation for Forth Islands SPA and is not included in the SiteLink entry for Forth Islands, so is apparently not a feature of that SPA. In all cases, fulmar qualifies as a component of the seabird assemblage, and not on the basis of sites supporting populations of European importance.

The SPAs in Great Britain were estimated to hold 50% of the Great Britain breeding population of fulmars present in 2000 (Stroud et al. 2016). Twenty-three sites are in Scotland and one in Northern Ireland.

Table 2. Summary of fulmar breeding SPA feature conservation status in the UK

SPA	Pairs in citation	Most recent count (pairs)	Change from designation	Percent change	SCM
Hermaness, Saxa Vord & Valla	19,539	13,208 in 2016	-6,331	**-32**	FR
Fetlar	9,500	9,213 in 2016	-287	**-3**	UD
Noss	6,350	5,092 in 2016	-1,258	**-20**	FM
Foula	46,800	8,438 in 2016	-38,362	**-82**	UD
Sumburgh	2,542	4,431 in 2017	+1,889	**+74**	FM
Fair Isle	35,210	32,061 in 2016	-3,149	**-9**	FM
West Westray	1,400	1,195 in 2017	-205	**-15**	FR
Hoy	35,000	18,000 in 2017	-17,000	**-49**	UNc
Calf of Eday	1,955	1,836 in 2018	-119	**-6**	FM
Rousay	1,240	2,129 in 2016	+889	**+72**	FM
Copinsay	1,615	1,585 in 2015	-30	**-2**	FM
North Caithness Cliffs	14,700	14,000 in 2016	-700	**-5**	FM
East Caithness Cliffs	15,000	13,864 in 2015	-1,136	**-8**	FM
Troup, Pennan and Lion's Hds	4,400	1,894 in 2017	-2,506	**-57**	UNc
Buchan & Collieston Coast	1,765	826 in 2019	-939	**-53**	UD

Fowlsheugh	1,170	525 in 2018	-645	-55	FM
North Rona & Sula Sgeir	11,500	1,438 in 2012	-10,062	-87	UD
Cape Wrath	2,300	1,477 in 2017	-823	-36	UD
Flannan Isles	4,730	2,263 in 2013	-2,467	-52	UR
Handa	3,500	1,423 in 2017	-2,077	-59	UNc
St Kilda	62,800	29,186 in 2016	-33,614	-54	UD
Mingulay & Berneray	12,500	8,614 in 2014	-3,886	-31	FM
Shiant Isles	6,820	1,506 in 2015	-5,314	-78	UD
Rathlin Island	1,482	1,518 in 2011	+36	+2	

Based on the most recent Site Condition Monitoring assessments, 12 of the 23 UK sites appear to be in Favourable conservation status for breeding fulmar (Table 2). However, based on the comparison between the most recent count and the number at site designation, only two of the 23 UK sites show an increase in breeding fulmar numbers while 21 show a decrease (Table 2). The total numbers on these 23 UK SPAs have decreased since designation by about 126,000 pairs (Table 2). The evidence therefore suggests that the UK fulmar SPA suite is in unfavourable condition.

There were no sites identified in the 3rd UKSPA review as qualifying for fulmars at sea (Stroud et al. 2016). However, new sites have been classified, so that fulmars are a breeding season feature of the Seas off St Kilda SPA, and are a breeding season and nonbreeding season feature of the Seas off Foula SPA, both of which were designated on 3 December 2020.

4.3.2 Demography

According to Horswill and Robinson (2015), adult survival 0.936 (s.d. 0.055), age of first breeding 9 years, incidence of missed breeding not reported, immature survival (from 0-8 years) 0.26 for the period of immaturity (s.d. 0.15), breeding success averages 0.419 chicks per pair nationally but varies considerably from 0.1 to 0.8 among colonies and among years, natal dispersal 0.915, adult dispersal considered to be low (Horswill and Robinson 2015). Ollason and Dunnet (1983) report that established breeding adults will miss breeding in some years, but small sample sizes prevent population rates being quantified. Adult survival is influenced by winter North Atlantic Oscillation (survival is lower when winter is warmer) (Grosbois and Thompson 2005). Cordes et al. (2015) report that adult survival of fulmars declined from 1974 to 2009 both at a Scottish colony and at an Irish colony, suggesting that survival is driven by a wide-scale environmental driver such as density-dependence or long-term change in climate. JNCC (2020) show that at a UK level, there has been a small decline in breeding success from 1986 to 2018, breeding success falling from around 0.5 chicks per pair in the 1980s to around 0.35 chicks per pair in the early 2000s, then recovering slightly to around 0.4 chicks per pair in the 2010s. Again, the trend for Scotland is almost identical to that for the UK, since most data on fulmar breeding success are collected at colonies in Scotland.

4.3.3 Ecology and threats to population

The fulmar is a medium-sized petrel that is essentially like a very small albatross in many aspects of its ecology. Fulmars feed on small fish and marine invertebrates, and on fishery waste, especially fish livers from whitefish gutted at sea on trawlers. Their anatomy limits their ability to swallow large fish such as those discarded by trawlers, but their aggressive behaviour allows them to dominate at the side of trawlers, getting first choice of the small items discarded. They can rip open large discarded fish to pull out the liver and intestines. Fulmars lay a single egg, and have a slow-growing chick that can withstand days without food. This allows breeding adult fulmars to forage over very large areas of ocean, which they traverse using highly efficient fast gliding and slope-soaring flight (taking energy from the differential wind speed with height over the wavetops, and lift provided by wind rising from waves). They digest food into a stomach oil that is fed to the chick which, like albatross chicks, grows very fat and heavier than an adult before developing its feathers and fledging without the adult in attendance. In autumn, some fulmars disperse widely across the North Atlantic while others remain over the European continental shelf (Quinn et al. 2016), but adults return to colonies in winter to start to re-occupy nest sites. Colonies are therefore only empty of fulmars for a relatively short period around October (which is when breeding adults undergo moult and spend much time sitting on the sea surface; Quinn et al. 2016, Grissot et al. 2020), although winter nest attendance tends to be sporadic and related to wind conditions (more birds at the colony in winter during calm weather, when presumably costs of flying would be higher for a bird designed to glide using the wind).

Foraging ranges of adults from colonies have been estimated by Woodward et al. (2019) as mean 135 km, mean maximum 542 km and maximum 2,736 km. These are based on eight studies from 16 colonies tracking 86 breeding adults, so are likely to be fairly reliable, but foraging range may differ considerably between early and late breeding season in this species. This very large foraging range means that most offshore wind sites in Scotland will be within the potential breeding season foraging range of birds from a breeding SPA in Scotland. However, Bolton et al. (2019) showed that in many seabird species birds forage almost exclusively in the ‘domain’ around their colony and avoid extensive overlap with conspecifics from neighbouring colonies. This may apply to pelagic long-range foraging seabirds such as fulmar (although the limited evidence suggests that there is probably much overlap for long-distance pelagic foragers such as fulmar). If segregation did occur in fulmar that would mean that impacts would most appropriately be attributed to the closest colony rather than to all colonies within theoretical foraging range limits, whereas if fulmars overlap at sea with fulmars from many other colonies, this would alter the appropriate method for apportioning impacts.

Breeding adult fulmars make a “pre-laying exodus” in May, during which birds travel large distances (with the female feeding to develop her egg). Edwards et al. (2016) showed that adult fulmars from a colony in the north-east UK foraged widely across several biogeographical regions, up to 2,900 km from the colony. Most (60%) males remained within the North Sea region, whereas most (68%) females flew north, foraging within the Norwegian and Barents Sea. A small subset of birds (15%) travelled to the central North Atlantic. Pre-laying foraging trips by males appeared to be shorter (mean of 18 days, n=20) than by females (mean of 25 days, n=19). Differences between the sexes imply potential sex-related variation in risk from threats such as fishery bycatch. During incubation, most birds remained within 200 km of their nest while foraging, but trips included some much longer journeys, including to coastal waters of Denmark and Sweden, and to the mid-Atlantic ridge.

The dramatic expansion of the fulmar out of the Arctic into UK breeding colonies from 1878 to the 1990s has been attributed either to some genetic change allowing the expansion, or to response to availability of offal from whaling and then from trawl fisheries (Mitchell et al. 2004). However, the cause of this increase in range and breeding numbers remains uncertain. Fulmars are a dominant scavenger at fishing boats, and the reduction in discharge of fishery waste at sea may contribute to the decline in breeding numbers at UK colonies, and to declines in adult survival and breeding success of fulmars in recent decades. The evidence that adult survival correlates with ocean climate metrics suggests that global climate change may be an important driver of fulmar demography and population. Fulmar is the seabird species most frequently caught in long-line fisheries in Norway, and probably elsewhere, so fishery bycatch is a threat to this species. Mitchell et al. (2004) report that bycatch mortality of fulmars in the North Atlantic may be as much as 50,000 to 100,000 birds per year. Fulmars are also hunted in Greenland and the Faroes in particular (Wernham et al. 2002), and may be caught at sea by fishermen as a source of meat. Although hunting of fulmars has probably decreased, the scale of this is uncertain, and while ring recoveries show that this occurs, the biases in reporting probability make it very difficult to assess the scale of fulmar hunting from ring recovery data.

Fulmars accumulate high concentrations of lipid-soluble contaminants such as persistent organic pollutants (POPs) and methyl-mercury. However, there is little evidence to suggest that these contaminants have an adverse effect on fulmar survival or fecundity. Fulmars also accumulate large quantities of ingested plastic in their gizzards. This is partly due to the anatomy of the Procellariiform digestive system, which has a narrowing between the proventriculus and the gizzard. Gulls and skuas, which lack that narrowing can regurgitate pellets of indigestible material that includes remains from the gizzard. Fulmars appear unable to do that, and so accumulate indigestible material in the gizzard until it has been ground down enough to pass through the intestine. It is possible that ingested plastic may reduce the digestive efficiency of fulmars by taking up space within the gizzard, but this is indicated only by some equivocal evidence.

4.3.4 Potential compensation measures and suitability for this species

Fulmar was not included in the review commissioned by Defra (Furness et al. 2013). However, some potential compensation measures can be identified. These include reducing the depletion of forage fish stocks by industrial fisheries, and reducing bycatch of fulmars.

Reducing the depletion of forage fish stocks by industrial fisheries

In the early 1980s, fulmars at Shetland fed their chicks predominantly on sandeels (sandeels formed 72% of fulmar chick diet, Furness and Todd 1984). At the same time, fulmars at St Kilda fed their chicks mainly on pelagic zooplankton (Furness and Todd 1984), indicating large differences in diet between regions. In the mid-1990s, after the collapse of the Shetland sandeel stock, fulmar chicks at Shetland were fed predominantly on fish offal (Hamer et al. 1997), presumably scavenged at local fishing boats as it was not much digested. Pennington et al. (2004) note that “since 1985-86, fluctuations in breeding success [of fulmars in Shetland] have broadly followed changes in sandeel abundance”. However, Pennington et al. (2004) also note that fulmar breeding success shows a less strong correlation with sandeel abundance than the breeding success of Arctic tern, kittiwake and Arctic skua, and infer that this is presumably because fulmars can switch to feeding chicks more on fishery waste when sandeels are scarce, so are less dependent on sandeels. These data suggest that fulmar breeding success would increase if stocks of forage fish such as sandeels were able to recover to a higher average abundance as a consequence of reducing fishing mortality on forage fish. However, data in Table 2 suggest that decreases in fulmar populations have been at least as large, and probably larger, in north-west Scotland compared to Orkney and Shetland. That suggests that factors other than forage fish abundance may be the most important drivers of fulmar breeding numbers.

Reducing bycatch of fulmars

Northridge et al. (2020) estimated that fulmar bycatch by UK-registered fishing vessels in UK waters is probably between 2,000 and 9,500 birds per year, representing 11% of total annual mortality of adult fulmars (although bycatch may include immature birds which have somewhat lower survival). Using those data, Miles et al. (2020) concluded that removal of fulmar bycatch by UK-registered fishing vessels in UK waters would result in an increase in fulmar population of about 7% over a 25 year period. However, there is evidence for much higher unintentional bycatch of fulmars in some fisheries outside UK waters, and of deliberate harvesting of fulmars as food, both at the coast and by fishermen at sea. Dunn and Steel (2001) suggested that long-line fisheries may take a bycatch of 50,000 to 100,000 fulmars per year, with almost all of these birds being killed, as the survival rate of hooked birds on long-lines is very low. Anon (2010) reported that there is a large bycatch of fulmars in long-line fisheries and high bycatch of seabirds in lumpsucker fisheries in Norway, Iceland and Greenland. Lokkeborg and Robertson (2002) reported a bycatch of 32 fulmars for 58,420 hooks set in the Norwegian long-line fishery when no mitigation measures were used, but a smaller bycatch with mitigation (streamer lines and setting funnel). Fangel et al. (2017) found a bycatch of about 100 fulmars per year in the small-vessel demersal long-line fishery for halibut in coastal Norway. Baerum et al. (2019) found that fulmar was the most frequently caught seabird in the Norwegian coastal gillnet fishery, despite the fact that fulmar is essentially a surface-feeding seabird, with an estimated annual catch of about 500 to 4,000 fulmars in that fishery. If the total bycatch of fulmars is at the scale of tens of thousands per year in North Atlantic fisheries, then reducing that mortality would provide a potential compensation measure. There are various ways in which bycatch can be reduced, including measures such as weighting lines to achieve more rapid sinking of demersal long-lines, deployment of plastic streamer lines, and setting lines through underwater setting funnels (e.g. Lokkeborg and Robertson 2002).

Reducing plastic pollution in the North Atlantic

Fulmars appear to accumulate more ingested plastic fragments than any other European seabirds. However, the impact of that on fulmar survival is uncertain. Reducing amounts of plastic in the waters of the North Atlantic may represent a compensation measure that would benefit fulmars, but the extent of the benefit is unclear, and it is not clear whether measures could be put in place to reduce amounts of plastic in, or entering, the ocean.

4.4 Manx shearwater *Puffinus puffinus*

4.4.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 all classified Manx shearwater as Amber. Birds Directive migratory species.

The world population was estimated at 374,500 pairs, of which 300,000 pairs breed in Great Britain (Mitchell et al. 2004). However, this number is highly uncertain because Manx shearwater is difficult to census because it is a burrow-nesting nocturnal species. More recently, Perrins et al. (2019) surveyed the colonies at Skokholm, Skomer and Midland Island (also known as Middleholm) in 2018 and estimated the total at these three sites to be 456,000 pairs. They suggest that the world population of the species may be 850,000 pairs, but this does not imply that numbers have increased. Rather the suggestion is that previous estimates used by Mitchell et al. (2004) were of low accuracy. It is unclear whether the population of this species is stable, increasing or decreasing. However, many historically large colonies have been extirpated by introduced non-native mammal predators, so if the world population has not decreased, the number of colonies of the species almost certainly has. JNCC (2020) provide no estimate of population trend for the UK or any constituent country.

Manx shearwater used to breed in small numbers at Foula, Fetlar and Horse of Burravoe in Shetland (Mitchell et al. 2004) but is now thought to be extinct as a breeding species in Shetland (Brown 2020). There may be a few pairs nesting on Hoy (Orkney) on Canna, Eigg and Muck (Lochaber), on Treshnish Isles, Sanda and Inchmarnock (Argyll & Bute) (Mitchell et al. 2004). Very few pairs breed at the Calf of Man, where the population has almost been eradicated by brown rats (Mitchell et al. 2004). They have recently colonised the Isle of May (Firth of Forth) in small numbers.

The SPA suite with breeding Manx shearwater as a designated feature has four sites in Great Britain (Skomer, Skokholm and seas off Pembrokeshire SPA; Aberdaron Coast and Bardsey Island SPA; Rum SPA; St Kilda SPA) and one in Northern Ireland (Copeland Islands SPA). The SPAs in Great Britain were estimated to hold 96% of the Great Britain breeding population of Manx shearwaters present in 2000 (Stroud et al. 2016). In addition to the UK and Northern Ireland SPA suite, there are five SPAs in the Republic of Ireland where breeding Manx shearwater is a designated feature (Puffin Island SPA 6,329 pairs at designation in 2003; Skelligs SPA 902 pairs at designation in 2003; Blasket Island SPA 19,543 pairs at designation in 2003; Cruagh Island SPA 3,286 pairs at designation in 2007; Deenish Island and Scariff Island SPA 2,311 pairs at designation in 2004).

Four of the five UK sites appear to be in Favourable conservation status for breeding Manx shearwater (Table 3). The total numbers on these five SPAs may have increased since designation by about 370,000 pairs, although part of that increase may just be a result of better census data. Although the evidence suggests that the UK Manx shearwater SPA suite is in favourable condition, this species has high conservation importance because a large proportion of the world population breeds in the British Isles, and many historical colonies have been lost.

Table 3. Summary of Manx shearwater breeding SPA feature conservation status in the UK

SPA	Pairs in citation	Most recent count (pairs)	Change from designation	Percent change	SCM
St Kilda	5,000	4,800 in 2000	-200	**-4**	FM
Rum	61,000	120,000 in 2001	+59,000	**+97**	FM
Copeland Islands	5,923	4,850 in 2007	-1,073	**-18**	
Aberdaron and Bardsey	6,930	16,000 in 2001	+9,070	**+131**	
Skomer, Skokholm & the seas off Pembrokeshire	150,968	456,000 in 2018	+305,032	**+202**	

The SPA suite includes two SPAs with "non-breeding" Manx shearwater as a designated feature (i.e. sites away from colonies). These are Irish Sea tidal front SPA and Outer Firth of Forth and St Andrews Bay Complex SPA.

The Irish Sea tidal front is an important zone of aggregation of Manx shearwaters (Kober et al. 2012) and part of this, an area of 150 km^2 in UK waters SW of the Isle of Man and NW of Anglesey, has been designated as an SPA for foraging Manx shearwaters (Irish Sea Front SPA). It is thought that 12,000 Manx shearwaters, some of these originating from several different colonies, forage in the SPA area in summer (the draft citation indicates 12,039 individuals in the area).

Manx shearwater is a breeding season feature of the Outer Firth of Forth and St Andrews Bay Complex SPA designated on 03/12/2020. There is only a very small (few pairs) known breeding colony near to this SPA, so the status of Manx shearwaters seen in that area is obscure, but it seems likely that those will mostly be immature birds or birds that have chosen not to breed and aggregate in this SPA to take advantage of good feeding conditions away from competition with large numbers of breeding conspecifics nearer to large breeding colonies.

4.4.2 Demography

Adult survival 0.87 (s.e. 0.08) but data limited to colonies in Wales, age of first breeding 5 years, incidence of missed breeding relatively high at about 0.16, immature survival uncertain, breeding success generally around 0.7 chicks per pair, natal and adult dispersal considered to be low (Horswill and Robinson 2015). JNCC (2020) show that at a UK level, there has been no trend in breeding success from 1986 to 2018 and no statistically significant annual variation from mean productivity of 0.62 chicks per pair (but averaging 0.67 chicks per pair on Rum). However, on Canna, breeding success was reduced close to zero by rats, and following eradication of rats, breeding success there increased to about 0.75 chicks per pair (JNCC 2020). Low breeding success in some years at Welsh colonies has been attributed to higher sea surface temperatures and reduced prey quality (Riou et al. 2011), to flooding of burrows by heavy rain and to predators such as crows (JNCC 2020). At Rum, flooding by heavy rain is also a major influence on breeding success (Thompson and Furness 1991).

4.4.3 Ecology and threats to population

The Manx shearwater is a medium-sized seabird that breeds in a small number of mostly rather large colonies. It only returns to land at night, and nests in burrows or under boulders. On land it is clumsy because its legs are towards the tail-end of the body and are not strong enough to allow the bird to stand upright, and so it normally only breeds on islands that lack terrestrial mammal predators. It feeds on small fish and marine invertebrates, especially along tidal fronts where hydrographic processes concentrate prey. It lays a single egg and has a slow-growing chick that can withstand days without food. This allows breeding adult Manx shearwaters to forage over very large areas of ocean, which they traverse using highly efficient fast gliding and slope-soaring flight (taking energy from the differential wind speed with height over the wavetops, and lift provided by wind rising from waves). The world population breeds mostly in the British Isles so we have a particular international responsibility for the conservation of this species. In autumn, Manx shearwaters migrate to South American waters, returning to colonies in late spring.

Foraging ranges of adults from colonies have been estimated by Woodward et al. (2019) as mean 136 km, mean maximum 1,347 km and maximum 2,890 km. These are based on ten studies so are likely to be fairly reliable, but foraging range may differ considerably between early and late breeding season in this species. Foraging range may also differ between colonies, with a high probability that it is much greater from larger colonies. It is not possible to identify immature birds from adults when seen at sea, so the distribution of immature birds is unknown. It is highly likely that many immatures will occupy waters distant from large colonies in order to avoid competing for food with breeding adults. This very large foraging range means that most offshore wind sites in Scotland will be within the potential breeding season foraging range of birds from a breeding SPA in Scotland, and within the theoretical range of foraging Manx shearwaters from SPAs in Wales and the Republic of Ireland. However, Bolton et al. (2019) showed that in many seabird species birds forage almost exclusively in the 'domain' around their colony and avoid extensive overlap with conspecifics from neighbouring colonies. This may apply to pelagic long-range foraging seabirds such as Manx shearwater (although the limited evidence for this species reviewed by Bolton et al. (2019) suggests it might not, based on a single case study). If segregation did occur between Manx shearwater colonies that would mean that impacts would most appropriately be attributed to the closest colony rather than to all colonies within theoretical foraging range limits, so this would alter the appropriate method for apportioning impacts.

Manx shearwaters are known to be highly vulnerable to introduced non-native mammal predators at breeding colonies, and many formerly important breeding sites have been devastated by mammal predators, especially brown rats (Mitchell et al. 2004). Eradication of rats has led to re-establishment of growing Manx shearwater numbers at several sites, but not yet to anywhere near the numbers that used to breed at those colonies (e.g. Booker et al. 2019). Thompson and Furness (1991) found that breeding success of Manx shearwaters at Rum SPA was strongly influenced by the amount of heavy rain during incubation; heavy rain led to flooding of burrows and breeding failure. Some Manx shearwaters are killed in fishery bycatch, mostly on long-lines in their migration and wintering areas and may be caught deliberately by some fishermen to eat (Mitchell et al. 2004). The species is at risk from oil pollution on the sea surface, and possibly from ingested plastic. Manx shearwaters can be at risk from being attracted to lights at night. This probably mainly affects fledglings during their departure from nest burrows to sea, and probably has less influence on adult birds, but wrecks of Manx shearwaters can occur at lights, and Guilford et al. (2018) found that adults could be affected by lights. Responses of Manx shearwaters to offshore wind farms are not well known, but collision risk appears likely to be low because most flight occurs close to the sea surface.

4.4.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified five potential measures that were likely to improve the conservation status of Manx shearwaters:

1. Eradication of rats from breeding sites of Manx shearwaters;
2. Eradication of feral cats and ferrets from breeding sites of Manx shearwaters;
3. Exclusion of large gulls from around Manx shearwater colonies;
4. Closure of sandeel and sprat fisheries close to breeding areas; and
5. Closure of sandeel and sprat fisheries in UK waters.

Of these five potential measures, only eradication of rats from breeding sites of Manx shearwaters was considered highly likely to be effective with high confidence based on the evidence. However, eradication of feral cats and ferrets from Manx shearwater breeding sites was considered likely to be effective, with moderate confidence. There is no clear evidence from the literature since 2013 to suggest any change of opinion regarding these measures. There is, however, new research that strengthens the evidence that eradication of rats is highly effective in promoting recovery of Manx shearwater breeding at islands where the population has been driven to local extinction or to very low numbers. Brown rats were eradicated from Lundy in 2002-2004, with the island declared rat-free in spring 2004. Booker et al. (2019) showed that Manx shearwater breeding numbers at Lundy increased from 297 pairs in 2001 (when brown rats were still present), to 1,081 pairs in 2008, 3,451 pairs in 2013, and 5,504 pairs in 2017-2018. The eradication of rats at Lundy also resulted in large increases in breeding numbers of puffins, common guillemots and razorbills, and colonisation of the island by European storm-petrels in 2014, all being attributed to the removal of predation pressure from rats (Booker et al. 2019). On the Pembrokeshire island of Ramsey, the recovery pattern of Manx shearwater has been similar to that observed on Lundy. On Ramsey, the pre-eradication population of 849 pairs in 1998 increased to 4,796 pairs in 2016, following rat eradication in 1999/2000, while European storm-petrel was first recorded breeding on Ramsay in 2008, with at least 12 pairs and probably more there in 2016 (Bell et al. 2019). The Isles of Scilly Seabird Recovery Project worked to eradicate rats from the islands of St Agnes and Gugh in 2013 after large declines in breeding numbers of many seabirds attributed to predation by rats (Pearson et al. 2019). These islands recorded the first Manx shearwater chicks to fledge there in living memory in August-September 2014, and in 2015 European storm-petrels were also found breeding there (Heaney and St Pierre 2017). Black rats, which had been present on the isles since the 18th century, were eradicated from the Shiant Isles in 2015-2016 with funding from EU LIFE and SNH (now NatureScot), with the isles being declared rat-free in March 2018 (Main et al. 2019). In 2018, the first known breeding of European storm-petrel occurred, and Manx shearwaters were heard calling for the first time at these islands (RSPB 2020). These rapid responses to rat eradication demonstrate both the devastating impact of these predators, and the huge conservation benefits these recovery projects can provide (Booker et al. 2019).

In relation to possible benefits to Manx shearwaters from conservation measures to increase abundance of forage fish such as sprats and sandeels, Waggitt et al. (2018) showed a strong association between foraging Manx shearwaters in the Celtic Sea and frontal systems between mixed and stratified water. These fronts also provided highest prey availability (especially clupeids) along frontal systems. This is consistent with the designation of the Irish Sea Front SPA as a foraging area for Manx shearwater. This association between aggregation areas for clupeids (i.e. sprats and juvenile herring in the case of Manx shearwater food) and foraging hotspots used by Manx shearwaters provides further evidence that Manx shearwaters are likely to benefit from increases in forage fish abundance that would result from reducing fishing mortality imposed on those stocks.

Manx shearwater fledglings can be attracted to, and grounded at, lights. There is a risk that artificial lights may result in mortality of attracted birds. This is a highly significant conservation concern for some petrel species in certain locations (Rodriguez et al. 2017) but appears not to be a significant problem with Manx shearwater at the population level, although some birds from Scottish SPA colonies do die as a consequence of attraction to artificial lights (Miles et al. 2010, Syposz et al. 2018). Given that there does not appear to be significant mortality impact on Manx shearwater populations as a result of grounding at artificial lights in the British Isles, there would probably not be scope for any compensation for this species through reduction in exposure to artificial lights.

The overall conservation status of Manx shearwater appears to be Favourable, as does the condition of the SPA suite for this species. However, if compensation was required, there would be scope for significant improvement in the conservation status of some colonies where Manx shearwater has been extirpated by invasive non-native mammals, or where their numbers are reduced close to extirpation. Those sites are not ones that have been designated as SPAs for the species, and the only Manx shearwater SPA known to have an established population of rats is Rum. Impacts of rats on Manx shearwaters at Rum remain uncertain (Lambert et al. 2015), as does Manx shearwater breeding population size and trend at that colony.

Strategic reduction of fishing impact on sandeel and sprat stocks for the benefit of seabirds would most likely enhance foraging for Manx shearwaters as well as for several other seabird species.

4.5 European storm-petrel *Hydrobates pelagicus*

4.5.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified European storm-petrel as Amber. Birds Directive Annex 1 and migratory species.

The world population has been estimated at between 310,000 and 690,000 pairs, with between 300,000 and 680,000 pairs of the nominate subspecies in the north-eastern Atlantic (Mitchell et al. 2004). Of these, an estimated 16,997 to 28,855 pairs bred in a total of 59 colonies surveyed in Scotland for Seabird 2000, mostly in Shetland, Orkney, northern and western Scotland (Mitchell et al. 2004). There are no known colonies in Northern Ireland, but about 99,000 pairs breed in the Republic of Ireland, almost all on uninhabited islands off the west coast. As a burrow-nesting nocturnal seabird, it is difficult to census, so breeding numbers are uncertain at most colonies, and population trends are unclear (Stroud et al. 2016). Bolton et al. (2010) present evidence suggesting a substantial increase in breeding numbers at the largest UK colony (Mousa, Shetland), but a decrease in numbers has been suggested at some colonies (e.g. Priest Island; Hounsome et al. 2006, and NatureScot site condition assessments for Auskerry and Sule Skerry). Based mainly on evidence from Mousa and Treshnish Isles, JNCC (2020) suggest that breeding numbers in the UK might possibly be increasing. BTO Birdfacts suggests that breeding numbers increased between 1969 and 2000.

The SPA suite with breeding European storm-petrel as a designated feature has nine sites in Great Britain (Auskerry SPA; Isles of Scilly SPA; Mousa SPA; North Rona and Sula Sgeir SPA; Priest Island (Summer Isles SPA); Skomer, Skokholm and Middleholm SPA; St Kilda SPA; Sule Skerry and Sule Stack SPA; and Treshnish Isles SPA). The SPAs in Great Britain were estimated to hold 99% of the Great Britain breeding population of European storm-petrels present in 2000 (Stroud et al. 2016). Seven of these nine SPA breeding sites are in Scottish waters.

In addition to the UK and Northern Ireland SPA suite, there are 11 SPAs in the Republic of Ireland where breeding European storm-petrel is a designated feature (Puffin Island SPA; Skelligs SPA; Blasket Island SPA; Deenish Island and Scariff Island SPA; Bills Rocks SPA; Magharee Island SPA; Duvillaun Islands SPA; Inishglora and Inishkeeragh SPA; Illanmaster SPA; The Bull and Cow Rocks SPA; Stags of Broadhaven SPA), and these will need to be considered where ScotWind sites fall within the maximum foraging range of breeding European storm-petrels from these Irish SPA colonies. The Republic of Ireland was estimated to hold about 99,000 pairs of European storm-petrels in 2000 (Mitchell et al. 2004), so these colonies represent an important component of the species' population.

Table 4. Summary of European storm-petrel breeding SPA feature conservation status

SPA	Pairs in citation	Most recent count (pairs)	Change from designation	Percent change	SCM
Mousa	4,750	11,000 in 2015	+6,250	**+132**	FM
Auskerry	3,600	994 in 2001	-2,606	**-72**	FD
Sule Skerry and Sule Stack	500-5,000	309 in 2001	declining	**n/a**	FD
North Rona and Sula Sgeir	unknown	377 in 2001	unknown	**n/a**	FM
St Kilda	850	1,121 in 2000	+271	**+32**	FM
Priest Island	2,200	4,640 in 2019	+2,440	**+111**	FM
Treshnish Isles	5,040	8,675 in 2018	+3,635	**+72**	FM
Skomer, Skokholm & the seas off Pembrokeshire	3,500	2,560 in 2001	-940	**-27**	
Isles of Scilly	5,406	1,281 in 2015	-4,125	**-76**	

All seven sites in Scotland within the SPA suite for European storm-petrel are classified as in Favourable condition for this feature (and although two are "Favourable Declining" some populations appear to be increasing). However, difficulties in counting breeding European storm-petrels make assessments of population conservation status very uncertain. The sites in Wales and SW England seem to be in Unfavourable conservation status. Overall, for the UK, there seems likely to have been a small increase, perhaps of a few thousand pairs, in breeding European storm-petrel numbers in the Natura suite.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding European storm-petrels (Stroud et al. 2016). However, since then the Seas off St Kilda SPA was designated on 3 December 2020 with European storm-petrel as a breeding season feature.

4.5.2 Demography

This species was not included in the review by Horswill and Robinson (2015). JNCC (2020) provide no estimates of breeding success or any other demographic parameters. BTO Birdfacts indicates adult survival of 0.87, age of first breeding typically 4 years, maximum lifespan at least 38 years. Insley et al. (2014) estimated adult survival rates from the Britain and Ireland ringing recoveries database and from mark-recapture analysis from data from Priest Island for individual years 2001-2012. The mean survival rate at Priest Island was 0.796, with annual values between 0.61 and 0.96. The long-term mean for Britain and Ireland, for 1967 to 2012 in order to limit the analysis to when longer lasting incoloy rings were used, was 0.86. However, the data for Britain and Ireland suggest a slow increase in adult survival, with values after 2008 being above 0.9. At two colonies in the western Mediterranean, Sanz-Aguilar et al. (2009) estimated age of first breeding 3 or 4 years old in most cases, breeding success averaging 0.54 chicks per pair, adult survival 0.89 in one colony, 0.86 and 0.94 in two different caves at the other. Low survival in one cave was attributed to predation of adults by locally nesting yellow-legged gulls (*Larus michahellis*). Matovic et al. (2017) found differences in adult survival between Mediterranean and Atlantic populations of European storm-petrels, with overwinter survival being lower in the Atlantic population (0.82) than in the Mediterranean (0.89).

4.5.3 Ecology and threats to population

The European storm-petrel nests in crevices, under rocks, in dry-stone walls or in burrows. It comes to land only at night. Although small, it is long-lived. Breeding adults can forage at considerable distances from their colony. European storm-petrels feed at the sea surface, picking up zooplankton such as fish larvae and crustaceans. After the breeding season, in September-October, they migrate south to waters off southern Africa for the winter, returning to breeding colonies around May, so they are absent from UK waters from October to April. Immature European storm-petrels are famous for wandering around the Atlantic in summer, visiting many different breeding colonies and prospecting along some coasts where there are no breeding birds.

Foraging ranges of breeding adults from colonies have been estimated by Woodward et al. (2019) as maximum 336 km. However, this was based on a single tracking study. Recently Bolton et al. (2020) determined foraging ranges by equipping breeding adults from Mousa with GPS tags. The median maximum distance was 159 km, with the single greatest maximum distance of 397 km. That implies that there could be some connectivity between ScotWind areas in west Scotland and SPA breeding populations of European storm-petrels in colonies in the Republic of Ireland as well as in Scotland.

European storm-petrels are highly vulnerable to introduced non-native mammal predators on the remote islands where they breed. Almost all colonies on islands with mammal predators have been extirpated by predators (de Leon et al. 2006). European storm-petrels are killed by predatory birds, despite only coming on land at night, but the impact of bird predation appears more sustainable than that of predation by mammals. The species is at risk from oil pollution on the sea surface, and possibly from ingested plastic. European storm-petrels can be at risk from being attracted to lights at night. This probably affects fledglings during their departure from nest cavities to sea, and probably has little influence on adult birds, but wrecks of storm-petrels can occur at lights.

European storm-petrels appear to be at low risk of collision with offshore wind turbines, as Cook et al. (2012) reported that 98% flew below potential collision risk height but based on very limited data for this species. Most offshore wind farms are in areas with few or no European storm-petrels, so their behavioural response to the presence of offshore developments is uncertain (Dierschke et al. 2016).

4.5.4 Potential compensation measures and suitability for this species

European storm-petrel was not included in the review commissioned by Defra (Furness et al. 2013). However, it is well known that European storm-petrel breeding distribution is constrained by presence of mammal predators, especially rats (de Leon et al. 2006), probably even more strongly than the breeding distribution of Manx shearwater. European storm-petrel would benefit very strongly from eradication of non-native invasive mammal predators such as rats and the evidence supporting this conclusion is extremely strong and comes from numerous examples. The eradication of rats at Lundy resulted in large increases in breeding numbers of Manx shearwaters, puffins, common guillemots and razorbills, and also led to colonisation of the island by European storm-petrels in 2014, all these changes being attributed to the removal of predation pressure from rats (Booker et al. 2019). European storm-petrel was first recorded breeding on Ramsay in 2008, with at least 12 pairs and probably more there in 2016, a direct consequence of the eradication of rats there in 1999/2000 (Bell et al. 2019). After eradication of rats from St Agnes and Gugh in the Scilly Isles in 2013, European storm-petrels were found breeding there for the first time (Heaney and St Pierre 2017). Black rats, which had been present on the Shiant Isles since the 18th century, were eradicated in 2015-2016 with funding from EU LIFE and SNH (now NatureScot), with the isles being declared rat-free in March 2018 (Main et al. 2019). In 2018, the first known breeding of European storm-petrel occurred there (RSPB 2020). These rapid responses to rat eradication demonstrate both the devastating impact of these predators, and the huge conservation benefits these recovery projects can provide (Booker et al. 2019). Following from the lead given by Ratcliffe et al. (2009) in the specific context of invasive non-native mammals impacting storm petrels and shearwaters in the UK, Channel Islands and Isle of Man, Stanbury et al. (2017) provide a prioritization framework for identifying islands where the greatest conservation gain can be expected from eradication of non-native invasive mammals. They list islands that would most benefit most from eradications, with the top 25 islands in their list being in Scotland, Northern Ireland and the Channel Islands. They emphasize that feasibility studies taking account of factors such as interspecific interactions, anthropogenic reinvasion, views of residents or 'social feasibility' and cost need to be undertaken before planning any eradication. They also prioritised biosecurity for rat-free islands to highlight where comprehensive measures might be most beneficial.

If compensation measures were required for European storm-petrel, removal of mammal predators from islands would be highly effective in improving the conservation status of this species, as well as having benefits for other vulnerable seabirds including Leach’s petrel (in some locations in the far north-west), Manx shearwater, puffin, razorbill, black guillemot, in some colonies common guillemot where these breed in caves or under boulders, and probably for ground-nesting terns, gulls, and shags.

European storm-petrel fledglings can be attracted to, and grounded at, lights. There is a risk that artificial lights may result in mortality of attracted birds. This is a highly significant conservation concern for some petrel species in certain locations (Rodriguez et al. 2017) but appears not to be a significant problem with European storm-petrel at the population level, although some birds from Scottish SPA colonies do die as a consequence of attraction to artificial lights (Miles et al. 2010). Given that there does not appear to be significant mortality impact on European storm-petrel populations as a result of grounding at artificial lights in the British Isles, there would probably not be scope for any compensation for this species through reduction in exposure to artificial lights.

Given that the largest colony in the UK is in the Broch at Mousa (a dry-stone building) artificial nesting structures might also be a potential approach. It seems likely that these stone structures provide nesting cavities that are less at risk of access by predators than might be the case for some natural nest sites. Building stone structures like a broch on predator free islands may enhance available nesting locations. European storm-petrels will use nest boxes, and these may also enhance breeding success if designed to reduce risk of access by predators. Nesting habitat enhancement might be an option, but would need to involve careful selection of locations that are remote from access by rats.

4.6 Leach's petrel *Oceanodroma leucorhoa* (*Hydrobates leucorhous* in North America)

4.6.1 Conservation status and populations

IUCN Red List classification was "Least Concern" until very large population declines in Canada were detected recently, and the classification has now been adjusted to "Vulnerable". BOCC 2, 3, and 4 all classified Leach's petrel as Amber. Birds Directive Annex 1 and migratory species.

The world population in the late 20th century was estimated to be around 9 to 11 million pairs, out of which the nominate subspecies in the Atlantic has been estimated at 4.9 to 5 million pairs (Mitchell et al. 2004). Millions breed in the northwest Atlantic (especially Canada) and there is apparently connectivity between the populations on the two sides of the ocean with some birds joining the UK population from Canadian colonies (Bicknell et al. 2012). There are about 80,000 to 150,000 pairs in Iceland. Relatively few breed on the east side of the Atlantic, though there are about 1,000 pairs in the Faroe Islands and 100 to 1,000 pairs in Norway. An estimated 37,000 to 65,000 pairs bred in colonies surveyed in Scotland for Seabird 2000, mostly in the extreme north-west of Scotland on islands closest to the continental shelf edge (Mitchell et al. 2004). There are no known colonies in Northern Ireland, but about 310 pairs breed in the Republic of Ireland on uninhabited islands off the west coast.

As a burrow-nesting nocturnal seabird, it is difficult to census, so breeding numbers are uncertain at most British colonies, and population trends are unknown (Stroud et al. 2016). However, a very large decline in breeding numbers has been reported from Canada (d'Entremont et al. 2020, Wilhelm et al. 2020). Between 1984 and 2013 the largest colony of this species in the world apparently declined by over 40% which represents a loss of about 3 million pairs from that colony (Wilhelm et al. 2020). The conservation status of this species in the Atlantic is therefore a major concern. In Shetland, Miles et al. (2012) found strong evidence of declines, with the species still present at only two sites in 2011 (Gruney and Gloup Holm). BTO Birdfacts indicates moderate decline in breeding numbers in the UK between 1996 and 2020.

The SPA suite with breeding Leach's petrel as a designated feature has six sites in Great Britain (Flannan Isles SPA; Foula SPA; North Rona and Sula Sgeir SPA; Ramna Stacks and Gruney SPA; St Kilda SPA; Sule Skerry and Sule Stack SPA). The SPAs in Great Britain were estimated to hold over 99% of the Great Britain breeding population of Leach's petrels present in 2000 (Stroud et al. 2016). All of these six SPA breeding sites are in Scottish waters. Numbers present at designation are very uncertain because the species was not censused accurately.

Of the six SPA sites in Scotland with Leach's petrel as a breeding feature, four are classified as Unfavourable Declining and two as Favourable Maintained (Table 5). However, the recent population trend at one of the FM sites (St Kilda) is known to be a steep decline, and there are no recent data from the other FM site where FM status was assessed in 2001 and status has not been reassessed since then (Flannan Isles). Overall, the UK Natura suite for breeding Leach's petrels appears to be in Unfavourable condition. In addition, the much larger populations of this species in eastern North America are apparently in rapid decline.

Table 5. Summary of Leach's petrel breeding SPA feature conservation status

SPA	Pairs in citation	Most recent count	Change from designation	Percent change	SCM
Ramna Stacks and Gruney	20	<4 in 2012	-16	**-80**	UD
Foula	50	0 in 2019	-50	**-100**	UD
Sule Skerry and Sule Stack	5	0 in 2001	-5	**-100**	UD
North Rona & Sula Sgeir	Not known	713 in 2010	n/a	**n/a**	UD
Flannan Isles	100-1,000	1,425 in 2001	n/a	**n/a**	FM
St Kilda	5,000?	12,800 in 2006	Declining?	**n/a**	FM

No sites were listed in the 3rd UKSPA review as designated for nonbreeding Leach's petrels (Stroud et al. 2016). However, Leach's petrel is listed as a non-qualifying migrant species of interest in the Dee Estuary SPA citation document.

4.6.2 Demography

This species was not included in the review by Horswill and Robinson (2015). JNCC (2020) provide no meaningful data on demography of this species. BTO Birdfacts indicates adult survival 0.88, age of first breeding typically 5 years, lifespan up to at least 29 years and typically 13 years. Fife et al. (2015) estimated adult survival at 0.78 for adults at Bon Portage Island off Nova Scotia, with a suggestion that survival was lower in plots where herring gulls were present than in plots where they were absent. Rennie et al. (2020) estimated adult survival at 0.975 at two colonies off the coast of British Columbia (but this is the North Pacific subspecies and demography may be very different there).

4.6.3 Ecology and threats to population

Leach's petrel, more correctly known as Leach's storm-petrel, is about twice the size of our more commonly known European storm-petrel. Most nest in burrows on sloping grass, often halfway down sea cliffs. Some nest in crevices. They come to, and depart from, the colony only at night. Although small, they can be long-lived. They lay a single egg and the chick is slow-growing and can survive with infrequent feeds. Breeding adults can forage at very considerable distances from their colony. Leach's petrels feed at the sea surface, on zooplankton such as euphausids, copepods and amphipods, but also on myctophids which are deep sea fish that come to the sea surface at night. They seem to feed especially in waters along the continental shelf edge, and they are rarely seen over shallow waters. After breeding, they migrate to areas of the North Equatorial Current and Benguela Current (Pollet et al. 2019). Mean foraging range of adults from colonies has been estimated by Woodward et al. (2019) as 657 km. This was based on a single study at a Canadian North Atlantic colony but is consistent with their pelagic ecology. This implies theoretical connectivity between ScotWind sites and SPAs with Leach's petrel as a breeding feature, but the evidence from at-sea surveys suggests that Leach's petrels tend to forage along the continental shelf edge west of Scotland rather than in North Sea waters.

Leach's petrels are highly vulnerable to introduced non-native mammal predators on the remote islands where they breed. Almost all colonies on islands with mammal predators have been extirpated by predators. The large colonies are all on islands without predatory mammals. Leach's petrels are killed by predatory birds, despite only coming on land at night. The impact of bird predation appears more sustainable than that of predation by mammals but has been implicated in population decline at some colonies. The species is at risk from oil pollution on the sea surface, and possibly from ingested plastic. Leach's petrels can be at risk from being attracted to lights at night. This probably affects fledglings during their departure from nest cavities to sea, and probably has little influence on adult birds, but wrecks of Leach's petrels can occur at lights.

Cook et al. (2012) estimated that 98% of Leach's petrel flight activity occurs well below potential collision risk height, but there is little of data on flight height in this species. Most offshore wind farms are in areas with few or no Leach's petrels, so their behavioural response to the presence of offshore developments is uncertain (Dierschke et al. 2016).

4.6.4 Potential compensation measures and suitability for this species

Leach's petrel was not included in the review commissioned by Defra (Furness et al. 2013). It is highly likely that Leach's petrel would benefit from eradication of non-native invasive mammal predators such as rats, in the same way that has been shown to be highly effective for Manx shearwater and European storm-petrel. However, Leach's petrel breeds only at a few sites in the far north-west of the British Isles, so few UK islands may be in suitable geographic location for this species. The loss of the small colony at Foula, Shetland, has been attributed to predation of Leach's petrels by feral cats (Sheila Gear, *pers. comm.*). Foula was identified by Stanbury et al. (2017) as the UK island top of the list in terms of priority for eradication of invasive non-native mammal predators.

If compensation measures were required for Leach's petrel, removal of mammal predators from islands in appropriate north-westerly locations for this species would be highly effective in improving its conservation status, as well as having benefits for other vulnerable seabirds including European storm-petrel, Manx shearwater, puffin, razorbill, black guillemot, in some colonies common guillemot where these breed in caves or under boulders, and probably for ground-nesting terns, gulls, and shags.

Leach's petrel fledglings can be attracted to, and grounded at, lights. There is a risk that artificial lights may result in mortality of attracted birds. This is a highly significant conservation concern for some petrel species in certain locations (Rodriguez et al. 2017) but appears not to be a significant problem with Leach's petrel at the population level in the British Isles, although a few birds from Scottish SPA colonies do die as a consequence of attraction to artificial lights (Miles et al. 2010). Given that there does not appear to be significant mortality impact on Leach's petrel populations as a result of grounding at artificial lights in the British Isles, there would probably not be scope for any compensation for this species through reduction in exposure to artificial lights.

4.7 Gannet *Morus bassanus*

4.7.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified gannet as Amber. Birds Directive migratory species.

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

The SPA suite with breeding gannet as a designated feature has ten sites in Great Britain, with eight of these in Scotland (Ailsa Craig SPA; Fair Isle SPA; Forth Islands SPA; Flamborough and Filey Coast SPA (England); Grassholm SPA (Wales); Hermaness, Saxa Vord and Valla Field SPA; North Rona and Sula Sgeir SPA; Noss SPA; St Kilda SPA; Sule Skerry and Sule Stack SPA). 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).

Table 6. Summary of gannet breeding SPA feature conservation status

SPA	Pairs in citation	Most recent count	Change from designation	Percent change	SCM
Ailsa Craig	23,000	33,226 in 2014	+10,226	**+44**	FM
Fair Isle	1,166	4,211 in 2019	+3,045	**+261**	FM
Forth Islands	21,600	75,259 in 2014	+53,659	**+248**	FM
Hermaness, Saxa Vord & Valla	16,400	25,580 in 2014	+9,180	**+56**	FM
North Rona & Sula Sgeir	10,400	11,230 in 2013	+830	**+8**	FM
Noss	6,860	13,765 in 2019	+6,905	**+101**	FM
St Kilda	50,050	60,290 in 2013	+10,240	**+20**	FM
Sule Skerry & Sule Stack	5,900	6,420 in 2013	+520	**+9**	FM
Flamborough and Filey Coast	8,469	13,392 in 2017	+4,923	**+58**	FM
Grassholm	26,300	36,011 in 2015	+9,711	**+37**	FM

All gannet SPA breeding populations are in Favourable conservation status (Table 6). Breeding numbers within the UK Natura suite for gannet have increased since site designation by at least 90,000 pairs, and these increases appear to be continuing.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding gannets (Stroud et al. 2016). However, since then Seas off St Kilda SPA and Outer Firth of Forth and St Andrews Bay Complex SPA, both classified on 3 December 2020, both include gannet as a breeding season feature.

4.7.2 Demography

Adult (5 years and older) survival 0.919 (s.e. 0.042), age of first breeding 5 years, incidence of missed breeding uncertain but thought to be low, immature survival 0.424 (s.e. 0.007) for 0-1 year, 0.829 (s.e. 0.004) for 1-2 year, 0.891 (s.e. 0.003) for 2-3 year, 0.895 (s.e. 0.003) for 3-4 year, breeding success generally around 0.7 chicks per pair, natal dispersal uncertain, adult dispersal considered to be low (Horswill and Robinson 2015). JNCC (2020) conclude that breeding success of gannets has varied very little since their monitoring started in 1986, averaging 0.69 chicks per pair and almost always falling between 0.6 and 0.9 in all years and colonies. Deakin et al. (2019) reported adult survival of gannets at Bass Rock as 0.951 for males and 0.956 for females (2010-2018), and at Grassholm 0.946 for males and 0.882 for females (also 2010-2018). Grémillet et al. (2020) report that survival rates of adults breeding at Rouzic Island, France, fell from over 0.9 in 2014-2015 to less than 0.6 in 2018-2019. They suggest that high mortality has occurred off West Africa during the nonbreeding season, possibly caused by decline in forage fish abundance, but thought more likely to be caused by fishery bycatch of gannets or deliberate harvesting of gannets by fishermen. That conclusion might be supported by the lower survival of females at Grassholm compared to males, as 90% of females wintered in the waters off West Africa whereas only 57% of males travelled that far.

4.7.3 Ecology and threats to population

Gannets tend to nest in large colonies that are remote from human activity (the latter most likely reflecting very intense historical harvesting of gannet chicks as food, fat and feathers). They feed by plunge-diving to catch pelagic fish, taking prey from the size of sandeels up to adult mackerel. They are highly competitive at fishing boats so can obtain discards even in competition with large numbers of gulls and skuas. They lay a single egg but have consistently high breeding success. Breeding adults are able to forage over large areas; Woodward et al. (2019) list gannet foraging ranges from colonies as mean 120 km, mean maximum 315 km, and maximum 709 km. However, foraging gannets tend to avoid moving into areas close to neighbouring gannet colonies (Wakefield et al. 2013), so tend to be in the areas of sea closer to their breeding site than to other colonies. After breeding, they migrate to overwintering areas. Most UK adults winter over the continental shelf off Iberia or West Africa, with few remaining in UK waters during winter.

Historically, harvesting has greatly reduced gannet numbers and limited colony distribution. There is a continuing licenced harvest of gannet chicks at Sula Sgeir, and the growth of that colony has been slower than that of most other UK gannet colonies. Harvesting of gannets for food continues in Iceland and the Faroe Islands, and among some fishermen at sea. Harvesting of gannets at sea off West Africa has been a concern in recent years, and has been highlighted recently by Grémillet et al. (2020). Protected since the 1890s in the UK, gannet breeding numbers in the UK have increased considerably, leading to formation of new colonies and huge increases in numbers at established colonies. Ability of breeding gannets to switch diet depending on abundance of mackerel, herring, and sandeel, and ability to make long foraging trips if necessary, buffers their breeding success against fluctuations in prey fish. There are no identified threats to gannets that are thought to strongly influence overall population trend, but there is a substantial fisheries bycatch of gannets, especially in some long-line fisheries. Compared to other European seabirds, gannets accumulate relatively high levels of lipid-soluble pollutants obtained from food, but there is no evidence of toxic impacts. Some gannets are killed by oil pollution, by entanglement in fragments of fishing net or on fishing lines, but these are not thought to represent major threats. Gannets are considered to be at moderately high risk of collision with offshore wind farm turbines (Lane et al. 2020), because of their flight height profile, and at risk of barrier effects/displacement by offshore wind farms because of evidence that they avoid these structures (which probably also reduces collision risk below levels that are currently used in assessments).

4.7.4 Potential compensation measures and suitability for this species

A case might be made that there should not be a requirement for compensation for offshore wind farm impacts on gannets because the UK SPA suite for gannet is certainly overall in Favourable conservation status, with breeding numbers on the suite as a whole about 90,000 pairs above the population level at designation of these sites, and with every SPA in Favourable conservation status for breeding gannets. Furthermore, it could be argued that there is over-provision of SPA protection of this species, with more than 95% of the UK population of gannets breeding within sites where they are a designated feature. However, Britain and Ireland hold most of the breeding population of gannets, so this species is particularly important for us in a global context.

Furness et al. (2013) identified three potential measures that were likely to improve the conservation status of gannets.

1. End harvest of chicks
2. Encourage establishment of new colonies
3. Reduce bycatch in fisheries

Of these three potential measures, only ending the harvest of gannet chicks was considered highly likely to be effective, with high confidence based on evidence. Since then, there has been population viability analysis to assess in more detail the impact of the harvest of gannets that is licenced to be carried out at Sula Sgeir (North Rona & Sula Sgeir SPA). There has been much work done since 2013 in relation to fisheries bycatch of seabirds that provides better evidence of benefits that could be achieved for gannets through fisheries modifications to reduce bycatch. Finally, there has also been recent evidence of change in gannet conservation status at some colonies outside the British Isles that may be relevant to predicting future population trends and ecology.

New evidence in relation to harvesting gannet chicks

The most recent counts of breeding numbers of gannets show less growth of breeding numbers in the colony at Sula Sgeir (an 8% increase from SPA designation to 2013) than in any other UK SPA with breeding gannet as a feature (Table 5). Sula Sgeir is the only colony at which a harvest of gannet chicks is allowed, with a licence to take up to 2,000 fully grown chicks per year. In practice, the reported numbers taken are close to this set limit (an average of 1,917 taken per year from 2004 to 2014; Trinder 2016). It seems likely that the legal harvest will also result in some reduction in the breeding success of pairs from which chicks are not harvested too, since there must be considerable disturbance in the colony resulting from the harvest. Losses of chicks from nests will most likely lead to a larger reduction in breeding success than just the number reported as harvested alone. Trinder (2016) concluded from population modelling that the harvest of gannet chicks at Sula Sgeir appears to have reduced the rate of population growth at that colony below the level predicted in the absence of a harvest. He suggested that this has probably also reduced the growth rate (slightly) of other gannet colonies in the region, since there is strong evidence (from population modelling) of natal emigration of birds between colonies. However, there is 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 from the population model that the Sula Sgeir population required just over 270 breeding age immigrant recruits each year in order to have achieved the level of growth observed. This analysis suggests that the Sula Sgeir population represents a sink for emigrant gannets from other colonies, and therefore that if the legal harvest of chicks was to end, that would lead to increased growth rates of neighbouring colonies, as well as that on Sula Sgeir. Trinder (2016) states "*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*".

It should be noted that although Sula Sgeir is the only UK gannet colony where a harvest of chicks occurs, there are also harvests of gannet chicks at colonies in the Faroes and Iceland. Reducing the harvest at those colonies would also represent compensation, since the entire northeast Atlantic gannet population is a meta-population with some natal dispersal occurring between all colonies. The new evidence does not change the overall conclusion that ending the harvest of gannet chicks is highly likely to be effective as compensation, with high confidence based on evidence.

New evidence in relation to fisheries bycatch and potential compensation

Bradbury et al. (2017) produced a GIS tool to show relative risk of UK seabird species to bycatch from fisheries operating in UK waters. They identified gannet as one of the seabird species most at risk of bycatch in fisheries in UK waters, with highest risk in summer, and particularly in the inshore waters of Scotland. Miles et al. (2020) made a preliminary assessment of seabird population response to potential bycatch mitigation in the UK-registered fishing fleet and concluded that bycatch mortality of gannets in this fishery represented more than 1% of the annual natural mortality (but only slightly more). Based on data in Northridge et al. (2020) they estimate that the annual bycatch of gannets by UK-registered fishing vessels is between 25 and 764 birds per year killed. There is, therefore, some scope for compensation by reducing bycatch mortality of gannets in UK waters, but that conservation action may take place as fishery mitigation (although it has not yet) so may not be available as potential compensation. Bycatch of gannets in UK waters may also be relatively small compared to bycatch of UK gannets occurring outside the breeding season in the main areas where UK gannets spend the winter (Bay of Biscay, Iberian shelf waters and off West Africa, with apparently high and increasing take of gannets off West Africa).

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

Calado et al. (2020) also 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 that remain in southern European waters while adults have returned to breeding colonies. These authors conclude that the scale of the bycatch could have significant impacts on the gannet population.

Gremillet et al. (2020) report high bycatch and substantial harvest of gannets as food in West African waters, but the scale of this problem is unclear. Mauritanian authorities confiscated eight containers of frozen seabirds 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 illegal practice may have increased recently to represent 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 makes it a difficult one to address. Regulation within West Africa may be especially difficult. However, landing frozen seabirds in countries in Asia could be regulated.

Clark et al. (2020) used GPS tracking to investigate behavioural responses of breeding adult gannets in Iceland to fishing vessels. Discarding is illegal in Iceland and Icelandic gannets 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). The authors concluded that the lack of association between gannets and fishing boats in Iceland was due to a combination of high availability of pelagic fish and lack of discarding by Icelandic fishing boats. This implies less risk of bycatch, so represents a potential management approach to reduce that problem.

New evidence of changing ecological conditions affecting some gannet colonies

Gannets have 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). However, although some colonies in that region are growing rapidly in gannet breeding numbers, some recently established colonies have declined and even been abandoned. The declines have been attributed to harassment from an increasing population of white-tailed eagles (Barrett 2008; Pettex et al. 2015). The continuing increase in numbers of white-tailed eagles in Norway is 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 while they have not yet been seen to affect gannet breeding colonies, this possibility clearly exists in future.

Gremillet et al. (2020) found a precipitous decline in the return rates of tagged breeding gannets at Rouzic, France, from 100% in 2006–2007 to less than 30% after 2015. This decline was consistent with a marked decrease in inter-annual survival probabilities for ringed adult gannets, from >90% in 2014–2015 to <60% in 2018–2019, and with a population decline of the Rouzic gannet breeding colony in recent years. Gremillet et al. (2020) suggest that this was most likely caused by a combination of heavy fishing pressure on pelagic fish stocks in that region combined with increased fishery bycatch and intentional harvest of adult gannets by fishermen as food. Most breeding gannets from Rouzic 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.

Montevecchi et al. (2021) documented an unprecedented abandonment of nests and breeding failure by gannets at colonies in eastern Canada in late summer 2012. 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.

4.8 Arctic skua *Stercorarius parasiticus*

4.8.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2 and 3 classified Arctic skua as Amber. However, BOCC 4 (2015) classified Arctic skua as Red due to a large and rapid decline in breeding numbers in Britain. Birds Directive migratory species.

Furness (1987) suggested a world population of several hundred thousand pairs, with largest numbers in Russia, Canada and Alaska. The world population was estimated by Mitchell et al. (2004) at between 85,000 and 340,000 pairs, with between 15,000 and 35,000 pairs in the northeast Atlantic. Of these, an estimated 2,100 pairs bred in colonies surveyed in Scotland for Seabird 2000, mostly in Shetland, Orkney, northern and western Scotland (Mitchell et al. 2004). There is evidence that breeding numbers in Scotland increased in the 1970s and 1980s (Furness 1987, BTO Birdfacts). However, breeding numbers have decreased dramatically during 1986 to 2020. There was a decrease of 62% between 1999 and 2011 (Stroud et al. 2016) which follows a decrease of 37% between 1985 and 2000 (Lloyd et al. 1991). JNCC (2020) report that the population index fell continuously from 1986 to 2018, being 82% lower in 2018 than at the start of their monitoring. There are no colonies in Ireland.

The SPA suite with breeding Arctic skua as a designated feature has seven sites in Great Britain (Fair Isle SPA; Foula SPA; Fetlar SPA; Hoy SPA; Papa Westray SPA; West Westray SPA; Rousay SPA). The SPAs in Great Britain were estimated to hold 16% of the Great Britain breeding population of Arctic skuas present in 2000 (Stroud et al. 2016). All of these SPA breeding sites are in the Northern Isles.

Table 7. Summary of Arctic skua breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Fetlar	130	9 in 2017	-121	**-93**	UD
Foula	133	19 in 2019	-114	**-86**	UD
Fair Isle	110	28 in 2018	-82	**-75**	UD
West Westray	78	17 in 2008	-61	**-78**	UD
Papa Westray	135	25 in 2015	-110	**-81**	UD
Rousay	130	37 in 2010	-93	**-72**	UNc
Hoy	59	12 in 2010	-47	**-80**	UD

Of the seven SPA sites in Scotland with Arctic skua as a breeding feature, six are classified as "Unfavourable Declining" and one as "Unfavourable No change" (Table 7). This is consistent with the reported national trend that indicated an 80% decline in breeding numbers (Perkins et al. 2018). This species is relatively easy to census, and so we can have high confidence in the large decline in breeding numbers that has occurred, both on the UK Natura suite for breeding Arctic skuas and at colonies that are not components of the Natura network.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding Arctic skuas (Stroud et al. 2016). However, since then Seas off Foula SPA, classified on 3 December 2020, includes Arctic skua as a breeding season feature.

4.8.2 Demography

Adult survival 0.91, age of first breeding 4 years, incidence of missed breeding relatively low at about 0.03, immature survival from fledging to 4 years old 0.346, breeding success generally around 0.5 chicks per pair but highly variable from 0 to 1.3, natal dispersal uncertain, adult dispersal considered to be low (Horswill and Robinson 2015). Breeding success monitored by JNCC Seabird Monitoring Programme (SMP) shows high variability among years but generally very low breeding success since 2000, averaging around 0.2 to 0.3 chicks per pair, but with complete failure in 43 out of 85 data points from 2003-2018 (JNCC 2020). Adult survival at Fair Isle varied considerably among periods as a result of variability in the extent of illegal shooting at that colony (O'Donald 1983). Although Horswill and Robinson (2015) report incidence of missed breeding and adult dispersal as low, there is some evidence that these can increase when conditions are unfavourable. It seems probable that some adults moved their breeding sites between islands in Orkney to evade impacts from increasing great skuas. Nonbreeding by established adults has become common in recent years of severe food shortage (Sheila Gear, pers. comm.).

4.8.3 Ecology and threats to population

Arctic skuas (called parasitic jaegers in North America) have two types of ecology. Most breed on Arctic tundra, spaced out by territorial behaviour and feed on lemmings, small birds, bird eggs and chicks, insects and berries. Those that breed on the southern edge of their circumpolar breeding range, as in Scotland, breed in loose colonies on moorland close to colonies of other seabirds, and feed outside the colony almost exclusively by stealing fish (kleptoparasitism) from other species of seabirds such as terns, puffins and kittiwakes (Furness 1987).

Foraging ranges of adults from colonies have been estimated by Woodward et al. (2019) as mean 2 km. However, this was based on only two studies. In Shetland during the 1970s, when sandeel abundance was high and most seabirds had high breeding success, Arctic skuas breeding at Foula fed almost exclusively within 2 km of the colony, patrolling along the coastline to intercept birds carrying food back to the island. That makes sense because the highest density of food-carrying seabirds occurred closest to the island. With breeding failures of seabirds in more recent decades some Arctic skuas still behave that way, but others commute out to foraging grounds used by other seabirds, so their foraging range has increased considerably as food availability has decreased. GPS tracking of small numbers of Arctic skuas breeding at Fair Isle and at Rousay, Orkney, found that successful birds at Rousay tended to feed within a few km of their colony. At the other extreme, one bird that failed at Fair Isle went on a prolonged foraging trip to Dogger Bank (585 km). A bird that bred successfully at Fair Isle made long foraging trips during incubation, but shorter ones once the chick had hatched (Harris 2020).

Perkins et al. (2018) identified the two main pressures driving Arctic skua declines at colonies in Scotland as top-down predation impacts from great skuas, and bottom-up impacts through the food chain reducing prey availability (reduced numbers of seabirds carrying food that Arctic skuas can steal). The increase in predation has been caused by increases in the numbers of great skuas, together with changes in food availability to great skuas leading to them switching to killing increased numbers of other seabirds (Votier et al. 2004, Church et al. 2018). The reduction in seabirds carrying fish has been attributed to declines in sandeel abundance, especially at Shetland, caused by climate change (ocean warming), fishery harvesting of sandeels and recovery in abundance of large predatory fish that eat sandeels (Frederiksen et al. 2004, 2007). There is probably also a direct effect of warming climate, as both Arctic and great skuas appear to be heat-stressed in warmer conditions during the breeding season at Scottish colonies (Oswald et al. 2008, 2011). Arctic skuas in some Scottish colonies have been subject to illegal persecution (including destruction of nests and shooting of adults). This was historically a particular problem at Fair Isle, where adult survival of Arctic skuas was much lower during periods when illegal shooting was regular compared to periods when research on Arctic skua ecology prevented that persecution (O'Donald 1983). Arctic skuas are scarce migrants at most operational offshore wind farms so little is known about their behaviour in relation to offshore turbines. Avoidance rate is unknown. However, Furness et al. (2013) classified Arctic skua as likely to be at moderately high risk of collision.

4.8.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified three potential measures that were likely to improve the conservation status of Arctic skuas.

1. Provision of supplementary food to breeding pairs;
2. Closure of sandeel and sprat fisheries close to breeding areas; and
3. Exclusion of great skuas from buffer zones around colonies

Of these three potential measures, only provision of supplementary food was considered highly likely to be effective, with high confidence based on evidence, but with some uncertainty as to whether this would be a practical method. However, there is some further evidence from the literature since 2013 to inform on the suitability of all three of these potential measures.

New evidence on provision of supplementary food to breeding pairs:

In 2019 and 2020, Sheila Gear carried out supplementary feeding experiments with Arctic skuas at Foula, Shetland. In 2019, half of the breeding pairs were given supplementary food and their breeding success was compared with the other half (control pairs) that were not fed. In 2020, all pairs were given supplementary food and their breeding success was compared with the situation in 2009-2018, when no supplementary feeding was carried out. Supplemented pairs were fed one feed per day, in the evening in order to encourage them to still hunt for themselves through the day. The feed was mainly of mackerel but also sometimes herring, saithe or octopus, cut up into small strips equivalent to a full grown sandeel. Three strips were fed per pair during incubation, increasing to six strips for pairs with one fledgling and eight strips for two fledglings. They were fed until they left at the end of the season or until 4 September by which time the latest hatched fledglings were old enough to leave the colony with the adults. All pairs accepted the supplementary food. Breeding success of pairs given supplementary food in 2019 was 1.5 chicks per pair (10 pairs), whereas unfed (control) pairs produced 0.5 chicks per pair (10 pairs). Breeding success in 2020 was 0.95 chicks per pair (21 pairs). Breeding success in 2009-2018 (years with no supplementary feeding) was between 0.0 and 0.86 chicks per pair, with a median of 0.11 chicks per pair (numbers of pairs declining from 63 pairs in 2009 to 20 pairs in 2018). These data are reported in annual reports on seabird monitoring by Shetland Oil Terminal Environmental Advisory Group (SOTEAG). This study shows that, as reported by Davis et al. (2005), supplementary feeding increases breeding success, and therefore further emphasises the cause of breeding failure and population decline being strongly influenced by food shortage, and the fact that supplementary feeding can compensate for that food shortage.

New evidence on closure of sandeel and sprat fisheries close to breeding areas:

There is very strong evidence that Arctic skua breeding success and survival are influenced by sandeel abundance. Breeding success of Arctic skuas at Foula between 1976 and 2004 shows a strong correlation with sandeel stock abundance, a natural logarithm regression fitting significantly better than a linear regression, and sandeel stock abundance explaining 71.2% of the variance in breeding success (Figure 1). Arctic skuas feed almost exclusively on sandeels at Shetland, stealing those from breeding puffins, kittiwakes and Arctic terns. Survival rates of adult Arctic skuas are also affected by sandeel abundance (Davis et al. 2005).

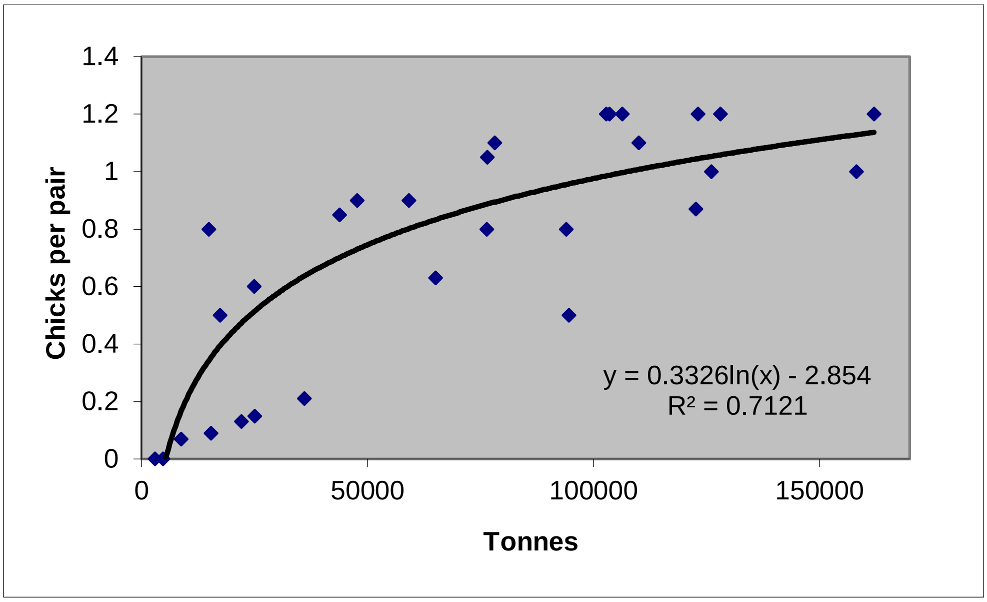

Figure 1. Breeding success of Arctic skua at Foula, Shetland, in relation to the Shetland sandeel total stock biomass for the years 1976 to 2004.

Unfortunately, with the decline in sandeel abundance, closure of the fishery resulted in loss of the fishery-derived data required for stock assessment, and fishery-independent surveys were ended some years after closure of the fishery on the basis that the small stock biomass and lack of a commercial fishery did not justify expenditure on sandeel stock assessment at Shetland after 2004. However, modelling of the impact of fishing on sandeel stocks (Lindegren et al. 2018), concluded that fishing mortality was a major driver of sandeel abundance: sandeel abundance in other North Sea sandeel stocks would be higher if fishing mortality on sandeels was reduced. Lindegren et al. (2018) cautioned that sandeel stock recovery may be inhibited by other environmental constraints such as bottom-up impacts of climate change and top-down impacts of predation mortality imposed on the reduced population of sandeels after heavy fishing pressures. The apparently very slow nature of recovery of sandeel abundance at Shetland is consistent with that. However, there is evidence that reduced fishing pressure in ICES area 4 (off east Scotland) has resulted in a slow recovery of sandeel biomass there (ICES 2020). There is a suggestion of some slight recovery of forage fish abundance at Shetland, with slightly improved breeding success of Arctic terns and kittiwakes especially in and since 2018 (Shetland Bird Reports and Shetland Bird Club Newsletters).

Tracking of Arctic skuas breeding at Fair Isle and Rousay (Harris 2020) showed interesting differences between these colonies; breeding adults from Rousay mostly foraged close to Rousay and achieved moderate breeding success. Arctic skuas attempting to breed at Fair Isle travelled much further in search of food, and all pairs but one failed to rear chicks. The difference was attributed to food availability, with adults at Fair Isle simply unable to find enough food to keep chicks alive, even when putting in high effort and foraging over exceptionally large areas from the colony.

New evidence on exclusion of great skuas from buffer zones around colonies:

Perkins et al. (2018) analysed data on numbers and breeding success of Arctic skuas in 33 colonies in Scotland in relation to changes in numbers of their hosts chased to steal food (kittiwakes, Arctic terns, guillemots, puffins) and of great skuas which represent a predation and territory displacement threat. They concluded that Arctic skuas declined by 81% and their hosts by 42%–92%, whereas at most colonies great skuas increased. Annual productivity declined in Arctic skuas and their hosts and reduced Arctic skua breeding success was a driver of the species' population decline. Arctic skua productivity was positively associated with annual breeding success of hosts and negatively with great skua density. Intercolony variation suggested Arctic skua trends and productivity were most sensitive to top-down pressures at smaller colonies of host species where great skuas had increased most, whereas bottom-up pressures dominated at large colonies of host species. This indicates that if it were possible to establish buffer areas around Arctic skua colonies from which great skuas were excluded, the evidence indicates that this would benefit Arctic skuas, but that it might be insufficient to prevent decline in Arctic skua numbers unless bottom-up recovery also occurred to provide these birds with more feeding possibilities.

4.9 Great skua *Stercorarius skua*

4.9.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified great skua as Amber. Birds Directive migratory species.

Furness (1987) estimated the world population in the 1980s at 12,500 pairs, with 7,150 pairs in northern Scotland. The world population around 2000 was estimated at 16,000 pairs, with 9,600 pairs in Scotland, most of those breeding in Shetland and Orkney (Mitchell et al. 2004). The breeding population in Scotland increased very considerably from only a few pairs in 1900, but the rate of increase has been much less since 2000, and recently numbers have fallen at a few colonies, especially the largest ones. The historical increase in numbers was accompanied by formation of many new colonies, but the range spread within Scotland has greatly slowed in recent years.

The SPA suite with breeding great skua as a designated feature has nine sites in Great Britain (Fair Isle SPA; Fetlar SPA; Foula SPA; Handa SPA; Hermaness, Saxavord and Vallafield SPA; Hoy SPA; Noss SPA; Ronas Hill SPA; St Kilda SPA). The SPAs in Great Britain were estimated to hold 74% of the Great Britain breeding population of great skuas present in 2000 (Stroud et al. 2016). All nine SPA breeding sites are in Scottish waters, with seven in Shetland and Orkney and two in north-west Scotland.

Table 8. Summary of great skua breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Hermaness, Saxa Vord & Valla	788	955 in 2018	+167	**+21**	FM
Fetlar	508	743 in 2017	+235	**+46**	FM
Foula	2,270	1,846 in 2015	-424	**-19**	FR
Ronas Hill	130	289 in 2017	+159	**+122**	FM
Noss	420	476 in 2018	+56	**+13**	FM
Fair Isle	110	490 in 2019	+380	**+345**	FM
Hoy	1,900	1,063 in 2019	-837	**-44**	UD
Handa	66	283 in 2018	+217	**+329**	FM
St Kilda	270	179 in 2012-19	-91	**-34**	FM

Of the nine SPA sites in Scotland with great skua as a breeding feature, seven are classified as "Favourable Maintained", one as "Favourable Recovered" and one as "Unfavourable Declining" (Table 8). Overall, this is largely consistent with the population in Scotland still growing slightly (especially in smaller colonies). This species is relatively easy to census and so we can have high confidence in the reported population distribution and recent trends. However, for the UK Natura suite for breeding great skua, the total numbers breeding has not changed, because large decreases at three sites offset increases at the other six sites.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding great skuas (Stroud et al. 2016). However, since then Seas off Foula SPA, classified on 3 December 2020, includes great skua as a breeding season and a nonbreeding season feature.

4.9.2 Demography

Adult survival 0.882 (s.d. 0.038), age of first breeding 7 years, incidence of missed breeding about 0.09, immature survival 0.73, breeding success generally around 0.65 chicks per pair, natal dispersal 0.015, adult dispersal considered to be very low (Horswill and Robinson 2015). JNCC SMP shows high variability in breeding success from 1986 to 2018, but with generally lower breeding success later in that period. Breeding success in the 1970s was usually around 1 to 1.2 chicks per pair (Furness 1987) whereas in 2000 to 2018 it varied between 0.16 and 0.76 chicks per pair and averaged around 0.4 (JNCC 2020).

4.9.3 Ecology and threats to population

The great skua feeds by killing birds, by scavenging (especially on fishery discards), by stealing food from other seabirds (especially gannets), by catching fish at the sea surface. Although the diet and feeding ecology of the species can be described as ‘generalist and opportunistic’ most individuals specialise in particular feeding methods or prey (Furness 1987, Jakubas et al. 2018). It nests in colonies, on moorland or grassland, usually close to large colonies of cliff-nesting seabirds.

After breeding, birds migrate to winter off southern Europe or north-west Africa. Most remain over the continental shelf, avoiding crossing land or going over deep oceanic habitats. However, birds breeding in Iceland winter either off North America or off Europe, and in winter tend to stay somewhat further north than birds from Scottish colonies. Juveniles and immatures (which look almost the same as adults) tend to travel more widely than adults.

Scotland is on the southern edge of this species’ breeding distribution, and this polar-adapted seabird may suffer from heat stress at colonies in summer. However, the population in Scotland has increased enormously over the last 150 years.

Foraging ranges of adults from colonies have been estimated by Woodward et al. (2019) as mean 67 km, mean maximum 443 km and maximum 1,003 km. Wade et al. (2014) showed that failed breeders travelled much further from the colony than birds that were still rearing chicks, that breeding success of great skuas in the year the tracking was carried out was low by comparison with historical data, and that duration of foraging trips was about five times longer than had been recorded in earlier years.

Illegal persecution (particularly shooting of adults) has prevented the development of some new colonies and has inhibited growth of several (Furness 1987). The amount of shooting of great skuas has almost certainly decreased but this practice may still occur. Great skuas accumulate high concentrations of pollutants, but there is no evidence that these are currently causing any impact on survival or breeding success (and some evidence that they are not). Some immature birds from Scotland are harvested as food (for example in Greenland). Some great skuas are killed as bycatch on long-line fisheries in southern Europe. However, the greatest threats are most likely climate change (great skuas are vulnerable to increasing temperature on their breeding areas; Oswald 2008, 2011) and changes in fishery practices (depletion of sandeel stocks has reduced that important food supply, and scavenging discards from trawl vessels has also supported large numbers of breeding great skuas, especially in their larger colonies).

Great skuas are scarce migrants at offshore wind farms so little is known about their behaviour in relation to offshore turbines. Avoidance rate at offshore wind farms is unknown. Based on a review of detailed evidence collected in Orkney, Furness (2015) recommended the use of an avoidance rate of 0.995 for great skua at terrestrial wind farms in Scotland. GPS tracking data from adults tagged at colonies (Ross-Smith et al. 2016) indicated that an average of around 6%, up to a maximum of 9%, of great skua flight activity occurred at collision risk height (for a 22-250 m turbine).

4.9.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified three potential measures that were likely to improve the conservation status of great skuas.

1. Supplementary feeding at colonies;
2. Closure of sandeel and sprat fisheries close to breeding areas; and
3. Reduction of fishery bycatch of great skuas

Of these three potential measures, supplementary feeding at colonies was considered highly likely to be effective, with high confidence based on evidence, but was considered unlikely to be cost-effective and of low practicality. There is no clear evidence from the literature since 2013 to suggest any change of opinion regarding that. Great skua breeding numbers (with about 20,000 breeding adults weighing about 1.3 kg per bird) are too large for supplementary feeding to be practical even if it was desirable. Given the unpopular nature of this species with many stakeholders, supplementary feeding of great skuas would also be very difficult to carry out in the probable face of opposition to such measures.

However, understanding of the current conservation status of great skuas has improved as a result of recent census work (numbers are still increasing in Scotland despite some declines at some of the largest colonies), as has the understanding of their ecological interactions with other species, which both provide relevant context. Church et al. (2018) showed that declines in sandeel abundance at Shetland, combined with reductions in fishery discarding rates, led to increases in bird prey in the great skua diet while breeding. Whereas about half of the pellets regurgitated by great skuas in the 1970s comprised remains of sandeels, those have almost completely disappeared from the diet. While great skuas still feed extensively on fishery discards, the species composition of discards in their diet has changed, with increases in blue whiting suggesting foraging further from Shetland over the continental shelf edge, and fish size consumed at fishing boats has decreased. Bird prey was especially kittiwake in the 1990s, but that has changed too, with the importance of auks and fulmars increasing in the diet.

CPotential benefits to great skuas from closure of sandeel and sprat fisheries, and reduction of fishery bycatch of great skuas, may both be further informed by evidence reported since 2013.

New evidence on potential of closure of sandeel and sprat fisheries close to breeding areas:

Modelling of the impact of fishing on sandeel stocks (Lindegren et al. 2018), concluded that fishing mortality was a major driver of sandeel abundance: sandeel abundance in North Sea sandeel stocks would be higher if fishing mortality on sandeels was reduced. Lindegren et al. (2018) cautioned that sandeel stock recovery may be inhibited by other environmental constraints such as bottom-up impacts of climate change and top-down impacts of predation mortality imposed on the reduced population of sandeels after heavy fishing pressures. The apparently very slow nature of recovery of sandeel abundance at Shetland is consistent with that. However, there is evidence that reduced fishing pressure in ICES area 4 (off east Scotland) has resulted in a slow recovery of sandeel biomass there (ICES 2020). There is a suggestion of some slight recovery of forage fish abundance at Shetland, with slightly improved breeding success of Arctic terns and kittiwakes especially in and since 2018 (Shetland Bird Reports and Shetland Bird Club Newsletters). While the ability of great skuas to switch diet according to food availability will somewhat buffer them from impacts of change in sandeel abundance (Church et al. 2018), the evidence is clear that great skuas fed very extensively on sandeels, and achieved very high breeding success, when sandeels were abundant at Shetland. Great skua breeding success at Foula, the largest colony of this species in the world, was low during years when Shetland sandeel total stock biomass was below 25,000 tonnes but was high in most years when the stock was above that threshold (Figure 2).

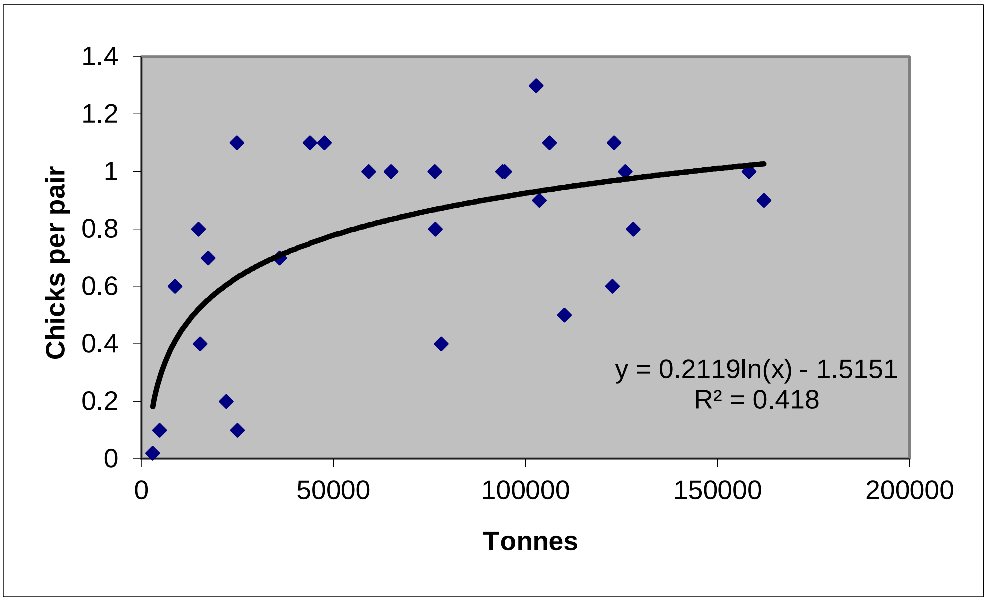

Figure 2. Breeding success of great skua at Foula, Shetland, in relation to the Shetland sandeel total stock biomass for the years 1976 to 2004

A recovery of sandeel stock at Shetland would therefore benefit great skua, though not as strongly as it would benefit sandeel specialists such as kittiwake.

New evidence on potential of reduction of fishery bycatch of great skuas:

Church et al. (2018) show that breeding great skuas in Shetland still feed extensively on fishery discards, and therefore are likely to be attracted to fishing boats and so at risk of bycatch. Ring recovery data show that bycatch on long-lines and in nets has occurred for many decades in great skua wintering areas, in particular in the Bay of Biscay, off Iberia, and in the western Mediterranean, where nonbreeding great skuas show strong association with fishing boats (Wernham et al. 2002). More recently, numbers of great skuas wintering off West Africa have increased (Magnusdottir et al. 2012), and this may be in response to the considerable increase in fishing activity off West Africa (Gremillet et al. 2020). Great skuas wintering off West Africa spend less time during winter flying compared to great skuas wintering off Europe, which suggests that they can obtain food very easily in West Africa (Magnusdottir et al. 2014). However, this behaviour will bring great skuas into increased risk of bycatch, and of being harvested as food, as seen in gannets (Gremillet et al. 2020).

Northridge et al. (2020) estimated that bycatch of great skuas by UK-registered fishing vessels is very low indeed, and can be considered negligible. Great skuas are mainly considered to be at risk of bycatch on long-lines rather than in nets (Bradbury et al. 2017) although ring recovery data include birds caught in nets off southern Europe (Furness 1987). Although Oliveira et al. (2015) and Calado et al. (2020) both report great skua among the most frequent scavenging seabirds at fishing boats off Iberia, neither reported cases of great skua bycatch in their sampling. There is, therefore, still a considerable uncertainty as to the scale of bycatch of great skuas in their nonbreeding grounds off southern Europe and West Africa. More evidence of the scale of bycatch in those regions would be required if bycatch reduction was to be considered as a compensation tool in relation to great skua.

4.10 Lesser black-backed gull *Larus fuscus*

4.10.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified lesser black-backed gull as Amber. Birds Directive migratory species.

The biogeographic population (of "our" subspecies *graellsii*) was estimated at 179,000 pairs, of which 110,000 pairs breed in Great Britain and 4,800 pairs breed in Ireland (Mitchell et al. 2004). Breeding numbers in the UK increased by 29% from 1969 to 1986 and by 40% from 1986 to 2000 (JNCC 2020). Breeding numbers in the UK declined from 2000 to 2018, with the population index about half in 2018 what it had been at the peak around 2000, although JNCC have relatively low confidence in that index (JNCC 2020). In addition, breeding numbers appear to have increased in many urban colonies that are not included in the index (JNCC 2020). Whereas lesser black-backed gull breeding numbers appear to have declined in England, they have increased in Northern Ireland (JNCC 2020). In Scotland, the population index indicates stable breeding numbers from 1986 to 2018, the index in 2018 being almost the same as in 1986 (JNCC 2020).

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; Ribble and Alt Estuaries SPA) and one in Wales (Skokholm, Skomer and Middleholm SPA). The SPAs in Great Britain were estimated to hold 38.5% of the Great Britain breeding population of lesser black-backed gulls present in 2000 (Stroud et al. 2016). Two sites in Northern Island also qualify (Lough Neagh and Lough Beg; Rathlin Island).

Table 9. Summary of lesser black-backed gull breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Forth Islands	1,500	2,051 in 2018	+551	**+37**	FM
Ailsa Craig	1,800	189 in 2019	-1,611	**-90**	UNc
Rathlin Island	155	143 in 2011	-12	**-8**	
Lough Neagh and Lough Beg	No data	>40 in 2018	n/a		
Morecambe Bay and Duddon	ca. 6,000	390 in 2019	-5,610	**-94**	
Bowland Fells	4,575	14,627 in 2018	+10,052	**+220**	
Ribble and Alt Estuaries	1,800	7,022 in 2016	+5,222	**+290**	
Skomer, Skokholm and seas	20,300	6,500 in 2018	-13,800	**-68**	
Isles of Scilly	3,608	2,465 in 2015	–-1,143	**-32**	
Alde-Ore Estuary	14,070	1,424 in 2018	-12,646	**-90**	

Five sites in the UK Natura network for breeding lesser black-backed gull are in Unfavourable conservation status while three are Favourable and one is uncertain (but holds rather few birds of this species) (Table 9). Overall, decreases sum to more pairs than increases, which gives a similar picture to that obtained from the JNCC SCM data.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding lesser black-backed gulls (Stroud et al. 2016).

4.10.2 Demography

Adult (2 years and older) survival 0.885 (s.d. 0.022), age of first breeding 5 years, incidence of missed breeding relatively high at about 0.34, juvenile survival 0.82, breeding success generally around 0.53 chicks per pair, natal dispersal 0.47, adult dispersal uncertain (Horswill and Robinson 2015). Breeding success has been monitored since 1986 and shows high variability, but little evidence of any long term trend, averaging around 0.52 chicks per pair (JNCC 2020). Breeding success in Scotland has not differed much from that monitored across the UK as a whole but the Scottish data are from a small sample size so may be less reliable. There is a hint in the Scottish data of lower breeding success since 2011 (JNCC 2020). Rock and Vaughan (2013) found a decline in adult survival of urban-nesting lesser black-backed gulls in Bristol, from above 0.9 in the early 1990s to <0.7 by 2007, suggesting that declines in some populations may be driven by declining adult survival. Sotillo et al. (2019) showed that chicks fed predominantly terrestrial food developed less well than chicks fed more marine prey. However, they attributed this to differences in parental quality/habitat use rather than to intrinsic quality of the food, as hand-reared chicks grew equally well on either diet. However, their results suggest that birds using marine resources performed better in terms of breeding success, probably due to differences in bird quality.

4.10.3 Ecology and threats to population

Lesser black-backed gulls nest in colonies on the ground, in locations selected to be away from impacts of mammal predators and human disturbance. These include 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, but still represents a small fraction of the population. 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) list 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. Large numbers of licences are still being issued to allow continued control of lesser black-backed gulls, but licences now require reporting, which should in future provide better understanding of the extent to which this control is driving population decline. 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.

4.10.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified five potential measures that were likely to improve the conservation status of lesser black-backed gulls.

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.

Literature published since 2013 does not seem to 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 consequences 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.

4.11 Herring gull *Larus argentatus*

4.11.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified herring gull as Amber, Red and Red respectively. Birds Directive migratory species.

The biogeographic population (subspecies *L. a. argenteus*) was estimated at 752,000 pairs, of which 130,000 pairs breed in Great Britain and 6,500 pairs in all-Ireland (Mitchell et al. 2004). Breeding numbers in the UK decreased by 48% between 1969 and 1986 and by 13% between 1986 and 2000 (JNCC 2020). JNCC SCM data show the population index in 2018 at 52% of the 1986 value for the UK as a whole. For Scotland, the situation is similar to that for the whole UK, with the index in 2018 at 58% that in 1986 (JNCC 2020).

Stroud et al. (2016) identified that the SPA suite with breeding herring gull as a designated feature has 11 qualifying sites in Great Britain, eight in Scotland (Ailsa Craig SPA; Buchan Ness to Collieston Coast SPA; Canna and Sanday SPA; East Caithness Cliffs SPA; Forth Islands SPA; Fowlsheugh SPA; St Abb's Head to Fast Castle SPA; Troup, Pennan and Lion's Heads SPA), three in England (Alde-Ore Estuary SPA; Flamborough and Filey Coast SPA; Morecambe Bay SPA). The SPAs in Great Britain were estimated to hold 12.5% of the Great Britain breeding population of herring gulls present in 2000 (Stroud et al. 2016). One site in Northern Island also qualifies (Rathlin Island).

Table 10. Summary of herring gull breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
East Caithness Cliffs	9,400	3,267 in 2015	-6,133	**-65**	UNc
Troup, Pennan & Lion's Heads	4,200	546 in 2017	-3,654	**-87**	UD
Buchan Ness to Collieston	4,292	2,077 in 2019	-2,215	**-52**	UNc
Fowlsheugh	3,190	1,035 in 2018	-2,155	**-68**	UD
Forth Islands	6,600	5,962 in 2018	-638	**-10**	FM
St Abb's Head to Fast Castle	1,160	283 in 2018	-877	**-76**	UD
Canna and Sanday	1,300	99 in 2019	-1,201	**-92**	UD
Ailsa Craig	2,250	213 in 2019	-2,037	**-91**	UNc
Flamborough & Filey Coast	1,110	351 in 2017	-759	**-68**	
Alde-Ore Estuary	No data	591 in 2018	n/a	**n/a**	
Rathlin Island	4,037	28 in 2011	-4,009	**-99**	
Morecambe Bay & Duddon	11,000	451 in 2020	-10,549	**-96**	

In summary, the UK Natura suite for breeding herring gull is in Unfavourable conservation status, with a decrease in breeding numbers in these SPA colonies in excess of 30,000 pairs since designation, and almost half the sites showing a decrease in excess of 90% (Table 10). Indeed, all but one of the colonies for which the decrease can be quantified show a greater decrease than the national trend identified by JNCC SCM data, suggesting that the SPA colonies (which are the largest colonies) have decreased faster than non-SPA colonies, as would be consistent with the change being density-dependent.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding herring gulls (Stroud et al. 2016). However, since then Solway Firth SPA, classified on 3 December 2020, includes herring gull as a nonbreeding season feature. Outer Firth of Forth and St Andrews Bay Complex SPA, also classified on 3 December 2020, includes herring gull as a feature in the breeding season and in the nonbreeding season.

4.11.2 Demography

Adult (2 years and older) survival 0.834 (s.d. 0.034), age of first breeding 5 years, incidence of missed breeding relatively high at about 0.35, juvenile survival 0.798, breeding success generally around 0.9 chicks per pair, natal dispersal 0.629, adult dispersal uncertain (Horswill and Robinson 2015). JNCC SMP data show breeding success in Scotland mostly around 0.6 chicks per pair (excluding urban colonies so possibly not representative of all pairs), with no clear long term trend between 1989 and 2018 (JNCC 2020). Adult survival has been monitored at one colony in Wales from 1986 to 2018 and has shown a declining trend, and an average of 0.82 (JNCC 2020). Rock and Vaughan (2013) also found a decline in adult survival of urban-nesting herring gulls in Bristol, from above 0.9 in the early 1990s to <0.7 by 2007, suggesting that declines in some populations may be driven by declining adult survival. Bosman et al. (2016) measured juvenile survival of herring gulls through their first year from a colony in Belgium as 0.45 in females and 0.38 in males, much lower values than presented in Horswill and Robinson (2015). Using seven years of capture-mark-recapture data, Robertson et al. (2016) estimated survival of adult herring gulls in eastern Newfoundland as 0.837, but noted that this relatively low survival rate may be influenced by gull control programmes and the poor quality habitat for this species.

4.11.3 Ecology and threats to population

Herring gull is a misnomer. In German, heringsmöwe is the lesser black-backed gull, and that makes more sense as lesser black-backed gulls feed extensively at sea on pelagic fish such as herring. Our herring gull does not. Herring gulls from the UK feed more in the intertidal zone, on crabs and other intertidal invertebrates, than they do out at sea. Herring gulls also feed extensively inland on a wide range of foods. Flocks of scavenging seabirds behind fishing boats contain many herring gulls when close to the shore, but increasingly fewer further out to sea. However, the northern subspecies of herring gull which comes to the UK in winter from arctic Norway and Russia, appears to be more marine in habit than our birds. UK herring gulls remain near home all year round. They will breed in colonies similar to those selected by lesser black-backed gulls, but also nest in loose small groups along cliff-tops and on grassy areas at the foot of cliffs or part way down a cliff. Some nest on rocky outcrops as individual pairs rather than being in a colony. Increasing numbers have taken to roof-top nesting in urban areas, where there breeding success tends to be high, probably in part because those nests are safe from mammal predators but also are usually safer from conspecific attack which is often seen in dense ground-nesting herring gull colonies, where cannibalism of chicks can be frequent.

Woodward et al. (2019) list the foraging range of breeding herring gulls as mean 14.9 km, mean maximum 58.8 km, maximum 92 km. However, these distances are likely to apply more along the coast than directly out to sea, given the preference of UK breeding herring gulls to remain near the coast.

Culling of large numbers of breeding herring gulls has been carried out over many years and has undoubtedly reduced numbers at many colonies (Mitchell et al. 2004). 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. Large numbers of licences are still being issued to allow continued control of herring gulls, but licences now require reporting, which should in future provide better understanding of the extent to which this control is driving population decline. Breeding success can be strongly reduced by foxes, mink, and rats (Mitchell et al. 2004). A few colonies are susceptible to flooding and to outbreaks of botulism (Mitchell et al. 2004). Although herring gulls can feed on a wide range of foods, individual birds tend to specialise. Food availability has almost certainly decreased as a result of altered management of landfill sites and sewage discharge into coastal seas, which may both have contributed to declines in numbers (Mitchell et al. 2004). Reduced quantities of discards and offal have probably also affected herring gull numbers (Mitchell et al. 2004, Foster et al. 2017). Herring gull flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.11.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified four potential measures that were likely to improve the conservation status of herring gulls.

1. Mink eradication at herring gull colonies;
2. Fencing out foxes from colonies;
3. End culling of herring gulls; and
4. Eradicate rats at herring gull colonies.

All four of these potential measures were considered highly likely to be effective with high confidence in that assessment based on evidence. Literature published since 2013 does not seem to 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 mainland 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 consequences 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.

The amount of culling of herring gulls has probably decreased since 2013. In 2019, herring gull was removed from the General Licence, so this species can no longer be killed without a specific licence. While specific licences are being issued in large numbers to permit control of herring gulls, this does mean that numbers being killed will now be monitored by SNCBs so that the extent of legal control can be assessed. It probably means that less killing of gulls will be taking place, although that is difficult to assess because no record of numbers killed under General Licence was required.

4.12 Great black-backed gull *Larus marinus*

4.12.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified great black-backed gull as Green, Amber, and Amber respectively. Birds Directive migratory species.

The biogeographic population (Europe excluding Russia) was estimated at 105,000 pairs, of which 17,000 pairs breed in Great Britain and 2,300 pairs in all-Ireland (Mitchell et al. 2004). National surveys indicate a decline in UK breeding numbers of 7% between 1969 and 1986, and of 4% between 1986 and 2000 (JNCC 2020). JNCC SMP data indicate no trend in UK breeding numbers between 1986 and 2016 but suggest a steep decrease from 2016 to 2018 (JNCC 2020). However, data from Scotland indicate a large decrease in breeding numbers between 2002 and 2018, with the index in 2018 at less than 40% of the values in 1986 and 2002 (JNCC 2020). In contrast, breeding numbers in England, Wales, and Northern Ireland have increased over the same period (JNCC 2020).

Stroud et al. (2016) identified that the SPA suite with breeding great black-backed gull as a designated feature has six qualifying sites in Great Britain, five in Scotland (Calf of Eday SPA; Copinsay SPA; East Caithness Cliffs SPA; Hoy SPA; North Rona and Sula Sgeir SPA), one in England (Isles of Scilly SPA). The SPAs in Great Britain were estimated to hold 16.8% of the Great Britain breeding population of great black-backed gulls present in 2000 (Stroud et al. 2016).

Table 11. Summary of great black-backed gull breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Calf of Eday	938	60 in 2018	-878	**-94**	UD
Hoy	570	50 in 2018	-520	**-91**	UD
Copinsay	490	84 in 2015	-406	**-83**	UD
East Caithness Cliffs	800	266 in 2015	-534	**-67**	UNc
North Rona and Sula Sgeir	730	191 in 2012	-539	**-74**	UD
Isles of Scilly	ca 900	Citation count	n/a	**n/a**	

All Scottish SPA sites show large decreases in great black-backed gull breeding numbers (Table 11) and decreases that appear to be greater than the national average for the species based on JNCC SCM data. Trend in the Isles of Scilly is unclear as the SPA citation uses counts from 2015 which are the most recent complete counts for the archipelago. However, there were 999 pairs in 1985-1988 so this suggests only a small decline at this site.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding great black-backed gulls (Stroud et al. 2016).

4.12.2 Demography

Adult survival 0.93 but data limited, age of first breeding 5 years, incidence of missed breeding uncertain, immature survival uncertain, breeding success generally around 1.1 chicks per pair, natal and adult dispersal uncertain (Horswill and Robinson 2015). Using seven years of capture-mark-recapture data, Robertson et al. (2016) estimated survival of adult great black-backed gulls in eastern Newfoundland as 0.864 but noted that this relatively low survival rate may be influenced by gull control programmes and the poor quality habitat for this species.

4.12.3 Ecology and threats to population

Great black-backed gull is more predatory than herring gull or lesser black-backed gull. In many areas great black-backed gulls nest as widely distributed pairs along cliff-tops or on grassy slopes part way down a cliff, and feed on eggs, chicks and adults of a range of local seabird species. They can catch seabirds such as puffins and turn them inside-out, leaving the skin as evidence. At petrel and shearwater colonies they can catch and eat adults at night, casting up pellets containing petrel remains. At large colonies most pairs feed on fishery discards rather than seabirds, because large numbers cannot be sustained just by killing other seabirds. After breeding, great black-backed gulls from UK colonies disperse but do not move far. However, conspecifics from arctic Norway and Russia migrate to the North Sea to spend winter here. Some of those birds join UK birds in terrestrial feeding areas such as landfill sites, but it seems that most migrants tend to remain more at sea, coming to headlands or remote islands to rest when not following fishing boats. Great black-backed gulls are especially successful at obtaining discards because of their large size, so can outcompete fulmars, herring gulls, lesser black-backed gulls and kittiwakes.

Woodward et al. (2019) list the foraging range of breeding great black-backed gulls as mean 16.7 km, mean maximum 73 km, maximum 73 km. However, these distances are likely to apply more along the coast than directly out to sea, given the preference of UK breeding great black-backed gulls to remain near the coast.

Culling of large numbers of breeding great black-backed gulls has been carried out over many years and has undoubtedly reduced numbers at many colonies (Mitchell et al. 2004). 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. Large numbers of licences are still being issued to allow continued control of great black-backed gulls, but licences now require reporting, which should in future provide better understanding of the extent to which this control is driving population decline, although many birds are shot by crofters and farmers to reduce their impact on lambs in spring and that activity in many cases is not licenced but represents traditional management in remote areas. Outbreaks of botulism in great black-backed gulls have been reported, especially in Ireland, but also in Wales (Mitchell et al. 2004). Breeding success can be reduced by mink and fox predation. Increases in great skua numbers appear to have contributed to declines in great black-backed gull breeding numbers at some colonies, such as Hoy, probably through competition for food and killing of gull chicks by skuas (Mitchell et al. 2004). Reductions in fishery discards are likely also to have contributed to the decline in numbers. Mitchell et al. (2004) stated '*it is probable that productivity during the breeding season and increased winter survival both increased as a consequence of feeding on discards*'. Removal of this food subsidy is likely to reverse that, leading to declines in great black-backed gull numbers, especially at the largest colonies, which are almost all SPAs for this species. Great black-backed gull flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.12.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified five potential measures that were likely to improve the conservation status of great black-backed gulls.

1. End culling of great black-backed gulls;
2. Mink eradication at great black-backed gull colonies;
3. Fencing out foxes from colonies;
4. Eradicate rats at great black-backed gull colonies; and
5. Closure of sandeel and sprat fisheries close to colonies.

The first four of these potential measures were considered highly likely to be effective with high confidence in that assessment based on evidence. There was a lack of clear evidence that this species would benefit from closure of sandeel and sprat fisheries. There was therefore only low confidence in the efficacy of closure of sandeel and sprat fisheries for great black-backed gull breeding success because although this species does feed on sandeels while breeding, it apparently does not rely heavily on sandeels but takes a diverse diet with much food derived from fishery discards and from predation on seabirds, rabbits and other animals as available in its locality.

Literature published since 2013 does not seem to 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. The amount of culling of great black-backed gulls has probably decreased since 2013. In 2019, great black-backed gull was removed from the General Licence, so this species can no longer be killed without a specific licence, except where killing is to prevent serious damage to agriculture (as for example in cases where great black-backed gulls kill newborn lambs). While specific licences are being issued to permit control of great black-backed gulls, this does mean that numbers being killed will now be monitored by SNCBs so that the extent of legal control can be assessed. It probably means that less killing of gulls will be taking place, although that is difficult to assess because no record of numbers killed under General Licence was required. The huge decreases in breeding numbers of great black-backed gulls at SPAs where they are a feature (Table 11) does not necessarily mean that the killing of this species will have ceased in the vicinity of those colonies. Crofters recognise that just a few individual great black-backed gulls can represent a significant threat to lambs, and so any bird attending lambing parks in spring may be considered to be unacceptable. There is a clear difficulty in establishing compensation measures for this species given the widespread view that it is an agricultural pest in much of Scotland and especially in areas of crofting with sheep kept on the hill.

4.13 Kittiwake *Rissa tridactyla*

4.13.1 Conservation status and populations

IUCN Red List classification “Vulnerable”. BOCC 2, 3, and 4 classified kittiwake as Amber, Amber, and Red respectively. Birds Directive migratory species. OSPAR list of threatened and/or declining species.

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). National surveys found an increase in breeding numbers in the UK of 25% between 1969 and 1986, 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 declined, but much less than in Scotland, reaching 68 in 2018 (JNCC 2020). Trends in Wales are similar to those in England.

Stroud et al. (2016) identified that the SPA suite with breeding kittiwake as a designated feature has 32 qualifying sites in Great Britain, 29 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; 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), one in Wales (Skokholm, Skomer and Middleton Island SPA). 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). One site in Northern Island also qualifies (Rathlin Island).

Table 12. Summary of kittiwake breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Hermaness, Saxa Vord & Valla	922	200 in 2016	-722	**-78**	UD
Foula	3,840	308 in 2020	-3,532	**-92**	UD
Noss	7,020	76 in 2019	-6,944	**-99**	UD
Sumburgh Head	1,366	241 in 2017	-1,125	**-82**	UD
Fair Isle	18,160	859 in 2015	-17,301	**-95**	UD
West Westray	23,900	2,755 in 2017	-21,145	**-88**	UD
Rousay	4,900	330 in 2016	-4,570	**-93**	UD
Calf of Eday	1,717	142 in 2018	-1,575	**-92**	UD
Copinsay	9,550	955 in 2015	-8,595	**-90**	UD
Hoy	3,000	n/a	n/a	**n/a**	UD
Marwick Head	7,700	906 in 2018	-6,794	**-88**	UD
North Caithness Cliffs	13,100	n/a	n/a	**n/a**	UD

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
East Caithness Cliffs	32,500	24,460 in 2015	-8,040	**-25**	FM
Buchan Ness to Collieston	30,452	11,295 in 2019	-5,992	**-20**	UNc
Troup, Pennan & Lion's Heads	31,600	10,503 in 2017	-21,097	**-67**	UNc
Fowlsheugh	36,650	9,444 in 2018	-27,206	**-74**	(FM)
Forth Islands	8,400	3,661 in 2018	-4,739	**-56**	UD
St Abb's Head to Fast Castle	21,170	5,000 in 2019	-16,170	**-76**	UD
Farne Islands	8,241	4,402 in 2019	-3,839	**-47**	
Flamborough & Filey Coast	44,520	51,535 in 2017	7,015	**+16**	
North Rona & Sula Sgeir	5,000	1,253 in 2012	-3,747	**-75**	UD
Cape Wrath	9,700	3,622 in 2017	-6,078	**-63**	UD
Handa	10,732	3,749 in 2018	-6,983	**-65**	UD
Flannan Isles	2,780	1,052 in 1998	-1,728	**-62**	UD
St Kilda	7,830	420 in 2015	-7,410	**-95**	UD
Shiant Isles	1,800	1,075 in 2015	-725	**-40**	UNc
Mingulay & Berneray	8,600	2,878 in 2014	-5,722	**-67**	UR
Canna & Sanday	930	1,457 in 2019	+527	**+57**	FM
Rum	1,500	788 in 2000	-712	**-47**	UNc
North Colonsay & Western Cliff	4,512	3,380 in 2016	-1,132	**-25**	FM
Ailsa Craig	3,100	300 in 2019	-2,800	**-90**	UD
Rathlin Island	6,822	7,922 in 2011	+1,100	**+16**	
Skomer, Skokholm & seas	4,472	>1,236 in 2018	n/a	**n/a**	

Kittiwakes are in Unfavourable conservation status at most UK sites designated for breeding kittiwakes (Table 12). Declines in breeding numbers are especially large in Shetland and Orkney (average of 90% since designation) but are less for the colonies furthest south.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding kittiwakes (Stroud et al. 2016). However, since then Outer Firth of Forth and St Andrews Bay Complex SPA, classified on 3 December 2020, includes kittiwake as a breeding season feature.

4.13.2 Demography

Adult (2 years and older) survival 0.854 (s.d. 0.051), age of first breeding 4 years, incidence of missed breeding about 0.19, juvenile survival 0.79, breeding success generally around 0.7 chicks per pair, natal dispersal 0.89, adult dispersal considered to be low at 0.01 for increasing populations but 0.06 for decreasing populations (Horswill and Robinson 2015). JNCC SMP data show average breeding success in Scotland to have declined from 0.9 chicks per pair in 1986 to less than 0.3 chicks per pair in 2008, but to have recovered slightly since 2008 to 0.6 chicks per pair in 2018 (JNCC 2020). However, the national average for Scotland hides regional variation, with much lower breeding success in Shetland and Orkney than in east Scotland or west Scotland. Breeding success in England shows a long-term decline from 1 chick per pair in 1987 to under 0.6 chicks per pair in 2018 (JNCC 2020). Breeding success in Wales shows a long-term decline similar to that in England (JNCC 2020). Breeding success in Northern Ireland has been higher, but very variable (JNCC 2020). Return rates of marked adults (not quite the same as survival rates) have been monitored at Isle of May and Skomer. At Isle of May return rate fell from 95% in 1987 to 65% in 2008, but then recovered to over 85% in 2018, showing a similar pattern to breeding success (JNCC 2020). At Skomer estimated survival showed little long-term trend but was around 85% in most years from 1986 to 2018 (JNCC 2020). McKnight et al. (2018) used an experimental approach to show that forced breeding failure in kittiwakes increased the likelihood of these birds breeding in subsequent years but did not affect long-term survival rate of the individuals. This provides evidence that reproduction has a cost for adults, but in this case the cost was in terms of the chances of birds having to take a year off from future breeding to recover condition, rather than a clear influence on adult survival.

4.13.3 Ecology and threats to population

The kittiwake is a small gull that is particularly marine. It mainly nests relatively low down on steep cliffs, though colonies can occur on waterside buildings and walls in places where natural cliff sites are unavailable. At almost all North Sea colonies in the UK, kittiwakes feed their chicks primarily on sandeels. A few small colonies are in areas where adults can feed on sprats (such as in the inner Firth of Forth), and sprats can be important in the diet of this species in the Irish Sea. Kittiwakes can also feed on zooplankton such as euphausiids and will scavenge offal and scraps of fish from fishing boats, especially in winter. British kittiwakes mostly winter in Canadian or mid-North Atlantic waters, although a small proportion of the population remains in British waters all year round (Frederiksen et al. 2012). Kittiwakes from other European and high latitude colonies in the North Atlantic and Barents Sea may pass through British waters on migration, but most of these birds also winter in the western North Atlantic in the same general area used by British birds (Frederiksen et al. 2012). A huge increase in kittiwake numbers in the UK between 1900 and 1985 was attributed to reduction in persecution combined with changes in fish stocks; overexploitation of large predatory fish led to increases in stocks of small forage fish, such as sandeels.

Woodward et al. (2019) list the foraging range of breeding kittiwakes as mean 54.7 km, mean maximum 156 km, maximum 770 km. These estimates include data from many colonies where kittiwakes were tagged during the RSPB FAME and STAR projects with tags that were above the limiting weight considered appropriate for seabirds, and there is evidence that birds equipped with relatively heavy tags travel further from the colony than normal birds, probably because they are less able to compete for food with unencumbered conspecifics closer to the colony. Therefore, these foraging range estimates may exaggerate the extent to which breeding kittiwakes make long foraging trips. Bolton et al. (2019) showed that in many seabird species birds forage almost exclusively in the 'domain' around their colony and avoid extensive overlap with conspecifics from neighbouring colonies. This seems to apply to kittiwake, which means that impacts would most appropriately be attributed to the closest colony rather than to all colonies within theoretical foraging range limits.

The main threats to kittiwake are considered to be impacts of climate change (because sandeel productivity declines with increasing sea temperature, and because warming seas have led to a reduction in the abundance of large zooplankton that store lipid so influence the whole food-web) and harvesting of sandeels, which reduces sandeel abundance. That directly affects breeding success of kittiwakes and reduces adult survival too (Oro and Furness 2002, Frederiksen et al. 2004, Mitchell et al. 2004). However, in northern Scotland, kittiwakes are also under pressure from great skua predation (Votier et al. 2001, Oro and Furness 2002). Red tides, oil pollution, and winter storms may also contribute to kittiwake mortality, although incidence of oil pollution has declined over decades so is unlikely to be a significant contributor to recent declines. However, the much greater decline of kittiwakes in the northern North Sea than in the southern North Sea may also reflect the fact that recovery of large predatory fish (adult herring, mackerel, cod, haddock, whiting, etc) has primarily occurred in the northern North Sea and this increases predation impact on sandeels and other forage fish considerably, so probably represents a top-down impact that adds to the bottom-up impact of climate change and the impact of sandeel fishing. Kittiwake flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.13.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified seven potential measures that were likely to improve the conservation status of kittiwakes.

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.

The first two of these potential measures were considered highly likely to be effective with high confidence in that assessment based on evidence. There was a lack of clear evidence that this species would benefit from measures 3 to 7. Since 2013, evidence that has become available continues to indicate that measures 3 to 7 are not likely to be suitable as compensation for impacts on kittiwake except possibly in a very few very limited locations.

Kittiwake is the seabird species where the likelihood of requiring compensation for impacts of offshore wind appears to be highest at present, and probably in the near future. Several projects have already developed and published some details of their compensation plans for kittiwake. These include Hornsea Three (Orsted 2020a-f) and Norfolk Boreas (Royal Haskoning DHV 2020). In both these cases, the compensation plan is based on provision of artificial structures for new kittiwake colonies. While the potential efficacy of closure of sandeel fisheries in UK waters has been recognised in these published plans, that approach has been seen as too difficult for developers to take forward because such closures would require legislation at a national and/or international level, so are beyond the remit of offshore wind farm developers. This raises the point that a strategic approach by government would be more appropriate. Therefore, these measures and new evidence relating to these, specifically in relation to kittiwake, are considered here, but the route of possible strategic compensation led by government, is considered later after these species-specific texts, particularly as no-take zones for sandeels would influence prey availability to many species of seabirds and not just kittiwake.

New evidence on closure of sandeel and sprat fisheries in UK waters in relation to kittiwake:

During the breeding season, kittiwakes breeding at most colonies around the North Sea feed mainly on sandeels (Furness and Tasker 2000, Coulson 2011). Sandeel abundance strongly influences breeding success of kittiwakes (Frederiksen et al. 2004, Cury et al. 2011, Carroll et al. 2017, Christensen-Dalsgaard et al. 2018), and breeding success strongly influences whether kittiwake colonies increase or decrease in breeding numbers (Monnat et al. 1990, Cadiou et al. 1994, Coulson 2011, 2017). In Shetland, kittiwake breeding success, and breeding numbers, decreased dramatically after the collapse of the Shetland sandeel stock (Furness and Tasker 2000). At Foula, kittiwake breeding success shows a strong relationship with Shetland sandeel total stock biomass (Figure 3). Kittiwake breeding success was much lower in most years of sandeel biomass below 40,000 tonnes but was high in almost all years when sandeel biomass was above that level.

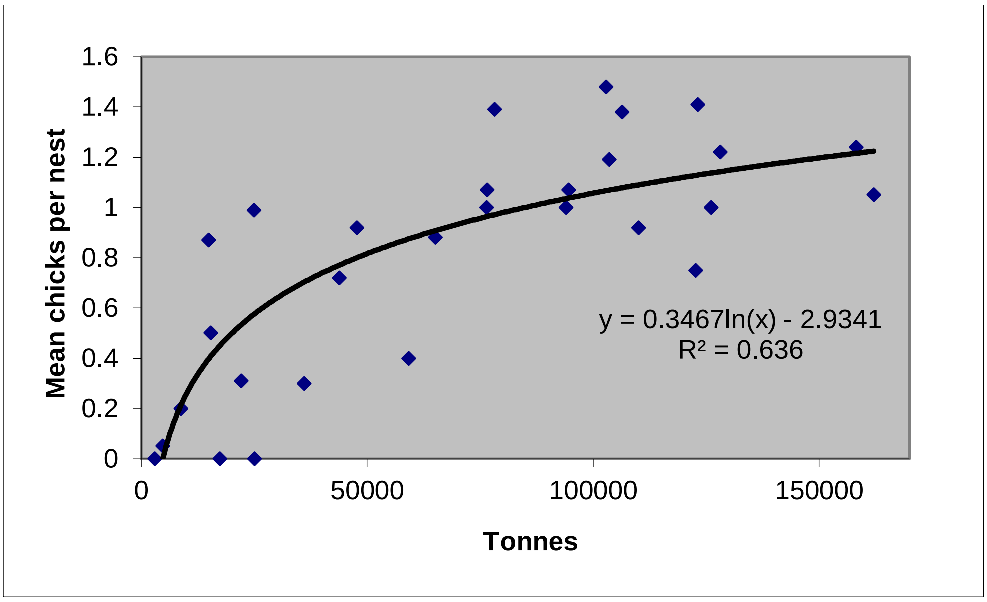

Figure 3. Breeding success of black-legged kittiwake at Foula, Shetland, in relation to the Shetland sandeel total stock biomass for the years 1976 to 2004

Kittiwake breeding success has also been affected at the Isle of May, when the sandeel stock in that area (which is distinct from the sandeel stocks at Shetland or in the southern North Sea; Frederiksen et al. 2005, ICES 2017, Olin et al. 2020) was heavily fished (Frederiksen et al. 2004). Sandeels are the target of what has been the largest single-species fishery in the North Sea over recent decades, with that fishery now particularly concentrated on the sandeel stock on Dogger Bank. Kittiwakes at Flamborough and Filey Coast SPA (FFC SPA), now the largest kittiwake colony in the British Isles, forage over a large area from that colony, and their foraging area includes some of the most important sandbanks supporting high densities of sandeels and the sandeel fishery (Carroll et al. 2017). There is strong evidence that the sandeel fishery has caused depletion of sandeel biomass in this region (Lindegren et al. 2018), and that reduced abundance of sandeels as a result of the high fishing effort on sandeels has led to reduced breeding success of kittiwakes at FFC SPA (Carroll et al. 2017). Reducing the level of fishing effort on sandeels or closing the fishery in waters close to the colony, would, therefore, represent mechanisms to improve breeding success of kittiwakes at that colony by making it possible for the biomass of the sandeel stock to recover from the high fishing mortality that has been imposed in recent decades. Reduction in fishing mortality would be anticipated to lead to rapid, though probably incomplete, recovery of sandeel abundance (Lindegren et al. 2018). Sandeel is a short-lived fish which starts to breed when only 1 or 2 years old, with high reproductive potential, and since kittiwakes will feed on all age classes of sandeels but especially on 1 and 2 year old sandeels, the increase in sandeel abundance would be likely to influence kittiwake breeding success with a time lag of only 1 or 2 years.

Frederiksen et al. (2004) showed that breeding success of kittiwakes at the Isle of May (part of Forth Islands SPA) 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 (Figure 4). Adult survival was also lower during years with sandeel fishing in the area (Frederiksen et al. 2004).

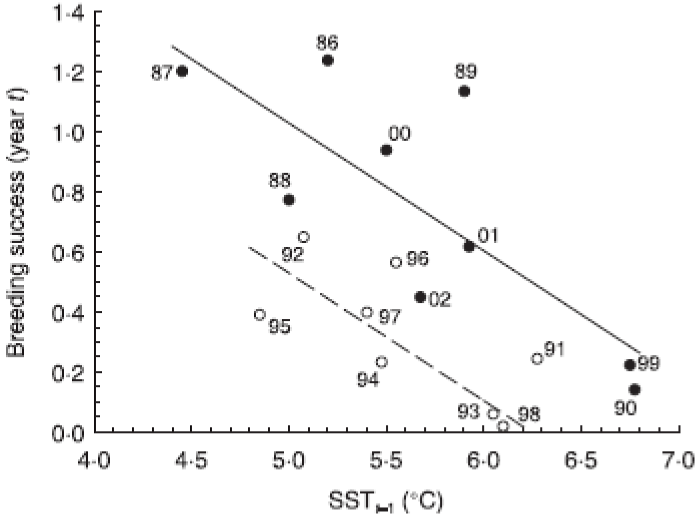

Figure 4. Kittiwake breeding success at the Isle of May in relation to local Sea Surface Temperature in February-March of the previous year, and the presence (open circles and dashed line) or absence (black dots and solid line) of a sandeel fishery off east Scotland. From Frederiksen *et al.* (2004).

A decision was taken to close an area to sandeel fishing (the 'sandeel box' off the east of Scotland) because of persistent low breeding success of kittiwakes indicative of the poor condition of the sandeel stock in the area. The consequence of that closure was monitored. 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), providing experimental evidence for the mitigation of fishery impact by closing the fishery. Recovery of sandeel abundance in the closed area has led to the sandeel fishing industry seeking the opportunity to resume fishing within the closed area, but until now the regulator has retained this closed box, although fishing for sandeels has occurred right up to the offshore (eastern) edge of the closed box.

Closure of the sandeel fishery off east Scotland also altered the age structure of the sandeel population. When the stock was heavily fished, very few sandeels lived beyond two years old, resulting in high variability on stock abundance from year to year depending on the highly variable level of production of young fish. When the fishery was closed, sandeels tended to live longer, with large cohorts remaining in the stock for up to six years (Peter Wright, pers. comm.). The longer life expectancy of sandeels when not subject to fishing not only increases mean biomass of the stock, but also reduces variability in abundance driven by variable recruitment. This in turn will also be beneficial to kittiwake breeding success, by ensuring that even if sandeel recruitment is poor, the biomass of the stock is buffered by presence of several older age classes of fish.

The abundance of sandeels in ICES area 4 (which includes the sandeel no-take box off east Scotland) declined during 1993-2001 (Figure 5). However, after the closure of the sandeel fishery off east Scotland, this stock eventually recovered (Figure 6).

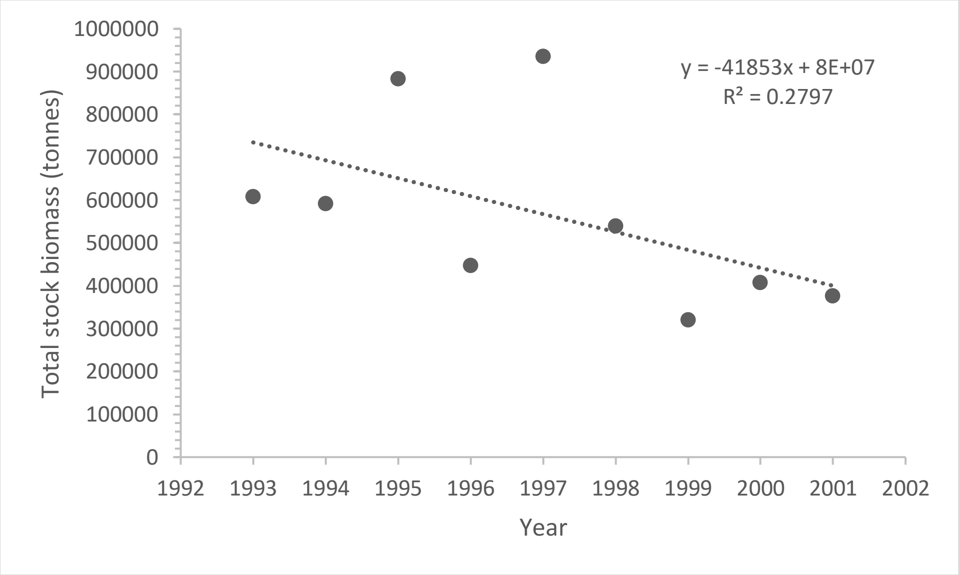

Figure 5. Abundance (total stock biomass in tonnes) of sandeels in ICES area 4 (which includes the no-take zone off east Scotland that was established in 2000) in the period 1993 to 2001. Data from ICES (2020).

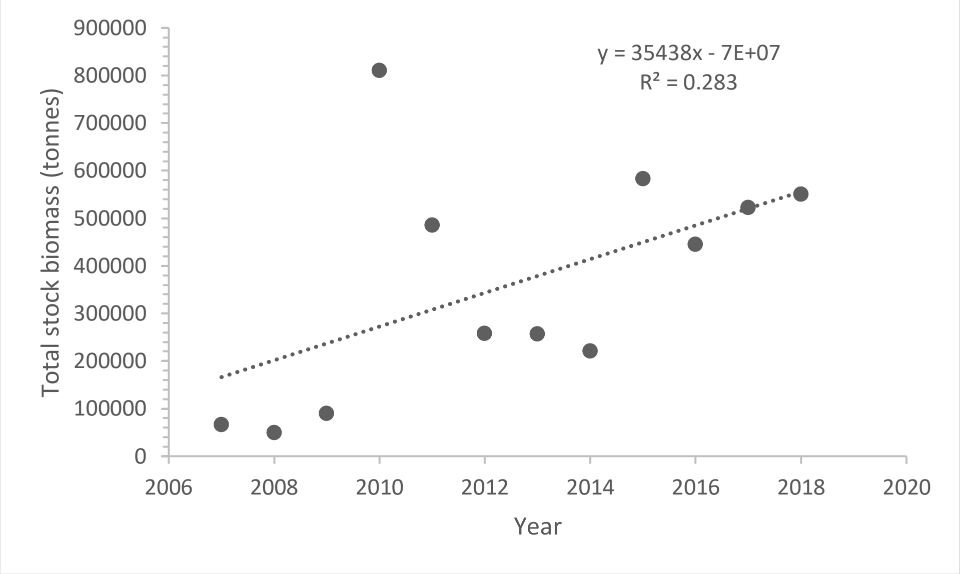

Figure 6. Abundance (total stock biomass in tonnes) of sandeels in ICES area 4 (which includes the no-take zone off east Scotland that was established in 2000) in the period 2007 to 2018. Data from ICES (2020).

Sandeel catches from ICES area 4 decreased considerably when the sandeel no-take box was established off east Scotland, but a large part of ICES area 4 remained open to sandeel fishing. Commercial catch from the open part of area 4 was low in 2005 to 2012 because the stock biomass had been depleted and so commercial fishing was no longer sufficiently profitable to justify fishing in that area (while better catches could still be obtained elsewhere such as area 1r Dogger Bank). However, as sandeel stock began to recover in area 4 the potential profitability of fishing there increased. Commercial catches have increased considerably in the last few years (Figure 7). The return to high fishing effort on sandeel in area 4 threatens to impact this recovering stock again, with potential effects on kittiwake breeding success in east Scotland.

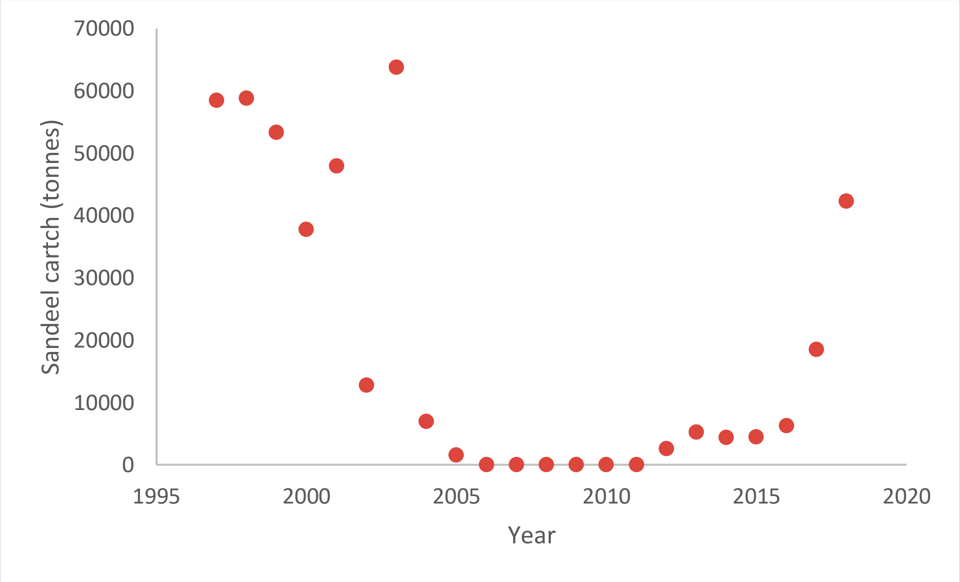

Figure 7. Catch (tonnes) of sandeel by the commercial sandeel fishery in ICES area 4 from 1997 to 2018 (data from ICES 2020). Note that the sandeel no-take box was established in 2000 but fishing in parts of area 4 that are outside the box was permitted throughout the period and the low catch from 2005 to 2015 was due to low stock biomass and not just to the existence of the no-take box in part of area 4.

It is also interesting to note that kittiwake breeding success and adult return rate from the previous year (an index of adult survival rate but not corrected for birds missed in that year but that returned in later years, so an underestimate of true survival) at the Isle of May, which is monitored every year by UKCEH, averaged 0.88 chicks/pair and 0.81 in 2011-2016 (data from UKCEH annual reports on Isle of May seabird studies) when the sandeel stock biomass in ICES area 4 was generally above 200,000 tonnes, but averaged only 0.44 chicks/pair and 0.75 in 2004-2010 (data from UKCEH annual reports on Isle of May seabird studies), when sandeel stock biomass was generally below 200,000 tonnes. This is further evidence that sandeel stock biomass is a major driver of kittiwake breeding success and also affects adult survival, as originally found at Shetland (Oro and Furness 2002). Indeed, the relationship between kittiwake return rates and sandeel stock biomass over the period of recovery of this stock (2004 to 2018) appears remarkably strong (Figure 8).

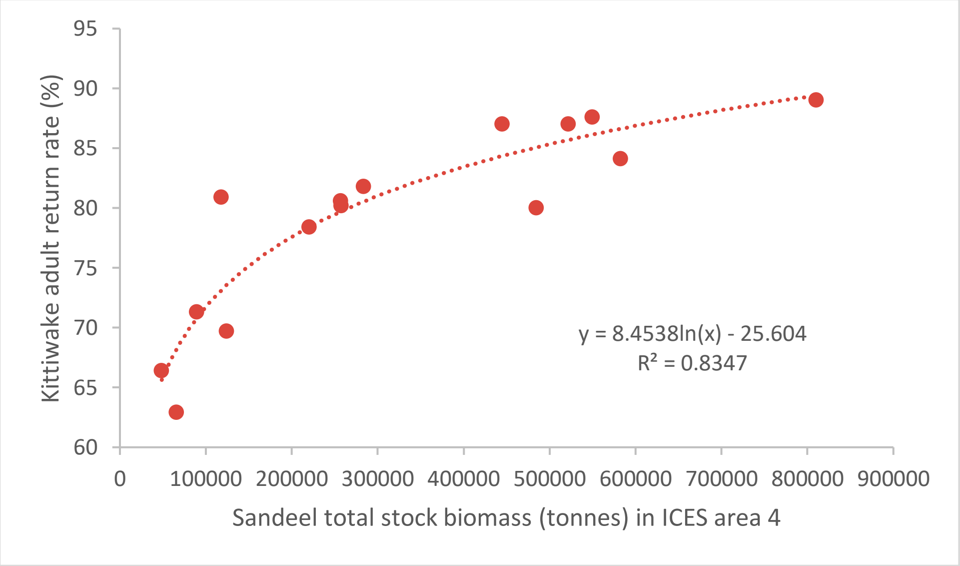

Figure 8. Annual return rates of adult kittiwakes at the Isle of May (UKCEH data from online annual reports) in relation to ICES estimates of annual sandeel total stock biomass (tonnes) in area 4 between 2004 and 2018. Data for 2019 and 2020 are not yet published.

The productivity of kittiwakes at FFC SPA is significantly correlated with sandeel stock biomass. The relationship found by Carroll et al. (2017) for kittiwakes at FFC SPA in relation to sandeel stock in ICES North Sea sandeel management Area 1r ('Dogger Bank' and neighbouring areas) is similar to that previously identified elsewhere: kittiwake breeding success and adult survival at Shetland was closely related to changes in sandeel stock biomass in that area (Furness and Tasker 2000, Oro and Furness 2002, Furness 2007), and kittiwake breeding success at the Isle of May was strongly influenced by effects of sea surface temperature and sandeel fishing on the sandeel stock off the Firth of Forth, east Scotland (Frederiksen et al. 2004).

Christensen-Dalsgaard et al. (2018) showed that kittiwake breeding success was higher at a colony in Norway when kittiwakes were able to feed chicks on sandeels but was much reduced in years when sandeel availability was lower and kittiwakes had to switch to feeding chicks on mesopelagic fish.

Lindegren et al. (2018) carried out a hindcast analysis of the Dogger Bank sandeel stock to assess the consequence of the high fishing mortality. They estimated that sandeel spawning stock biomass would have been about twice as large now as it is, if the fishery had maintained fishing mortality (F) at F=0.4 rather than at the levels of F=0.8 to 1.2 as seen during 1999-2009 in the history of this fishery. Indeed, the stock would be even larger now if there had been no fishery harvesting sandeels, although Lindegren et al. (2018) did not report on that scenario. However, their results further support the conclusion that the high fishing mortality imposed on the sandeel stock has been a major influence on the abundance of the sandeel, and hence on the breeding success of kittiwakes. Lindegren et al. (2018) also identified influences of sea temperature and copepod abundance on the abundance of sandeels and suggested that long term trends in those drivers may inhibit recovery of sandeels if fishing pressure was reduced. In addition, severe reduction in forage fish stock biomass can lead to increased natural mortality that may inhibit recovery, and there is evidence of this with sandeel declines to low biomass (Saraux et al. 2020). At present, the Dogger Bank sandeel stock remains considerably below its long term average abundance and is subject to a fishing mortality around F=0.6 (ICES 2020), a figure above the level tested in the scenario of Lindegren et al. (2018), and a figure which their scenario modelling clearly demonstrates has a negative impact on sandeel abundance. Indeed, at present the spawning stock biomass in this area is less than 10% of its highest historical level and is slightly below the limiting spawning stock biomass at which ICES should recommend closure of the fishery (B_{lim} of 110,000 tonnes SSB) because there is an increased risk of recruitment failure in this stock (ICES 2020).

Off east Scotland, there is still fishing for sandeels, but that is limited to outside the sandeel box. However, the modelling by Lindegren et al. (2018) suggests that closure of sandeel fishing in the area of UK waters beyond the existing sandeel box would contribute to recovery of the sandeel stock biomass throughout that area, so could be a compensation measure that would be effective for kittiwakes at colonies in east Scotland.

Cury et al. (2011) used empirical evidence from several seabird-fishery interactions around the world to suggest that management should aim to keep food fish stocks such as sandeels above a threshold of one-third of their historical maximum biomass in order to achieve good productivity among dependent seabird populations. The southern North Sea sandeel stock has fallen far below that rule of thumb management objective. Maximum total stock biomass in ICES area 1r was just below 2,000,000 tonnes during the 1980s at a time of high fishing effort, so likely to be reduced relative to unfished biomass (Lindegren et al. 2018). Nevertheless, if we take 2,000,000 tonnes as maximum biomass for this stock, then the Cury et al. (2011) threshold to avoid impacts on dependent predators such as kittiwakes would be a fished total stock biomass of 666,667 tonnes. Using this rule of thumb, the sandeel fishery has been harvesting from a stock biomass that was below this threshold abundance in 13 of the 16 years 2003-2018 (ICES 2020). The long-term deterioration of this heavily fished stock and its tendency to be below the Cury et al. (2011) threshold in recent years is clear in Figure 9.

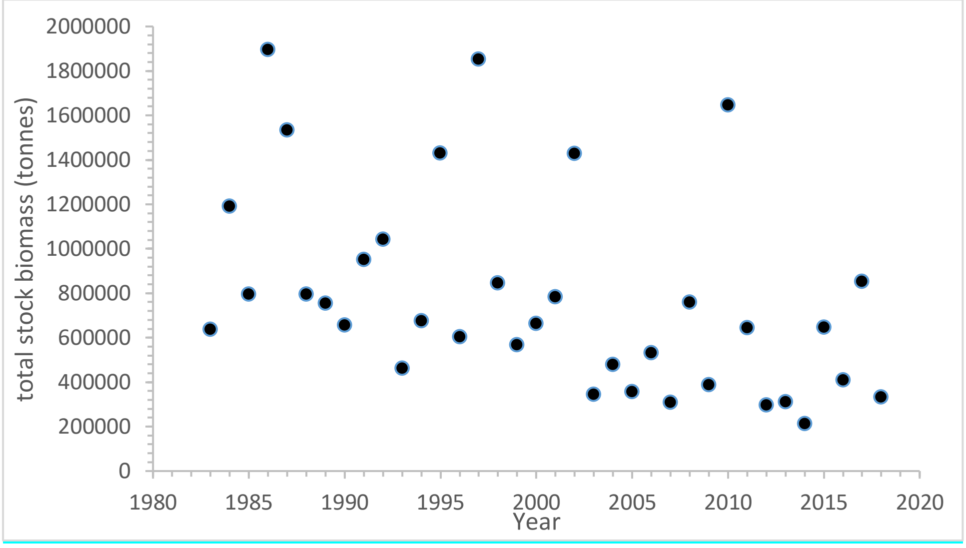

Figure 9. Total stock biomass (tonnes) of sandeels in ICES area 1r (Dogger Bank stock) from 1983 to 2018 (ICES 2020), in relation to the Cury et al. (2011) 'rule of thumb' that stock biomass should be maintained above one-third of the historical maximum (in this case above 666,667 tonnes) to avoid adverse impacts on dependent seabird populations.

This suggests that the continuation of sandeel fishery is likely to continue to cause mortality of many thousands of kittiwake chicks per year compared to a scenario with no fishing of the sandeel stock. It also identifies that the single most effective practical management action to assist the kittiwake population would be closure of the sandeel fishery (Carroll et al. 2017, Lindegren et al. 2018, Wright et al. 2018). Mortality of chicks has less impact on the kittiwake population than the same mortality of adults. On the basis of the demographic parameters of kittiwakes in the North Sea (adult survival 0.854, juvenile survival 0.79, age of first breeding 4 years; Horswill and Robinson 2015), two fledglings would be required, on average, to give rise to one adult surviving to recruit into a local colony at 4 years of age. If sandeel fishing reduced productivity at FFC SPA by an average of 0.5 chicks per pair per year which appears to be approximately the scale of the impact indicated by the data for this region and equals the estimate for the kittiwakes at the Isle of May, that would be equivalent to 50,000 pairs x 0.5 chicks per pair, or 25,000 chicks that die due to scarcity of sandeels. If those chicks had survived to fledge, they would result in about 12,000 adults per year surviving to recruit into colonies at 4 years of age. That is more than an order of magnitude more than losses estimated to be caused by collision mortality at offshore wind farms in the region, so represents a potential for far greater compensation than the precautionary estimate of losses incurred due to offshore wind.

In view of the large numbers of kittiwake chicks dying at Flamborough and Filey Coast SPA as a consequence of reduced abundance of sandeels due to fishing impacts, there is evidently scope for compensation through either reducing fishing effort directed at sandeels, or through closing areas within the main foraging range of this kittiwake population to sandeel fishing. ICES promotes 'ecosystem-based management' of fish stocks. However, their management of the sandeel stock has recently been criticised as not being 'ecosystem-based' because it sets a quota only on the basis of sustaining the sandeel stock and not on the basis of the needs of higher trophic level predators such as kittiwakes (Hill et al. 2019). ICES should therefore be highly receptive to the need to better manage that sandeel stock to avoid adverse impacts on kittiwakes and other top predators. An extension to a proposed fisheries management area or a new proposal would need to be facilitated by the UK Government in allocating appropriate powers to a relevant management body and, potentially, through the delivery of legislation to secure the necessary powers.

The same case applies though to a smaller extent for the separate sandeel stock in ICES area 4; reduced fishing effort on that stock can be expected to result in an increase in sandeel biomass, and a concomitant increase in kittiwake breeding success, but that recovery may fail if fishing effort increases in the part of area 4 that remains open, as seems to be the trend since 2015 (Figure 7).

Sandeel is the main breeding season food of kittiwakes in the North Sea, but sprat can also be taken by birds at a few colonies. During summer, sprats tend to be in shallow marine habitats influenced by freshwater inflows into the sea. Their distribution in the North Sea is predominantly southern, mainly south and east of the Dogger Bank, but spreads over much larger areas when the sprat stock increases in biomass, and then extends as far as Shetland, with concentrations in the Moray Firth, Tay, and Firth of Forth (Heessen et al. 2015, ICES 2020). Similarly, sprat catches tend to focus on the highest sprat density areas in the southern North Sea, but catches are taken as far north as Shetland in years of high stock biomass, as in 2019 (ICES 2020). Diets of breeding kittiwakes are not well known for many colonies, but at most sites where these have been studied there has been an overwhelming dominance of sandeel in kittiwake breeding season diet. Exceptions to this include small colonies of kittiwakes in the upper Firth of Forth and at Lowestoft, where sprats are believed to represent a significant part of breeding kittiwake diet and to support high breeding success at those colonies (Lothian Ringing Group, pers. comm., Mike Swindells, pers. comm.). It is therefore likely that kittiwakes at such colonies would benefit from a sprat no-take zone being established since that would be expected to lead to an increase in sprat stock biomass.

New evidence on provision of artificial structures for new kittiwake colonies:

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 by 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. While artificial nest sites offshore in the southern North Sea may provide similar advantages in terms of proximity to kittiwake feeding grounds and protection from disturbance and predators, at sea artificial colonies would be more expensive to construct, may increase collision risk if located near to offshore wind farms, and would be much more difficult and expensive to monitor to demonstrate effective compensation. For these reasons, this approach may be low priority in terms of suitability for compensation.

Construction of new artificial breeding sites for kittiwakes on the coast has been reviewed in detail by Ørsted (2020a,b,c,d,e,f). These reviews by Ørsted would be appropriate as a template for other plans to use artificial nest sites for kittiwakes as a means of compensation, as the reviews are up to date and thorough. Kittiwakes readily use artificial nest sites where natural sites are not available or are in limited supply. These include harbour walls, buildings as diverse as warehouses, stone bridges, metal bridges, castles, churches, oil and gas platforms, power station water pipes, and purpose-made artificial colony sites to replace buildings being demolished. Breeding success on artificial structures can be at least as high as in natural colonies and can be higher where artificial sites are distant from any large kittiwake colonies, close to food supplies and safe from predators (Christensen-Dalsgaard et al. 2019). In the Norwegian context, Christensen-Dalsgaard et al. (2019) conclude “the 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”.

Hornsea Three has proposed constructing four new artificial colonies for kittiwakes at two sites in the vicinity of Lowestoft to Sizewell, and two sites in the vicinity of the Tees Estuary to south of Seaham. Their plan (Ørsted 2020b) states “The design specifications for the artificial nesting structures are at this stage unconstrained. They may take the form of a bespoke structure or be a modification to an existing building or piece of infrastructure (such as a seawall). Where two structures are located in the same search zone, the intent is that they are different designs to maximise the opportunity for kittiwake to colonise. The structure designs will likely be influenced by landowner negotiations, landscape character, and existing environment of the selected location.” Similar structures, in the same part of England, have been proposed as compensation, if required, by Norfolk Boreas (Royal Haskoning DHV 2020).

To provide successful compensation, new artificial colonies need to be adopted by kittiwakes which were otherwise unable to breed due to lack of nesting opportunities, or to result in improved breeding success relative to existing nearby natural colonies. Pairs breeding at those sites need to produce about 0.8 chicks per nest just to maintain the population at the new artificial site. So only breeding success in excess of 0.8 chicks per nest will represent potential compensation for losses of birds through collision mortality. Artificial sites therefore need not only to be used but need to achieve higher breeding success than at natural colonies of kittiwakes so that the surplus production provides compensation.

A strategic Government-led approach to the creation of high quality artificial colonies for kittiwakes could help ensure coherent and optimal measures were taken, in a way that may be difficult for any individual developer to achieve.

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 Lincolnshire 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 (as in much of the Scottish coastline), 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 as a result 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. The use of artificial nest sites in Scotland should therefore not be completely ruled out just because Scotland has lots of available natural habitat. However, the effectiveness of providing artificial sites in Scotland would be likely to be much less than in locations where natural habitat is lacking or scarce, and this may not be very effective as a compensation measure in places, such as east Scotland, where there are numerous natural colonies already occupying much of the coastline.

New evidence relating to impacts of predators on breeding kittiwakes:

Coulson (2011) states 'predation by mammals on kittiwakes is extremely rare'. Furness et al. (2013) found one instance of rat predation, one of fox, one of feral cat, two of mink, one of peregrine and three of large gull predation affecting kittiwake breeding success at the many colonies monitored by JNCC over many years. However, those rare instances compare with 43 cases where food shortage was identified as the cause of reduced breeding success, so cases of predation are few. Furthermore, some of those reported cases may be mistaken. The one instance of rat impacts was reported at Scilly Isles, but eradication of rats there had no beneficial effect on kittiwakes. Brown rats were eradicated from St Agnes and Gugh islands in the Isles of Scilly in 2013-14 (Heaney 2018). There were immediate signs of recovery and recolonization by seabirds that had been affected by rats. However, kittiwake breeding numbers on St Agnes have decreased since the eradication of rats; none bred there in 2017 or 2018. Kittiwake breeding numbers on Gugh fell to zero before the eradication of rats; 30 pairs bred there in 2017 and 35 pairs in 2018, but they fledged only 9 chicks in 2017 and none in 2018. Overall across the Isles of Scilly, kittiwake breeding numbers fell from around 70 pairs in 2010-2015 to only 30 and 35 pairs in 2017 and 2018. Changes in kittiwake numbers and breeding success since eradication of rats from St Agnes and Gugh were not considered to be related in any way to absence of rats, but rather to be a response to a shortage of food for kittiwakes in the waters around the Isles of Scilly (Heaney 2018). Rats were eradicated from Lundy, Bristol Channel, in 2002-04. While Manx shearwaters showed a strong and immediate response to the removal of rats, as did puffins, guillemots and razorbills (Booker et al. 2019), there is no clear evidence that breeding success or numbers of kittiwakes responded. Kittiwake numbers breeding on Lundy fell from 933 pairs in 1981 to 237 pairs in 2000. After eradication of rats, kittiwake numbers continued to fall, to 148 pairs in 2004, and 127 pairs in 2013, but increased to 238 pairs in 2017 (JNCC 2020). Breeding success in 1986-1996 averaged about 0.6 chicks per nest. After eradication of rats, breeding success of kittiwakes in 11 years between 2007 and 2018 averaged about 0.4 chicks per nest (JNCC 2020). So there is no evidence to suggest any improvement in kittiwake breeding success at Lundy after eradication of rats there, and Booker et al. (2019) state that in their opinion it is very unlikely that eradication of rats from Lundy was of any benefit to the kittiwake colony there. Brown rats were eradicated from Ailsa Craig (Clyde Sea) in 1991 (Zonfrillo 2001). While this has led to recolonization of the island by puffins and black guillemots, spread of the breeding area occupied by razorbills, and large increases in breeding success of fulmars, herring gulls and lesser black-backed gulls, there is no evidence that breeding success of kittiwakes was influenced by the eradication of rats. B. Zonfrillo (in litt.) reports that he only ever observed one instance of a kittiwake chick that had been killed by rats on Ailsa Craig, at an unusually accessible nest in the 1970s. In his opinion, eradication of rats has had no influence on kittiwake numbers or breeding success at Ailsa Craig. Brown rats were eradicated from Canna (Inner Hebrides) in 2008 (Bell et al. 2011). In this case too, there were immediate signs of recovery and recolonization by seabirds that had been affected by rats at Canna. However, Bob Swann, who has led the seabird monitoring work at Canna for 50 years, reports that he had no evidence of rats ever targeting kittiwake nests at Canna prior to the eradication. He considers that kittiwake nests at Canna would have been impossible for rats to access, and that eradication of rats from Canna has had no direct influence on the kittiwake population there (R.L. Swann, in litt.). This is the same conclusion as reached by Luxmoore et al. (2019) for this colony. There may be a very few particular situations where control of predators might benefit kittiwake breeding success. One example may be making the artificial colony at Lowestoft harbour inaccessible to foxes. However, such instances appear to be exceptionally rare.

4.14 Sandwich tern *Sterna sandvicensis*

4.14.1 Conservation status and populations

IUCN Red List classification “Least Concern”. BOCC 2, 3, and 4 all classified Sandwich tern as Amber. Birds Directive Annex 1 and migratory species.

The biogeographic population (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). However, JNCC SMP data suggest a decline in Scotland, the index in 2018 being about half that in 1986, though based on small sample sizes (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).

Table 13. Summary of Sandwich tern breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Loch of Strathbeg	280	0 in 2015	-280	**-100**	UNc
Ythan Estuary, Sands of Forvie	1,125	1,010 in 2019	-115	**-10**	FM
Forth Islands	440	10 in 2019	-430	**-98**	UD
Farne Islands	862	417 in 2019	-445	**-52**	
Coquet Island	1,500	1,652 in 2019	+152	**+10**	
North Norfolk Coast	3,700	4,850 in 2018	+1,150	**+31**	
Alde-Ore Estuary	170	0 in 2018	-170	**-100**	
Foulness	320	0 in 2019	-320	**-100**	
Chichester & Langstone Harb	31	0 in 2019	-158	**-100**	
Solent & Southampton Water	231	90 in 2018	-141	**-61**	
Carlingford Lough	717	24 in 2019	-693	**-97**	
Larne Lough	189	1,010 in 2019	+821	**+434**	
Strangford Lough	593	252 in 2020	-341	**-58**	

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Morecambe Bay & Duddon	804	805 in 2019	+1	**+0**	
Anglesey terns	460	0 in 2019	-460	**-100**	

Although slightly more sites show decreases than show increases in Sandwich tern breeding numbers since site designation, the overall change in numbers since designation is small (Table 13), so it can be inferred that the UK Natura suite for breeding Sandwich tern shows favourable status overall. There is no clear geographic pattern to the changes, suggesting that these may be due to colony-specific features rather than to regional differences in environmental conditions. It is a recognised behaviour of Sandwich terns that colony sites can be abandoned and the entire colony can relocate between years.

Three sites were listed in the 3rd UKSPA review as designated for nonbreeding (passage) Sandwich terns (Stroud et al. 2016). These are Firth of Forth SPA (classified 30 October 2001, a winter peak mean count of 1,617 Sandwich terns 1993/94 to 1997/98) Teesmouth and Cleveland Coast SPA (classified 1995, 1,900 Sandwich terns in autumn 1988-1992) and The Dee Estuary SPA (classified 17 July 1985, revised 10 December 2009, 957 Sandwich terns 5-year mean 1995-1999).

Since the Stroud et al. (2016) review, Poole Harbour SPA has been designated, with Sandwich tern as a feature.

4.14.2 Demography

Adult (5 years and older) survival 0.898 (s.e. 0.029), age of first breeding 3 years, incidence of missed breeding uncertain, immature (2, 3, 4 year old) survival 0.741, juvenile survival 0.358, breeding success generally around 0.7 chicks per pair, natal and adult dispersal uncertain (Horswill and Robinson 2015). JNCC SMP data suggest Sandwich tern breeding success has shown no clear trend from 1986 to 2018, but has been rather variable, from almost zero up to 0.7 chicks per pair, but generally around 0.3 chicks per pair (JNCC 2020). In contrast, breeding success in England has tended to be higher, around 0.5 chicks per pair, but shows some decline from around 0.6 chicks per pair in 1986-2002 to 0.4 chicks per pair in 2004-2015, but higher breeding success, around 0.6, in 2016-2018 (JNCC 2020). Liechty et al. (2017) estimated apparent adult survival at 0.68 at a colony in Louisiana subject to high rates of land loss and habitat degradation (this is the North American population, now considered to be a separate but very closely related species). They suggest that this low survival estimate may reflect high emigration of adults due to the unstable habitat.

4.14.3 Ecology and threats to population

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. Sandwich tern flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.14.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified eight potential measures that were likely to improve the conservation status of Sandwich terns.

1. Fencing out foxes from colonies;
2. Stoat control/eradication;
3. Flood 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.

4.15 Common tern *Sterna hirundo*

4.15.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 classified common tern as Green, Amber, and Amber respectively. Birds Directive Annex 1 and migratory species.

The biogeographic population (Europe) was estimated at 280,000 pairs, of which 10,000 pairs breed in Great Britain and 4,200 pairs in all-Ireland (Mitchell et al. 2004). National surveys found an increase of 9% in breeding numbers in the UK between 1969 and 1986 and a decrease of 9% between 1986 and 2000 (JNCC 2020). JNCC SMP data show almost no change in breeding numbers, in the UK as a whole, or in Scotland alone, from 1986 to 2018 (JNCC 2020).

Stroud et al. (2016) identified that the SPA suite with breeding common tern as a designated feature has 19 qualifying sites in Great Britain, six in Scotland (Cromarty Firth SPA; Forth Islands SPA; Glas Eileanan SPA; Imperial Dock Lock, Leith SPA; Inner Moray Firth SPA; Ythan Estuary, Sands of Forvie and Meikle Loch SPA), 12 in England (Breydon Water SPA; Coquet Island SPA; Dungeness to Pett Level SPA; Farne Islands SPA; Foulness SPA; Mersey Narrows and North Wirral Foreshore SPA; North Norfolk Coast SPA; Poole Harbour SPA; Ribble and Alt Estuaries SPA; Solent and Southampton Water SPA; The Dee Estuary SPA; The Wash SPA) and one in Wales (Ynys Feurig, Cemlyn Bay and The Skerries SPA, also known as Anglesey terns SPA). The SPAs in Great Britain were estimated to hold 45.6% of the Great Britain breeding population of common terns present in 2000 (Stroud et al. 2016). Four sites in Northern Island also qualify (Carlingford Lough; Larne Lough; Lough Neagh and Lough Beg; Strangford Lough).

Table 14. Summary of common tern breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Cromarty Firth	294	0 in 2015	-294	**-100**	UD
Inner Moray Firth	310	0 in 2015	-310	**-100**	UNc
Ythan Estuary, Sands of Forvie	265	278 in 2018	+13	**+5**	UNc
Forth Islands	334	179 in 2019	-155	**-46**	UD
Imperial Dock Lock	558	246 in 2019	--312	**-56**	FM
Farne Islands	183	100 in 2020	-83	**-45**	
Coquet Island	740	1,300 in 2020	+560	**+76**	
The Wash	152	114 in 2019	-38	**-25**	
North Norfolk Coast	460	290 in 2020	-170	**-37**	
Dungeness to Pett Level	188	100 in 2020	-88	**-47**	
Foulness	220	25 in 2000	-195	**-89**	
Breydon Water	155	100 in 2020	-55	**-35**	
Poole Harbour	178	200 in 2015	+22	**+12**	
Solent & Southampton Water	267	94 in 2018	-173	**-65**	
Glas Eileanan	530	55 in 2015	-475	**-90**	UD
Carlingford Lough	443	56 in 2019	-387	**-87**	

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Larne Lough	247	300 in 2019	+53	**+21**	
Lough Neagh & Lough Beg	185	150 in 2019	-35	**-19**	
Strangford Lough	603	250 in 2020	-352	**-59**	
Ribble & Alt Estuaries	182	6 in 2018	-176	**-97**	
Mersey Narrows & N Wirral	177	190 in 2019	+13	**+7**	
Dee Estuary	392	0 in 2019	-392	**-100**	
Anglesey terns	189	191 in 2019	+2	**+1**	

Based on the most recent count in relation to numbers at designation, numbers of common terns breeding on SPAs have decreased strongly at 16 sites, decreased moderately at one site, and increased at six sites (Table 14). The overall net change is a decrease of just over 3,000 pairs across the UK Natura suite for breeding common tern. There is no clear geographical pattern as the minority of sites showing increases in numbers are distributed from Aberdeenshire to Wales to southern England.

One site was listed in the 3rd UKSPA review as qualifying for nonbreeding (passage) common terns (Stroud et al. 2016). This is Mersey Narrows and North Wirral Foreshore SPA (the citation, compiled July 2013, lists 1,475 individual common terns nonbreeding 2004-2008). Since then, one more site has been added for terns at sea; Outer Firth of Forth and St Andrews Bay Complex SPA, designated on 3 December 2020, includes common tern as a breeding season feature.

Since the Stroud et al. (2016) review, Teesmouth and Cleveland Coast SPA has been designated with common tern as a feature.

4.15.2 Demography

Adult (6 years and older) survival 0.883 (s.d. 0.014), age of first breeding 3 or 4 years, incidence of missed breeding relatively low, immature survival (0-2 years) 0.441, (3-4 years) 0.85, breeding success generally around 0.76 chicks per pair, natal dispersal 0.07, adult dispersal 0.1 (Horswill and Robinson 2015). JNCC SCM data show relatively poor breeding success in Scotland in most years from 1986 to 2018, with an average of about 0.5 chicks per pair, whereas in England breeding success was around 0.6 chicks per pair in most years, though varying from as low as 0.3 to as high as 1.02 (JNCC 2020). Palestis and Hines (2015) estimated common tern adult survival at Pettit Island, New Jersey, at 0.88. However, estimates of adult survival at other colonies in the area were lower, around 0.71. Palestis and Hines (2015) attributed these lower values to the fact that these colonies were in decline, with adults showing permanent emigration as a result of frequent flooding due to sea level rise and severe storms. Szostek and Becker (2012) studied demography of the Jade Bay (German Wadden Sea) common tern colony over an 18 year period that transitioned from colony stability to declining breeding numbers. They measured adult survival at 0.9 throughout the study period but found that breeding success and immature survival had declined. The decline in breeding success was attributed to decline in forage fish stocks, especially sprats.

4.15.3 Ecology and threats to population

Common tern is a medium-sized tern that breeds on shingle beaches, on rocky and grassy islands, and also inland on gravel shores of lakes and rivers. Colonies can be large, but this species can also nest in small groups or occasionally as individual pairs. Common terns feed by plunge diving to catch small fish close to the water surface. They are less marine than the Arctic tern and tend to breed in smaller colonies widely distributed around the UK. After breeding they migrate southwards out of UK waters to winter along the West African coast. Migrants from elsewhere may pass through the UK in moderate numbers during autumn and spring migration, especially in autumn when large flocks can gather on some UK estuaries. Common terns tend to feed on small pelagic fish such as sprats and sandeels but will take a wide variety of small fish and sometimes insects too.

Woodward et al. (2019) list the foraging range of breeding common terns as mean 6.4 km, mean maximum 18 km, maximum 30 km. However, these distances are likely to apply more along the coast than directly out to sea, given the preference of UK breeding common terns to remain near the coast.

Mammal predators, especially foxes and American mink, but also stoats and rats, strongly influence common tern breeding habitat choice and their breeding success. Breeding sites can be lost due to erosion, flooding or vegetation succession, but common terns are less susceptible to these than are Sandwich terns, and common terns can more often exploit human activity that provides artificial nesting habitat. Common terns will nest in dockland or coastal industrial areas on gravel, on islands in gravel pits or reservoirs, on flat roofs of industrial buildings and on purpose-built rafts. Breeding success is often higher on these artificial sites because predation risk and disturbance can be lower than at many natural sites (Mitchell et al. 2004). Common terns can be displaced by gulls, and predation of eggs and chicks by gulls can result in decline in numbers or colony abandonment. This impact can be made much worse when human disturbance increases the opportunities for predators to access tern nests (Davis 2019). Although their diet is varied so may be somewhat buffered against reductions in abundance of sandeels, the evidence from north Scotland suggests that the decline of sandeels there has had an adverse effect on common tern breeding numbers. That may be partly due to reduced food availability but partly due to impacts from predators such as gulls and skuas switching diet to take more terns. Common terns may also be influenced by fisheries off West Africa to the extent that these alter abundance of forage fish stocks in that region. Common tern flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.15.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified eight potential measures that were likely to improve the conservation status of common terns.

1. Provision of nest platforms;
2. Fencing out foxes from colonies;
3. Stoat control/eradication;
4. Exclusion of large gulls;
5. Mink eradication;

6. Rat eradication;
7. Closure of sandeel and sprat fisheries close to common tern colonies; and
8. Feral cat eradication.

The first six 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 a lack of evidence in this regard specifically for common tern. There was a lack of clear evidence that this species would benefit from feral cat eradication. Studies since 2013 do not alter any of these conclusions, but some work strengthens the evidence of success in the deployment of measures that could be used as compensation. The use of specially-built tern nest boxes on the Isle of May has been successful in allowing common terns (and other tern species) to increase in numbers and to breed successfully despite the presence of potential nest predators such as large gulls (Steel and Outram 2020). In contrast, presence of American mink and increases in human disturbance together with predation pressures from large gulls and crows resulted in a large decline in common tern breeding numbers at Imperial Dock Lock SPA (Davies 2019), demonstrating the potential benefits that can be gained from measures that reduce access of invasive mammal predators and the potential benefits from reducing human disturbance at tern colonies where this may increase impact of predators on common terns.

4.16 Arctic tern *Sterna paradisaea*

4.16.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 all classified Arctic tern as Amber. Birds Directive Annex 1 and migratory species.

The biogeographic population (Europe) was estimated at 629,000 pairs, of which 53,000 pairs breed in Great Britain and 3,500 pairs in all-Ireland (Mitchell et al. 2004). National surveys found a 50% increase in breeding numbers in the UK from 1969 to 1986, but a 31% decline from 1986 to 2000 (JNCC 2020). JNCC SCM data (JNCC 2020) show very little long-term trend in breeding numbers in England (2018 index about the same as in 1986) but show a large decline in Scotland (2018 index close to 50% that of 1986).

Stroud et al. (2016) identified that the SPA suite with breeding Arctic tern as a designated feature has 15 qualifying sites in Great Britain, 12 in Scotland (Auskerry SPA; Fair Isle SPA; Fetlar SPA; Forth Islands SPA; Foula SPA; Mousa SPA; Papa Stour SPA; Papa Westray SPA; Pentland Firth Islands SPA; Rousay SPA; Sumburgh Head SPA; West Westray SPA), two in England (Coquet Island SPA; Farne Islands SPA) and one in Wales (Ynys Feurig, Cemlyn Bay and The Skerries SPA, also known as Anglesey terns SPA). The SPAs in Great Britain were estimated to hold 20.6% of the Great Britain breeding population of Arctic terns present in 2000 (Stroud et al. 2016). Three sites in Northern Island also qualify (Copeland Islands; Outer Ards; Strangford Lough).

Table 15. Summary of Arctic tern breeding SPA feature conservation status

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Fetlar	1,065	16 in 2018	-1,047	**-98**	UD
Foula	1,500	150 in 2020	-1,300	**-87**	UD
Papa Stour	850	0 in 2018	-850	**-100**	UNc
Mousa	1,000	109 in 2019	-891	**-89**	UD
Sumburgh Head	700	42 in 2019	-658	**-94**	UNc
Fair Isle	1,100	286 in 2019	-814	**-74**	UD
Papa Westray	1,700	176 in 2000	-1,524	**-90**	UNc
West Westray	1,140	No count	n/a	**n/a**	UNc
Rousay	790	9 in 2018	-781	**-99**	UNc
Auskerry	780	8 in 2016	-772	**-99**	UD
Pentland Firth Islands	1,000	No count	n/a	**n/a**	UNc
Forth Islands	540	490 in 2019	-50	**-9**	FD
Farne Islands	2,003	1,900 in 2019	-103	**-5**	
Coquet Island	700	1,300 in 2019	+600	**+86**	
Copeland Islands	566	150 in 2019	-416	**-73**	
Outer Ards	207	517 in 2017	+310	**+150**	
Strangford Lough	210	210 in 2020	0	**0**	

SPA	Pairs in citation	Most recent published count	Change from designation	Percent change	SCM
Anglesey terns	1,290	382 in 2019	-908	-70	

Based on the most recent count in relation to numbers at designation, numbers of Arctic terns breeding on SPAs have decreased strongly at 11 sites, decreased moderately at two sites, remained the same at one site and increased at two sites (Table 15). For another two sites there are no published recent counts, but both sites are classified by NatureScot as in Unfavourable conservation status. The overall net change is a decrease of over 9,000 pairs across the UK Natura suite for breeding Arctic tern from a total population at designation of around 17,000 pairs, so that about half of the birds have been lost. There is a clear geographical pattern to the trend, with very high losses from the northernmost colonies.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding Arctic terns (Stroud et al. 2016). Since then, Outer Firth of Forth and St Andrews Bay Complex SPA, designated on 3 December 2020, includes Arctic tern as a breeding season feature.

Since the Stroud et al. (2016) review, Northumbria Coast SPA has been designated, with Arctic tern as a feature.

4.16.2 Demography

Adult (2 years and older) survival 0.837 (s.d. 0.035), age of first breeding 4 years, incidence of missed breeding uncertain, immature survival uncertain, breeding success generally around 0.4 chicks per pair, natal dispersal 0.61, adult dispersal 0.09 (Horswill and Robinson 2015). JNCC SCM data show breeding success in Scotland being particularly low, below 0.3 chicks per pair in most years from 1986 to 2018, whereas breeding success in England has fluctuated but mainly between 0.4 and 0.8 chicks per pair (JNCC 2020). Mallory et al. (2018) measured adult survival of an apparently declining population of Arctic terns breeding in the Canadian High Arctic, at 0.883. Petersen et al. (2020) estimated survival of Arctic terns in western Iceland through the first year at 0.119 (s.e. 0.024), whereas survival of birds after one year of age was 0.952 (s.e. 0.030) overall, but survival declined since 2000 coincident with a collapse in the local sandeel stock, to 0.851 (s.e. 0.034). They concluded that reduced adult survival due to sandeel scarcity was the cause of declining breeding numbers in western Iceland after 2000.

4.16.3 Ecology and threats to population

Arctic tern is slightly smaller than the common tern and is more strictly marine. It feeds by plunge-diving to catch small fish, often foraging in areas of strong tidal flow such as around headlands. It can feed also on insects, but at UK colonies that may largely be a response to shortage of forage fish. In the UK, Arctic tern is near the southern limit of its breeding range. It often breeds in large colonies, but colonies can be abandoned if forage fish are scarce or if breeding success is impacted by predation. After breeding, Arctic terns migrate to overwinter in the southern hemisphere, many birds travelling as far as South Africa and even Antarctica.

Woodward et al. (2019) list the foraging range of breeding Arctic terns as mean 6.1 km, mean maximum 25.7 km, maximum 46 km.

Decline in sandeel abundance at Shetland in the 1980s resulted in breeding failure of Arctic terns and rapid decline in breeding numbers (though some adults remained alive but did not breed). Breeding failures and colony abandonment can be caused by mammal predators, and this has been the case in west Scotland due to American mink. However, Arctic terns will redistribute to breed in predator-free sites so declines in numbers may be less than reduction in distribution. Foxes and rats can also cause breeding failure, as can predation and competition from gulls (Mitchell et al. 2004). Arctic tern flight heights suggest moderate risk of collision with offshore wind farm turbines.

4.16.4 Potential compensation measures and suitability for this species

Arctic tern was not included in the review commissioned by Defra (Furness et al. 2013). The measures likely to be successful for common tern may apply as likely to be successful for Arctic tern. However, Arctic terns do not show the same propensity to nest on rafts or other artificial structures that are used by common terns, so that construction of artificial nest platforms may be less likely to work for Arctic tern. Nevertheless, Steel and Outram (2020) showed that Arctic terns used the ‘tern terraces’ composed of purpose-built tern nest boxes provided on the Isle of May, and as a result increased in breeding numbers from 77 pairs using nest boxes in 2016 to 152 pairs in nest boxes in 2019 (then representing 31% of the entire Isle of May Arctic tern colony). Birds using these nest boxes achieved good breeding success.

Although highly variable for a number of reasons including effects of predators and weather (especially heavy rainfall during the incubation period causing local flooding in colonies), Arctic tern breeding success at Foula, Shetland, shows a high correlation with the abundance of sandeels at Shetland in the period 1976-2004 (Figure 10). The relationship does not show any tendency to reach an asymptote, implying that any increase in sandeel abundance resulting from management to reduce depletion of sandeels caused by sandeel-fishing would be expected to lead to higher breeding success of Arctic terns, though with high stochastic variation due to other factors that also influence success.

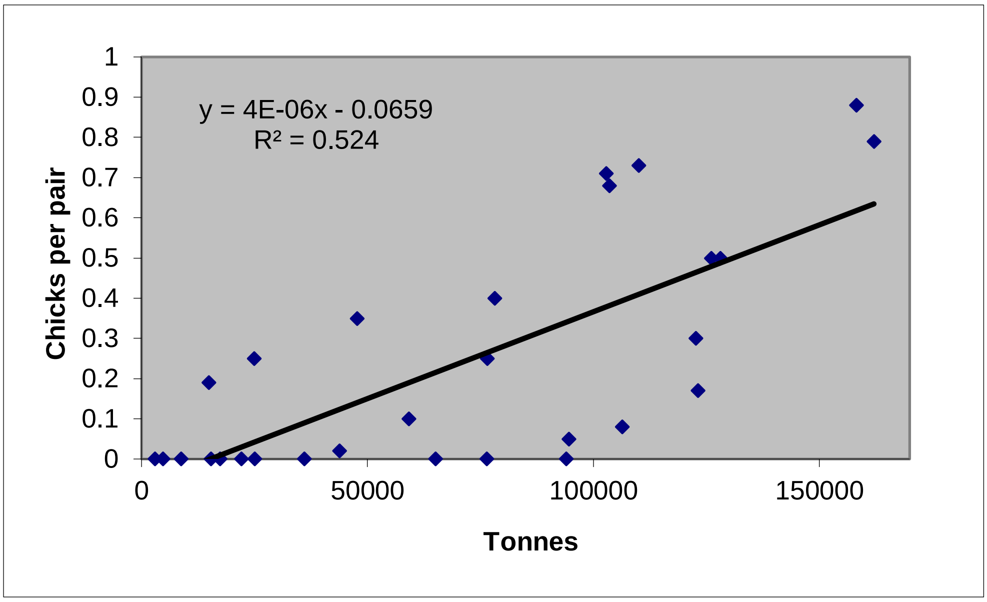

Figure 10. Breeding success of Arctic tern at Foula, Shetland, in relation to the Shetland sandeel total stock biomass for the years 1976 to 2004.

Arctic terns are highly dependent on sandeels when breeding in Shetland (Furness and Tasker 2000). Robertson et al. (2016) noted that Arctic terns fed chicks at Coquet Island, Northumberland, almost exclusively on sandeels, whereas common terns at the same colony fed chicks on sandeels and sprats so appeared to have a wider range of prey. A strong influence of sandeel abundance on Arctic tern breeding success and adult survival has also been found in west Iceland, where sandeel abundance fell as stocks of important predators haddock, herring and especially mackerel increased considerably in the region (Lilliendahl et al. 2013, Vigfusdottir et al. 2013, Petersen et al. 2020). Petersen et al. (2020) concluded that reduced adult survival due to sandeel scarcity was the cause of declining breeding numbers after 2000 in western Iceland.

4.17 Common guillemot *Uria aalge*

4.17.1 Conservation status and populations

IUCN Red List classification "Least Concern". BOCC 2, 3, and 4 all classified common guillemot as Amber. Birds Directive 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). Mitchell et al. (2004) state that numbers of pairs can be derived by multiplying colony census counts of individual adults by 0.67. National surveys found a 77% increase in 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 as a whole 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).

Stroud et al. (2016) identified that 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).

Table 16. Summary of common guillemot breeding SPA feature conservation status

SPA	Individuals in citation	Most recent published count (individuals)	Change from designation	Percent change	SCM
Hermaness, Saxa Vord & Valla	25,000	6,109 in 2016	-18,891	-76	UD
Noss	38,970	24,456 in 2015	-14,514	-37	UNc
Foula	37,500	24,799 in 2007	-12,701	-34	UD
Sumburgh Head	16,000	7,749 in 2018	-8,251	-52	UD
Fair Isle	32,300	20,924 in 2015	-11,376	-35	UD
West Westray	42,150	28,697 in 2017	-13,453	-32	UD
Calf of Eday	12,645	5,524 in 2018	-7,121	-56	UD

SPA	Individuals in citation	Most recent published count (individuals)	Change from designation	Percent change	SCM
Rousay	10,600	6,500 in 2016	-4,100	-39	UD
Hoy	13,400	12,198 in 2017	-1,202	-9	UNc
Marwick Head	37,700	11,985 in 2018	-25,715	-68	UD
Copinsay	29,450	18,454 in 2015	-10,996	-37	UNc
North Caithness Cliffs	38,300	25,000 in 2016	-13,300	-35	FM
East Caithness Cliffs	106,700	149,228 in 2015	-42,528	-40	FM
Troup, Pennan and Lion's Hd	44,600	23,800 in 2017	-20,800	-47	UD
Fowlsheugh	56,450	69,828 in 2018	+13,378	+24	FM
Buchan Ness to Collieston	8,640	29,187 in 2019	+20,547	+238	FM
Forth Islands	16,000	25,956 in 2018	+9,956	+62	FM
St Abb's Head to Fast Castle	31,750	43,000 in 2018	+11,250	+35	FM
Farne Islands	65,751	64,042 in 2019	-1,709	-3	
Flamborough & Filey Coast	62,100	84,647 in 2017	+22,547	+36	
Cape Wrath	13,700	38,109 in 2017	+24,409	+178	FM
North Rona & Sula Sgeir	43,200	4,961 in 2012	-38,239	-89	UD
Sule Skerry & Sule Stack	6,298	10,068 in 2018	+3,770	+60	FM
Handa	98,686	54,664 in 2016	-44,022	-45	UNc
St Kilda	22,700	10,300 in 2016	-12,400	-55	UD
Mingulay & Berneray	30,900	22,265 in 2014	-8,635	-28	FR
Flannan Isles	21,930	9,807 in 2000	-12,123	-55	UD
Canna and Sanday	5,800	2,850 in 2018	-2,950	-51	UD
Rum	4,000	2,454 in 2000	-1,546	-39	UNc
Shiant Isles	18,380	9,054 in 2015	-9,326	-51	UNc
North Colonsay & Western Cl	6,656	18,724 in 2018	+12,068	+181	FM
Ailsa Craig	3,350	6,180 in 2019	+2,830	+84	FM
Rathlin Island	41,887	130,445 in 2011	+88,558	+211	
Skomer, Skokholm and seas	15,262	17,895 in 2017	+2,633	+17	

Based on the most recent count in relation to numbers at designation, numbers of common guillemots breeding on SPAs have decreased strongly at 21 sites, decreased moderately at two sites, and increased at 11 sites (Table 16). There is a clear geographical pattern to the trend, with high losses from Orkney and Shetland and parts of west Scotland, mixed fortunes at colonies in north Scotland, but increases in south-west Scotland, Northern Ireland, Wales, south-east Scotland and east England. The colonies that have declined have lost 335,898 individuals. The colonies that have increased have gained 211,946 individuals. The overall net change is therefore a decrease of 123,952 individuals across the UK Natura suite for breeding common guillemot from a total population at designation of 1,058,755 individuals. That represents an overall decrease of 12%, which is a stronger decrease than suggested by the JNCC SCM data. This could possibly reflect the fact that decreases may have been greater in larger colonies or that more of the larger colonies designated as SPAs are in the northern areas of greatest population decrease. However, the poor condition of the northern part of the Natura suite for this species is clearly evident from Table 16.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding guillemots (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.

4.17.2 Demography

Adult (4 years and older) survival 0.939 (s.d. 0.015), age of first breeding 6 years, incidence of missed breeding about 0.08, juvenile survival 0.56, immature survival (1-2 years) 0.792, immature survival (2-3 years) 0.917, breeding success generally around 0.67 chicks per pair, natal dispersal 0.58, adult dispersal very low at 0.007 (Horswill and Robinson 2015). JNCC SCM data show breeding success in Scotland to have been around 0.7 to 0.8 chicks per pair in 1986-2002, but much lower in 2003-2008 (reaching a low of 0.2 chicks per pair in 2007), then recovering towards 0.7 chicks per pair by 2018. This picture hides high regional variation within Scotland. Breeding success in Shetland was only 0.14 chicks per pair in 2011-2013 compared to 0.72 in south-east Scotland (JNCC 2020). Breeding success in England has generally remained higher than in Scotland (JNCC 2020). JNCC SMP has monitored survival rates of guillemots at the Isle of May and Skomer. Adult survival estimates were broadly consistent with the data presented by Horswill and Robinson (2015) but suggest decreases in adult survival in some years. At Isle of May adult survival fell to about 0.8 in the mid-2000s, but subsequently recovered. At Skomer, adult survival was low in years affected by ‘wrecks’ of birds in winter relating to climate, and by oil spills (JNCC 2020). Sarzo et al. (2019) used a Bayesian framework to estimate survival of immature common guillemots from a mark-recapture study at Stora Karlso, obtaining values of 0.53 for first year, 0.87 for 2nd year, 0.96 for 3rd year birds in that Baltic Sea population.

4.17.3 Ecology and threats to population

Common guillemot is one of the most abundant seabirds in the cooler seas of the northern hemisphere. It uses its wings to propel itself underwater in pursuit of small fish, and can dive to at least 100 m. In the North Sea, in summer, it especially feeds on sandeels and sprats, while in winter it takes a more varied but mostly fish diet. Even in winter, when sandeels are buried in the seabed, it can dig or scare these fish out of the sediment so still takes some sandeel throughout the year. Common guillemots breed at very high density on broad ledges of cliffs, on rocky platforms, or under large boulders at the foot of cliffs, though they are mainly found on ledges. After breeding, they disperse, males taking the chick to sea to find food. Common guillemots from UK colonies mostly remain in UK waters, and generally not far from their breeding site, but a few migrate to the Barents Sea to moult, returning to winter in UK waters. Common guillemots become flightless during autumn moult. After moult, birds may start to revisit breeding ledges as early as December, although at some colonies few adults return until April. Timing of pre-breeding return may relate to competition for sites and available food.

Woodward et al. (2019) list the foraging range of breeding guillemot as mean 33.1 km, mean maximum 73.2 km, maximum 338 km.

When monofilament set nets were used in UK waters, common guillemots were adversely affected by drowning in nets, especially in nets set close to colonies. That pressure has been reduced to some extent (Mitchell et al. 2004, Northridge et al. 2020), but current levels of bycatch remain a concern in UK waters and removal of that bycatch has been estimated to potentially lead to an increase of about 1% in common guillemot numbers over a 25 year period (Miles et al. 2020). Wrecks of common guillemots 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. 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 guillemots and other UK seabirds so probably do not influence population trends. Common guillemots are very vulnerable to oil pollution, but 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 common guillemots (Mitchell et al. 2004). 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 guillemot chicks and some chicks from nest sites. Predation by rats has been recorded, especially where guillemots nest under boulders. Common guillemot 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 common guillemots, at least at some offshore wind farms though apparently not all, so possible impacts of displacement are of concern but may vary considerably among sites.

4.17.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified four potential measures that were likely to improve the conservation status of common guillemots.

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.

New evidence relating to gains from closure of sandeel or sprat fisheries:

There is evidence of guillemot return rates to the Isle of May being higher when sandeel stock biomass in ICES Area 4 is higher, but that relationship is not yet fully analysed or published. 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.

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 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 prep.), so this behaviour is not unique, despite the particularly high energy cost of flight in this species.

New evidence in relation to 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 ground nesting habitat and whether or not guillemot numbers can increase into such habitat or are constrained by other factors such as food availability.

4.18 Razorbill *Alca torda*

4.18.1 Conservation status and populations

IUCN Red List classification "Near Threatened". BOCC 2, 3, and 4 all classified razorbill as Amber. Birds Directive 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). Mitchell et al. (2004) state that numbers of pairs can be derived by multiplying colony census counts of individual adults by 0.67. 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 (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 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).

Table 17. Summary of razorbill breeding SPA feature conservation status

SPA	Individuals in citation	Most recent published count (individuals)	Change from designation	Percent change	SCM
Foula	6,200	559 in 2007	-5,641	**-91**	UD
Fair Isle	3,400	1,930 in 2015	-1,470	**-43**	UD
West Westray	1,946	2,159 in 2017	+213	**+11**	FM
North Caithness Cliffs	4,000	2,800 in 2016	-1,200	**-30**	FR
East Caithness Cliffs	15,800	30,042 in 2015	+14,242	**+90**	FM
Troup, Pennan and Lion's Head	4,800	4,518 in 2017	-282	**-6**	UD
Fowlsheugh	5,800	14,063 in 2018	+8,263	**+142**	FM
Forth Islands	1,400	5,636 in 2018	+4,236	**+303**	FM
St Abb's Head to Fast Castle	2,180	2,761 in 2018	+581	**+27**	FM
Flamborough & Filey Coast	15,776	30,228 in 2017	+14,452	**+92**	
North Rona & Sula Sgeir	2,300	513 in 2012	-1,787	**-78**	UD

SPA	Individuals in citation	Most recent published count (individuals)	Change from designation	Percent change	SCM
Cape Wrath	1,800	3,241 in 2017	+1,441	**+80**	FM
Handa	16,394	8,207 in 2019	-8,187	**-50**	UD
Flannan Isles	3,160	No data	n/a	**n/a**	UR
Mingulay & Berneray	16,890	17,400 in 2014	+510	**+3**	FR
St Kilda	3,810	820 in 2016	-2,990	**-78**	UD
Shiant Isles	10,950	8,029 in 2015	-2,921	**-27**	FR
Rathlin Island	8,922	22,975 in 2011	+14,053	**+158**	
Skomer, Skokholm and seas	4,300	7,663 in 2018	+3,363	**+78**	

Based on the most recent count in relation to numbers at designation, numbers of razorbills breeding on SPAs have decreased strongly at seven sites, decreased moderately at one site, increased at ten sites, while for one site there is no recent published count (Table 17). There appears to be a geographical pattern to the trend, with high losses from Shetland and parts of west Scotland but increases at most colonies along the North Sea coast from Orkney to Flamborough & Filey, and increases in Northern Ireland and Wales. The colonies that have declined have lost 24,478 individuals. The colonies that have increased have gained 61,354 individuals. The overall net change is therefore an increase of 36,876 individuals across the UK Natura suite for breeding razorbill from a total population at designation of 129,828 individuals. That represents an overall increase of 28%.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding 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.

4.18.2 Demography

Adult (3 years and older) survival 0.895 (s.d. 0.067), age of first breeding 5 years, incidence of missed breeding relatively low at about 0.03, immature survival (0-2 years) 0.63, breeding success generally around 0.57 chicks per pair, natal dispersal 0.088, adult dispersal 0.047 (Horswill and Robinson 2015). JNCC SCM data show that breeding success in monitored sites in Scotland was consistently around 0.5 to 0.7 chicks per pair in 1986-2002, but then fell to about 0.4 chicks per pair in 2004-2008, before recovering towards 0.6 chicks per pair in 2017 (JNCC 2020). However, colonies in Shetland showed much lower breeding success than colonies in east Scotland. Return rates of breeding adults have been monitored on the Isle of May and Skomer (JNCC 2020). Years with low annual survival were often associated with 'wrecks' of birds during the non-breeding period.

4.18.3 Ecology and threats to population

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, 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. Razorbills are at risk of fishery bycatch, and although this risk has decreased in UK waters (Mitchell et al. 2004, Northridge et al. 2020), there is a small but detectable impact on their population from fishery bycatch (Miles et al. 2020). 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, so possible impacts of displacement are of concern.

4.18.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified four potential measures that were likely to improve the conservation status of razorbills.

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.

New evidence relating to gains from closure of sandeel or sprat fisheries:

There is evidence of razorbill return rates to the Isle of May being higher when sandeel stock biomass in ICES Area 4 is higher, but that relationship is not yet fully analysed or published. In a 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.

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. Razorbills' trophic position increased in the poor survival winter and the population foraged in more distant southerly waters of the North Sea.

Cherenkov et al. (2016) found that razorbills breeding at colonies on the Kola Peninsula, in the White Sea and on the coast of the Gulf of Finland, in Russia, have been increasing in numbers during the last 20 years. Many of these birds overwinter in UK North Sea waters, so are likely to be mixing with birds from UK colonies during the nonbreeding season.

New evidence in relation to 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).

4.19 Puffin *Fratercula arctica*

4.19.1 Conservation status and populations

IUCN Red List classification "Vulnerable". BOCC 2, 3, and 4 classified Atlantic puffin as Amber, Amber, and Red respectively. Birds Directive migratory species.

The biogeographic population (subspecies *arctica* in NE Atlantic) was estimated at 5,676,000 pairs, of which 580,000 pairs breed in Great Britain and 21,000 pairs in all-Ireland (Mitchell et al. 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 large declines in breeding numbers of puffins at some colonies, especially in Shetland, but the overall national trend since 2000 is uncertain (JNCC 2020).

The SPA suite with breeding puffin as a designated feature has 19 qualifying sites in Great Britain, 15 in Scotland (Canna and Sanday SPA; Cape Wrath 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; Shiant Isles SPA; St Kilda SPA; Sule Skerry and Sule Stack SPA), three in England (Coquet Island SPA; 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 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). Stroud et al. (2016) include East Caithness Cliffs SPA in their list of sites having puffin as a breeding feature, but that is incorrect.

Table 18. Summary of puffin breeding SPA feature conservation status

SPA	Pairs, or individuals, in citation	Most recent published count	Change from designation	Percent change	SCM
Hermaness, Saxa Vord & Valla	55,000	1,757 in 2017	-53,243	**-97**	UD
Noss	2,348	1,174 in 2017	-1,174	**-50**	UD
Foula	48,000	6,351 in 2016	-41,649	**-87**	UNc
Fair Isle	23,000	6,666 in 2015	-16,334	**-71**	UD
Hoy	3,500	361 in 2017	-3,139	**-90**	UD
North Caithness Cliffs	2,080	3,036 in 2016	+956	**+46**	FM
Forth Islands	14,000	43,585 in 2018	+29,585	**+211**	FD
Farne Islands	38,399	43,752 in 2019	+5,353	**+14**	
Coquet Island	15,843	25,029 in 2019	+9,186	**+58**	
Flamborough & Filey Coast	958	4,279 in 2018	+3,321	**+347**	
North Rona & Sula Sgeir	5,300	No data	n/a	**n/a**	UNc

SPA	Pairs, or individuals, in citation	Most recent published count	Change from designation	Percent change	SCM
Sule Skerry & Sule Stack	46,900	47,742 in 2018	+842	**+2**	FD
Cape Wrath	5,900	1,520 in 2017	-4,380	**-74**	UNc
Flannan Isles	4,400	15,761 in 2001	+11,361	**+258**	FM
St Kilda	155,000	142,264 in 2000	-12,736	**-8**	FM
Mingulay & Berneray	4,000	3,816 in 2009	-184	**-5**	FM
Canna & Sanday	1,200	1,935 in 2019	+735	**+61**	FM
Shiant Isles	77,000	64,695 in 2015	-12,305	**-16**	FM
Rathlin Island	2,398	695 in 2011	-1,703	**-71**	
Skomer, Skokholm & seas off Pembrokeshire	9,500	38,342 in 2018	+28,842	**+304**	

Based on the most recent count in relation to numbers at designation, numbers of puffins breeding on SPAs have decreased strongly at seven sites, decreased moderately at three sites, increased at nine sites, and there is no recent published count for one site (Table 18). There is a very clear geographical pattern to the trend, with very high losses from Orkney and Shetland, some losses from parts of west Scotland, but large increases at colonies on the North Sea coast from Caithness to Flamborough & Filey. The colonies that have declined have lost 146,847 pairs/individuals. The colonies that have increased have gained 90,181 pairs/individuals. The overall net change is therefore a decrease of 56,666 pairs/individuals across the UK Natura suite for breeding puffin from a total population at designation of 514,726 pairs/individuals. That represents an overall decrease of 11%.

No sites were listed in the 3rd UKSPA review as designated for nonbreeding puffins (Stroud et al. 2016). Since then, three new marine sites, Seas off St Kilda SPA, Seas off Foula SPA, and Outer Firth of Forth and St Andrews Bay Complex SPA, all designated on 3 December 2020, include puffin as a breeding season feature.

4.19.2 Demography

Adult (6 years and older) survival 0.906 (s.d. 0.083), age of first breeding 5 years, incidence of missed breeding about 0.08, immature survival (0-3 years) 0.709, (3-4 years) 0.76, (4-5 years) 0.805, breeding success generally around 0.62 chicks per pair, natal dispersal 0.23, adult dispersal 0.042 (Horswill and Robinson 2015). JNCC SCM provides some evidence on breeding success, though mainly from a few accessible colonies such as Isle of May and Farne Islands. Breeding success in Scotland was around 0.7 chicks per pair in 1986-1996 but fell to around 0.4-0.5 chicks per pair in 2000-2012, then recovered to around 0.6 chicks per pair in 2014-2018 (JNCC 2020). Breeding success in England in 1992-2017 has generally been higher than at Scottish colonies, averaging about 0.7-0.8 chicks per pair in most years but with two years of much lower breeding success due to flooding by exceptionally heavy rain (JNCC 2020). Survival rates of breeding adults are monitored on the Isle of May and Skomer.

4.19.3 Ecology and threats to population

The puffin (also known as the Atlantic puffin) is a burrow-nesting auk. It 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 themself 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) list the foraging range of breeding puffins as mean 62 km, mean maximum 137 km, maximum 383 km.

Mitchell et al. (2004) identify the greatest threat to puffins as changes in breeding season food supply, especially declines in sandeel abundance caused by fisheries or by climate change. However, they also highlight impacts of rats, mink, foxes and other predatory mammals, including feral cats and ferrets. Breeding can also be affected by heavy rainfall causing flooding of burrows, and by soil erosion, which can be caused by large numbers of puffins burrowing through the fragile soil of sea cliffs. Numbers can be reduced locally by predation by great skuas and gulls, especially great black-backed gulls which kill adult puffins at many colonies. Oil pollution, chemical pollution and wrecks can affect puffins, but are thought not to be major influences on their populations (Mitchell et al. 2004). Puffin 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 puffins, so possible impacts of displacement are of concern, although this may be less of a problem than for common guillemot or razorbill given the more pelagic and widespread low density distribution of puffins at sea.

4.19.4 Potential compensation measures and suitability for this species

Furness et al. (2013) identified three potential measures that were likely to improve the conservation status of puffins.

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. It was recognised that strong efforts are already made to prevent oil spills, so that this was unlikely to be a practical option, and that the very extensive distribution of puffins across the North Atlantic during the nonbreeding season made this less appropriate 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 puffin. 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.

New evidence relating to gains from closure of sandeel or sprat fisheries:

There is evidence of puffin return rates to the Isle of May being higher when sandeel stock biomass in ICES Area 4 is higher, but that relationship is not yet fully analysed or published. Puffins, like most other seabirds breeding in Shetland, used to feed chicks primarily on sandeels (Furness and Tasker 2000). When sandeel stock collapsed at Shetland, puffins were strongly affected. Breeding numbers of puffins have declined dramatically at sites where they are SPA features in Shetland (Table 17). Owen et al. (2018) emphasise this decline at Hermaness, but it is a feature of all the (formerly) large puffin colonies in Shetland. Miles et al. (2015) attribute the decline in breeding numbers of puffins at Fair Isle to reduced breeding success caused by collapse of the Shetland sandeel stock, so that recruitment into the population was reduced. Puffin breeding success at Shetland colonies has been reduced much more than that of many other seabird species (JNCC 2020), indicating that, like Arctic tern, kittiwake, and shag, the puffin is a sandeel specialist while breeding in Shetland and finds it difficult to switch to other food types or to increase foraging effort enough to sustain breeding when sandeel abundance is low (annual Shetland Bird Reports, Fair Isle Bird Observatory Reports). Experimental evidence supports this: puffin chicks at a colony with low forage fish abundance grew better when given supplementary food (Fitzsimmons et al. 2017). This provides strong evidence that puffin conservation would benefit from measures that increase forage fish stocks (supplementary feeding would not be practical as a compensation measure because puffin numbers are large, and because most puffin burrows at most puffin colonies are inaccessible, or where they could be accessible the disturbance caused by attempting to put food into burrows would be highly damaging). The modelling of Lindegren et al. (2018) indicates that the most effective measure available to achieve improvement to forage fish stocks is to limit fishing effort on forage fish stocks, although recovery may 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 significantly differed between good and bad years (in terms 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. However, lower survival of puffins in the low-abundance years implies that maintenance of moderate forage fish stock biomass is necessary to ensure good survival of puffins. This indicates that measures to reduce depletion of forage fish stocks in UK waters will be beneficial to puffin survival, as well as beneficial to other seabirds. Fayet et al. (2017) tracked overwinter movements of 270 puffins from numerous colonies to compare movement patterns. They found that puffins from larger colonies or with poorer local winter conditions migrated further and visited less-productive waters; this in turn led to differences in flight activity and energy expenditure. They showed that competition and local winter resource availability are important drivers of migratory movements, and most likely are major drivers of adult survival. This further emphasises the value to puffin conservation of healthy stocks of forage fish in the home waters of the populations.

It is difficult to assess diet of puffins in the nonbreeding period, but Harris et al. (2015) reported the stomach contents of 176 puffins shot legally around the Faroe Islands in October-January. Small sandeels were the most frequent winter prey item, present in 82% of stomachs, with large sandeels present in 32%. This further supports the evidence that auks can feed on sandeels even in winter, when sandeels spend much of the time buried in the sand.

New evidence in relation to eradication of rats and other invasive mammal predators:

Eradication of rats from Lundy resulted in puffin breeding numbers increasing from 5 individuals in 2004, the year rat eradication was completed, to 375 individuals in 2017; the increase is attributed to the removal of the pressure of predation by rats (Booker et al. 2019). At Canna, where rats were eradicated in 2005-2006, puffins, formerly confined to offshore stacks, have recolonised sites on the mainland of Canna and a count of over 2,000 was recorded in 2016, an increase of about 500% from the numbers when rats were present (Luxmoore et al. 2019). Eradication of rats on Ailsa Craig resulted in puffins recolonising that island (Zonfrillo 2002, 2007). There is, therefore, clear evidence that eradication of rats can be highly beneficial for puffin populations.

5 STRATEGIC COMPENSATION APPROACHES

5.1 No-take zones for forage fish

The seabird species for which there is strongest evidence that creation of forage fish no-take zones would be highly effective is the kittiwake. The seabird species for which compensation measures relating of offshore wind impacts are most likely to be necessary is kittiwake. However, no-take zones for forage fish would be likely to benefit several different seabird species, both in terms of breeding success and foraging effort impacts on adult body condition, but also in terms of overwinter survival (summarised in Table 19). It is important to stress that this measure would need to be over very large areas of sea to be effective. A no-take zone would be most effective if applied over the whole area occupied by a particular sandeel stock (i.e. hundreds of km^2). For example, the no-take zone for sandeels off east Scotland represents only about one-quarter of the area delineated by ICES as holding the sandeel stock “North Sea area 4”. The sandeel fishery in ICES area 4 has a quota set on the basis of the estimated abundance of sandeels within the closed area as well as the open area, and therefore fishing mortality that can be applied in the open area can be even higher than would be the case if applied across the whole stock. The no-take zone does not, therefore, necessarily reduce the amount of sandeel taken from the stock, but simply affects the spatial distribution of the harvest.

Table 19. Traffic light indications for significant linkages between seabird ecology and consequences of management systems that result in recovery of small forage fish stock biomass. Green indicates strong evidence of a strong effect. Yellow indicates weak evidence or a weak effect. No colour indicates that a relationship is unlikely for this combination.

Species	Small forage fish (sandeels, sprats, juvenile herring) form a major part of the breeding season diet	Breeding success is strongly affected by forage fish stock biomass	Evidence from UK waters that breeding success is constrained by shortage of forage fish	Small forage fish (sandeels, sprats, juvenile herring) form a major part of the winter diet in UK waters	Evidence from UK waters that overwinter survival is constrained by shortage of forage fish
Kittiwake	🟩	🟩	🟩		
Arctic skua	🟩	🟩	🟩		
Sandwich tern	🟩	🟩	🟩		
Arctic tern	🟩	🟩	🟩		
Common tern	🟩	🟩	🟩		
Puffin	🟩	🟩	🟩	🟩	🟨

Species	Small forage fish (sandeels, sprats, juvenile herring) form a major part of the breeding season diet	Breeding success is strongly affected by forage fish stock biomass	Evidence from UK waters that breeding success is constrained by shortage of forage fish	Small forage fish (sandeels, sprats, juvenile herring) form a major part of the winter diet in UK waters	Evidence from UK waters that overwinter survival is constrained by shortage of forage fish
Shag	🟩	🟩	🟩	🟩	🟨
Red-throated diver	🟩	🟨	🟨	🟩	
Common guillemot	🟩			🟩	🟨
Razorbill	🟩			🟩	🟨
Great skua	🟨	🟨	🟨		
Fulmar	🟨	🟨			
Lesser black-backed gull	🟨				
Manx shearwater	🟨				
Great northern diver				🟨	
Great black-backed gull	🟨				
Herring gull					
European storm-petrel					
Leach's petrel					
Gannet					

The adverse effects of forage fish stock depletion by fisheries on multiple UK seabird species and colonies could most effectively be reversed by declaring all UK waters a no-take zone for sandeel, sprat and juvenile herring. Evidence from ICES area 4 is that creation of a no-take zone can result in an increase in fishing pressure along the edge of that zone (Peter Wright, pers. comm.), and increases fishing pressure on unprotected parts of the management area. There is some limited evidence of redistribution of sandeels from marginal into optimal habitat as fish are removed from optimal habitat. This appears to have been the case at Shetland, where high catches were maintained at Mousa while other sandbanks became depleted around the rest of Shetland, and there is similar evidence from groups of sandbanks in Norwegian waters (ICES 2017). This suggests that a no-take zone should include not only the core foraging grounds used by breeding kittiwakes (and other seabirds that rely on sandeels, sprats or juvenile herring) but should include areas adjacent to those too. **To be most effective, the entire stock should be protected from directed fishing effort.**

With the UK now having control of its waters following the Brexit arrangements, implementation of a no-take zone for small forage fish is within the competence of the UK government. Because that measure would represent sound management of marine resources and is entirely consistent with ecosystem-based management, it is in line with the Brexit agreement to pursue fisheries management that protects the marine ecosystem. Scotland's Future Fisheries Management Strategy 2020-2030 (Scottish Government 2020), Point 11 in the 12 point plan states "We will work with our stakeholders to deliver an ecosystem-based approach to management, including considering additional protections for spawning and juvenile congregation areas **and restricting fishing activity or prohibiting fishing for species which are integral components of the marine food web, such as sandeels**." (our emphasis). This statement of intent suggests that there is strong support from Scottish Government for improving management of the North Sea sandeel (and other forage fish) fisheries in Scottish waters.

5.2 Eradication of invasive mammal predators

Eradication of invasive non-native mammal predators from seabird islands has been demonstrated globally to be highly effective in restoring healthy populations of some seabird species. This applies in the UK too, with several excellent examples of good practice (Ailsa Craig, Canna, Lundy, Shiants, Ramsay Island, and others). Evidence is strong that several seabird species in the UK are constrained in their use of breeding sites or are impacted by invasive non-native mammals, and that eradication can have strong beneficial effects very rapidly. The evidence is summarised in Table 20.

Table 20. Seabird species in the UK where there is evidence of impacts on populations from invasive non-native mammals, and evidence that eradication leads to conservation gains for the particular seabird species. Green indicates strong evidence of a strong effect. Yellow indicates weak evidence or a weak effect or colony-specific variation depending on local ecology and habitat. No colour indicates that a relationship is unlikely for this combination. The table refers to island colonies and does not include impacts at mainland colonies or impacts of native mammals such as foxes and otters. For completeness, the table includes three species that are not listed in the species accounts (black guillemot, shag, and common gull).

Species	Evidence that breeding or survival is affected by invasive alien mammals	Evidence that eradication of invasive alien mammals results in conservation gains for this species
Manx shearwater	🟩	🟩
European storm-petrel	🟩	🟩
Leach's petrel	🟩	🟩
Black guillemot	🟩	🟩
Puffin	🟩	🟩
Razorbill	🟩	🟩
Common guillemot	🟨	🟨
Shag	🟩	🟨
Arctic tern	🟩	🟨
Common tern	🟩	🟨
Common gull	🟨	🟨
Sandwich tern	🟨	

A strategic programme to eradicate invasive non-native mammals from seabird islands should follow established methods of setting priorities and ranking sites on the basis of these priorities, as outlined by Ratcliffe et al. (2009) and by Stanbury et al. (2017). Developing such a programme would best be carried out by an organisation with appropriate oversight of UK conservation management, such as JNCC. This would allow the programme to be integrated with other actions to eradicate invasive mammal populations for other reasons, and to ensure appropriate emphasis was put on biosecurity measures to minimize risk of recolonisation of sites by non-native mammals.

5.3 Artificial colonies for kittiwakes

This topic is discussed in detail under kittiwake. However, it is mentioned again here because it seems probable that more effective compensation arrangements could be made if that was done strategically rather than as a scramble competitive process by individual offshore wind farm developers. Therefore, although this measure would apply only to a single seabird species, in this case there would be benefit in adopting a strategic approach. We also emphasise that an artificial colony approach will most likely not be appropriate as compensation for kittiwake in areas where natural nest sites are widely available on natural sea cliffs.

In general, it seems likely that artificial nest/colony site provision could be beneficial for several seabird species (examples include nest platforms for divers and common terns, habitat management/creation for Sandwich terns, nest boxes for terns at colonies subject to gull and corvid predation, nest boxes for storm petrels). However, the specific case of artificial colonies for kittiwakes is currently a particular focus because it seems to represent a viable approach to compensation for kittiwake in southeast England in areas where natural nesting habitat is lacking. In principle, should artificial nest sites be considered for other seabird species, the argument that a strategic approach to provision would be likely to be most appropriate would also apply in such cases too.

6 RECOMMENDATIONS

- Given the scale and nature of the compensation likely to be required, we recommend a strategic approach to compensation for seabirds coordinated by Government, in order to optimize the compensation and to align closely with UK Government's Seabird Conservation Strategy, and with Scottish Government's Seabird Conservation Strategy and National Marine Plan, with developer contributions made in line with this approach.
- Where compensation may be project-based rather than strategic, we advise close alignment with UK Government's and Scottish Government's Seabird Conservation Strategies, and consideration of effects of climate change over the time scale of the measure, as well as longer term consideration on predicted changes in seabird distributions as a result of climate change as that may influence which compensation measures would be most appropriate.
- There is strong evidence that sandeel abundance would increase if fishing mortality on sandeels was reduced. There is strong evidence that increased abundance of sandeels would benefit kittiwake populations, and some evidence that it would be likely to benefit many other species, either in terms of breeding success or overwinter survival. We therefore recommend that closure of UK waters to directed fishing for sandeels should be a strategic approach to compensation, noting also that over the longer term, such an approach would benefit UK fishermen in terms of increasing condition of predatory fish such as cod, haddock and whiting.
- There is strong evidence that many seabird species are adversely affected by invasive non-native mammals that have colonised seabird islands. There is also strong evidence that eradicating these invasive mammal populations leads to rapid recovery of affected seabird populations. This therefore represents a highly effective means of compensation that would be relevant for a particular (but limited) set of seabird species. We therefore recommend that this is taken forward as a strategic approach to compensation, but note also that sustained long-term support for biosecurity measures would be essential to prevent recolonisation by invasive mammals.
- We recommend that wider impacts of strategic compensation options should be considered before they are endorsed, ideally via a strategic framework overseen by Government.
- Given that significant conservation gains could be achieved for individual seabird species through a variety of species-specific management measures outlined in this report, we recommend that in cases where these may not be required as compensation, they should be considered as potential management under the UK Government's and Scottish Government's Seabird Conservation Strategies to enhance the conservation status of those seabirds.

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8 ANNEX 1. STATUS OF SPA POPULATIONS WITH SEABIRDS AS BREEDING FEATURES

8.1 Great northern diver

There are no breeding populations of great northern diver in the UK. SPA sites with nonbreeding great northern divers have only been designated very recently and so no Site Condition Monitoring data are available for those sites yet.

8.2 Red-throated diver

Caithness and Sutherland Peatlands SPA; According to SiteLink, red-throated diver on Caithness and Sutherland Peatlands SPA was "Favourable Maintained" when most recently assessed (on 31/07/2006). The citation for Caithness and Sutherland Peatlands SPA (amended February 2017) lists 46 pairs of red-throated divers for this site in 2006.

Foula SPA; According to SiteLink, red-throated diver on Foula SPA was "Favourable Maintained" when most recently assessed (on 28/08/2013). The citation for Foula SPA (27/11/1995) lists 11 pairs of red-throated divers for this site in 1994. There were 11 pairs in 2019 (Sheila Gear, in litt.).

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, red-throated diver on Hermaness, Saxa Vord and Valla Field SPA was "Unfavourable Declining" when most recently assessed (on 02/07/2013). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001) lists 26 pairs of red-throated divers for this site in 1994-1999.

Hoy SPA; According to SiteLink, red-throated diver on Hoy SPA was "Favourable Maintained" when most recently assessed (on 30/08/2007). The citation for Hoy SPA (27/11/1995) lists 58 pairs of red-throated divers for this site.

Lewis Peatlands SPA; According to SiteLink, red-throated diver on Lewis Peatlands SPA was "Unfavourable Declining" when most recently assessed (on 31/08/2004). The citation for Lewis Peatlands SPA (07/12/2000) lists 80 pairs of red-throated divers for this site in 1994-1996.

Mointeach Scadabhaig SPA; According to SiteLink, red-throated diver on Mointeach Scadabhaig SPA was "Favourable Maintained" when most recently assessed (on 28/05/2015). The citation for Mointeach Scadabhaig SPA (Feb. 1999) lists 48 pairs of red-throated divers for this site.

Orkney Mainland Moors SPA; According to SiteLink, red-throated diver on Orkney Mainland Moors SPA was "Favourable Maintained" when most recently assessed (on 31/07/2007). The citation for Orkney Mainland Moors SPA (13/10/2000) lists 18 pairs of red-throated divers for this site.

Otterswick and Graveland SPA; According to SiteLink, red-throated diver on Otterswick and Graveland SPA was "Unfavourable Declining" when most recently assessed (on 12/06/2018). The citation for Otterswick and Graveland SPA (Dec. 2001) lists 26 pairs of red-throated divers for this site in 1992-1999.

Ronas Hill – North Roe and Tingon SPA; According to SiteLink, red-throated diver on Ronas Hill – North Roe and Tingon SPA was "Favourable Declining" when most recently assessed (on 05/06/2014). The citation for Ronas Hill – North Roe and Tingon SPA (27/11/1995) lists 56 pairs of red-throated divers for this site.

Rum SPA; According to SiteLink, red-throated diver on Rum SPA was "Favourable Maintained" when most recently assessed (on 22/08/2013). The citation for Rum SPA (31/08/1982) lists 10 pairs of red-throated divers for this site in 1994. This site was extended on 03/12/2000 to include foraging grounds of breeding birds at sea off Rum.

West Coast of the Outer Hebrides SPA; This site was classified on 03/12/2000 to protect breeding red-throated diver foraging grounds at sea.

Bluemull & Colgrave Sounds SPA; This site was classified on 03/12/2000 to protect breeding red-throated diver foraging grounds at sea.

East Mainland Coast Shetland SPA; This site was classified on 03/12/2000 to protect breeding red-throated diver foraging grounds at sea.

Orkney Waters hold large numbers, and pSPAs are currently being considered for those sites.

In addition, there are SPAs that include red-throated diver as a nonbreeding (wintering) feature. These include Outer Thames Estuary SPA; Greater Wash SPA; Liverpool Bay SPA in England, Firth of Forth SPA and several new sites designated on 03/12/2000 (Solway Firth SPA; Outer Firth of Forth and St Andrews Bay Complex SPA; Moray Firth SPA). Ringing and tracking studies suggest that most red-throated divers that breed in Scotland overwinter in Scottish waters and possibly in Liverpool Bay (Okill 1994; Wernham et al. 2002), whereas almost all of the red-throated divers wintering in the Outer Thames Estuary SPA and Greater Wash SPA originate from breeding areas in Russia and Fenno-Scandia (Dorsch et al. 2019).

8.3 Fulmar

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, fulmar on Hermaness, Saxa Vord and Valla Field SPA was "Favourable Recovered" when most recently assessed (on 20/07/2016). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001, revised September 2009) lists 19,539 pairs of fulmars for this site in 1999 as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 14,890 pairs in the 1990s and 21,079 pairs in the 2000s. There were 13,208 pairs in 2016 (JNCC 2020).

Fetlar SPA; According to SiteLink, fulmar on Fetlar SPA was "Unfavourable Declining" when most recently assessed (on 26/06/2016). The citation for Fetlas SPA (29/03/1994, revised September 2009) lists 9,500 pairs of fulmars for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 9,800 pairs in the 1990s and 8,912 pairs in the 2000s. There were 9,213 pairs in 2016 (Shetland Bird Report for 2016).

Noss SPA; According to SiteLink, fulmar on Noss SPA was "Favourable Maintained" when most recently assessed (on 26/06/2016). The citation for Noss SPA (16/08/1996, revised September 2009) lists 6,350 pairs of fulmars for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 5,870 pairs in the 1990s and 5,248 pairs in the 2000s. There were 5,428 pairs in 2011 and 5,092 pairs in 2016 (JNCC 2020).

Foula SPA; According to SiteLink, fulmar on Foula SPA was "Unfavourable Declining" when most recently assessed (on 24/06/2015). The citation for Foula SPA (27/11/1995, updated September 2009) lists 46,800 pairs of fulmars for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 46,800 pairs in the 1990s and 21,106 pairs in the 2000s. There were 19,758 pairs in 2007 (JNCC 2020), but only 9,137 pairs in 2015 (Shetland Bird Report for 2015) and 8,438 pairs in 2016 (JNCC 2020).

Sumburgh Head SPA; According to SiteLink, fulmar on Sumburgh Head SPA was "Favourable Maintained" when most recently assessed (on 14/06/2017). The citation for Sumburgh Head SPA (27/03/1996, revised September 2009) lists 2,542 pairs of fulmars for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 2,542 pairs in the 1990s and 2,444 pairs in the 2000s. There were 4,431 pairs in 2017 (JNCC 2020).

Fair Isle SPA; According to SiteLink, fulmar on Fair Isle SPA was "Favourable Maintained" when most recently assessed (on 01/06/2016). The citation for Fair Isle SPA (16/12/1994, revised September 2009) lists 35,210 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 43,320 pairs in the 1990s and 29,649 pairs in the 2000s. There were 32,061 pairs in 2016 (JNCC 2020).

West Westray SPA; According to SiteLink, fulmar on West Westray SPA was "Favourable Recovered" when most recently assessed (on 08/06/2017). The citation for West Westray SPA (16/08/1996, revised September 2009) lists 1,400 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,400 pairs in the 1990s and 677 pairs in the 2000s. There were 1,195 pairs in 2017 (JNCC 2020).

Hoy SPA; According to SiteLink, fulmar on Hoy SPA was "Unfavourable No change" when most recently assessed (on 10/06/2017). The citation for Hoy SPA (07/12/2000, revised September 2009) lists 35,000 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 35,000 pairs in the 1990s and 19,586 pairs in the 2000s. There were about 18,000 pairs in 2017 (JNCC 2020).

Calf of Eday SPA; According to SiteLink, fulmar on Calf of Eday SPA was "Favourable Maintained" when most recently assessed (on 08/06/2016). The citation for Calf of Eday SPA (29/06/1998, revised September 2009) lists 1,955 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,955 pairs in the 1990s and 1,842 pairs in the 2000s. There were 1,836 pairs in 2018 (JNCC 2020).

Rousay SPA; According to SiteLink, fulmar on Rousay SPA was "Favourable Maintained" when most recently assessed (on 24/06/2016). The citation for Rousay (02/02/2000, revised September 2009) lists 1,240 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,240 pairs in the 1990s and 535 pairs in the 2000s. There were 2,129 pairs in 2016 (JNCC 2020).

Copinsay SPA; According to SiteLink, fulmar on Copinsay SPA was "Favourable Maintained" when most recently assessed (on 11/06/2015). The citation for Copinsay SPA (29/03/1994, revised September 2009) lists 1,615 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,615 pairs in the 1990s and 1,630 pairs in the 2000s. There were 1,585 pairs in 2015 (JNCC 2020).

North Caithness Cliffs SPA; According to SiteLink, fulmar on North Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 13/06/2016). The citation for North Caithness Cliffs SPA (16/08/1996, revised September 2009, amended 26/04/2018) lists 14,700 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 16,310 pairs in the 1990s and 14,250 pairs in the 2000s. There were about 14,000 pairs in 2015-2016 (JNCC 2020).

East Caithness Cliffs SPA; According to SiteLink, fulmar on East Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 30/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996, revised September 2009, amended March 2017) lists 15,000 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 15,000 pairs in the 1990s and 14,202 pairs in the 2000s. There were 13,864 pairs in 2015 (Swann 2016).

Troup, Pennan and Lion's Heads SPA; According to SiteLink, fulmar on Troup, Pennan and Lion's Heads SPA was "Unfavourable No change" when most recently assessed (on 13/07/2017). The citation for Troup, Pennan and Lion's Heads SPA (14/03/1997, revised September 2009) lists 4,400 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 4,400 pairs in the 1990s and 1,795 pairs in the 2000s. There were 1,894 pairs in 2017 (JNCC 2020).

Buchan Ness to Collieston Coast SPA; According to SiteLink, fulmar on Buchan Ness to Collieston Coast SPA was "Unfavourable Declining" when most recently assessed (on 16/06/2017). The citation for Buchan Ness to Collieston Coast SPA (30/03/1998, revised September 2009) lists 1,765 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,765 pairs in the 1990s and 1,389 pairs in the 2000s. There were 826 pairs in 2019 (JNCC 2020).

Fowlsheugh SPA; According to SiteLink, According to SiteLink, fulmar on Fowlsheugh SPA was "Favourable Maintained" when most recently assessed (on 18/06/1999). The citation for Fowlsheugh SPA (31/08/1992, revised September 2009) lists 1,170 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,170 pairs in the 1990s and 193 pairs in the 2000s. There were 525 pairs in 2018 (JNCC 2020).

Forth Islands SPA; Stroud et al. (2016) lists Forth Islands SPA as a member of the SPA suite for breeding fulmar. However, fulmar is not named in the citation (25/04/1990, extended 16/02/2004, amended 25/05/2018) and is not included in the SiteLink entry for Forth Islands SPA. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,600 pairs in the 1990s and 676 pairs in the 2000s. There were about 650 pairs in 2018-2019 (JNCC 2020).

North Rona and Sula Sgeir SPA; According to SiteLink, fulmar on North Rona and Sula Sgeir SPA was "Unfavourable Declining" when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001, revised September 2009) lists 11,500 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 11,500 pairs in the 1990s but no count available for the 2000s. There were 1,438 pairs in 2012 (JNCC 2020).

Cape Wrath SPA; According to SiteLink, fulmar on Cape Wrath SPA was "Unfavourable Declining" when most recently assessed (on 04/06/2017). The citation for Cape Wrath SPA (15/03/1996, revised September 2009) lists 2,300 pairs of fulmar for this site as a component of the seabird assemblage.

Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 2,300 pairs in the 1990s and 2,115 pairs in the 2000s. There were 1,477 pairs in 2017 (JNCC 2020).

Flannan Isles SPA; According to SiteLink, fulmar on Flannan Isles SPA was “Unfavourable Recovering” when most recently assessed (on 03/06/2013). The citation for Flannan Isles SPA (31/08/1992, revised September 2009) lists 4,730 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 4,700 pairs in the 1990s and 8,143 pairs in the 2000s. There were no counts for this site since 2000 in the JNCC database, but the SCM monitoring report lists 2,263 pairs in 2013 (JNCC 2020).

Handa SPA; According to SiteLink, fulmar on Handa SPA was “Unfavourable No change” when most recently assessed (on 20/06/2012). The citation for Handa SPA (25/04/1990, revised September 2009) lists 3,500 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 3,500 pairs in the 1990s and 1,915 pairs in the 2000s. There were 1,870 pairs in 2012, and 1,423 pairs in 2017 (JNCC 2020).

St Kilda SPA; According to SiteLink, fulmar on St Kilda SPA was “Unfavourable Declining” when most recently assessed (on 08/06/2016). The citation for St Kilda SPA (31/08/1992, revised September 2009) lists 62,800 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 62,800 pairs in the 1990s and 66,055 pairs in the 2000s. There were 29,186 pairs in 2015-2016 (JNCC 2020).

Mingulay and Berneray SPA; According to SiteLink, fulmar on Mingulay and Berneray SPA was “Favourable Maintained” when most recently assessed (on 21/06/2014). The citation for Mingulay and Berneray SPA (16/12/1994, revised September 2009) lists 10,450 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 12,500 pairs in the 1990s and 9,046 pairs in the 2000s. The most recent counts in the JNCC database are 6,292 pairs on Mingulay in 2017 and 7,191 there in 2014, and 1,423 pairs on Berneray in 2014, giving a total of 8,614 pairs in 2014 (JNCC 2020).

Shiant Isles SPA; According to SiteLink, fulmar on Shiant Isles SPA was “Unfavourable Declining” when most recently assessed (on 18/06/2015). The citation for Shiant Isles SPA (31/08/1992, revised September 2009) lists 6,820 pairs of fulmar for this site as a component of the seabird assemblage. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 6,820 pairs in the 1990s and 4,387 pairs in the 2000s. There were 1,506 pairs in 2015 (JNCC 2020).

Rathlin Island SPA; The citation for Rathlin Island SPA (25/02/1999) lists 1,482 pairs of fulmar in 1985 as a component of the seabird assemblage at this site. Stroud et al. (2016) report breeding numbers of fulmars in this SPA as 1,482 pairs in the 1990s and 1,518 pairs in the 2000s. There were 1,518 pairs in 2011 (JNCC 2020).

8.4 Manx shearwater

Rum SPA: The accuracy of colony counts is low and population trends are highly uncertain on Rum, where many birds nest in boulder screes on steep mountain slopes rather than in burrows. According to SiteLink, Manx shearwater on Rum SPA was “Favourable Maintained” when most recently assessed (on 31/05/2003). The Citation for Rum SPA (31/08/1982) lists 61,000 pairs of Manx shearwaters for this site. Wormell (1976) estimated 116,000 pairs in 1965-69, Thompson and Thompson (1980) estimated 150,000 pairs in 1980, Philips (1982) estimated 79,000 pairs in 1982, Furness (1990) estimated 62,800 pairs in 1990, and Seabird2000 reported 120,000 pairs in 2001 based on a visual assessment (Murray and Shewry 2002), and 112,600 pairs based on tape-playback (Mitchell et al. 2004). These various estimates give no clear picture of any long-term trend in numbers, and also demonstrate the difficulty in assessing the size of this colony. However, there are brown rats on Rum and there have been suggestions that predation by rats may affect the shearwater population there.

St Kilda SPA: According to SiteLink, Manx shearwater on St Kilda SPA was “Favourable Maintained” when most recently assessed (on 31/07/2000). The citation for St Kilda SPA (31/08/1992) lists up to 5,000 pairs of Manx shearwaters for this site. Seabird2000 reported that there were at least 4,800 pairs at St Kilda (Mitchell et al. 2004).

Copeland Islands SPA (Northern Ireland): The Copeland Islands held 4,633 pairs in 2000-03 (Mitchell et al. 2004). Copeland Islands SPA was designated in September 2009, with an estimated 5,923 pairs of Manx shearwaters there at designation (Stroud et al. 2016). There were 4,850 AOS (pairs) in 2007 (JNCC 2020).

Aberdaron Coast and Bardsey Island SPA (Wales): This site was designated in June 1992, with the Natura standard data form updated in December 2015. There were thought to be 6,930 pairs at designation in the 1990s (Stroud et al. 2016). It held about 16,000 pairs in 2001 but had been thought to hold 4,000 to 5,000 pairs in 1972-1982 (Mitchell et al. 2004).

Skomer, Skokholm and seas off Pembrokeshire SPA (Wales): This site, comprising three colonies, was designated in August 1982, with the Natura standard data form updated in January 2017 and the site renamed. The Standard data form states that it held an estimated 150,968 AOS (pairs). These were counted in 1997-1998, with 67% of the pairs on Skomer (Mitchell et al. 2004). However, a repeat census at Skomer in 2011 suggested a population of 316,000 pairs (Perrins et al. 2012), indicating the uncertainty in population estimates for this species. More recently, Perrins et al. (2019) re-surveyed the colonies at Skokholm, Skomer and Midland Island (also known as Middleholm) in 2018 and estimated the total at these three sites to be 456,000 pairs.

8.5 European storm-petrel

Mousa SPA (Shetland): The SPA citation (October 1994) lists 4,750 pairs of European storm-petrels. Seabird2000 reports a count of 6,800 pairs in 1996 based on tape playback responses (Mitchell et al. 2004), but that included calculation errors, corrected by Bolton et al. (2010) to 5,410 AOS. Resurvey in 2008 using the same method found 11,781 AOSs and population modelling predicted a similar increase (Bolton et al. 2010). The colony was estimated to have held almost 11,000 pairs in 2015, suggesting no increase in numbers between 2008 and 2015 (Bolton et al. 2017). The feature was assessed as “Favourable Maintained” on 31/07/2015.

Auskerry SPA (Orkney): The SPA citation (site designated 29/03/1994) lists 3,600 pairs of European storm-petrels. Seabird2000 reported 994 AOS in 2001 (Mitchell et al. 2004). The feature was assessed as “Favourable but Declining” on 17/07/2018, but JNCC presents no data for the SPA site since 2001 (JNCC 2020).

Sule Skerry and Sule Stack SPA (Orkney): The SPA citation (August 1996) lists 500 to 5,000 pairs of European storm-petrels. Seabird2000 reported 309 AOS in 2001 (Mitchell et al. 2004). The feature was assessed as "Favourable but Declining" on 19/07/2018, but JNCC presents no data for the SPA site since 2001 (JNCC 2020).

North Rona and Sula Sgeir SPA: The SPA citation simply states >1% of GB in 1986 but does not cite the numbers present; Lloyd et al. (1991) suggest possibly 'thousands or tens of thousands of pairs'. Seabird2000 reported 377 AOS in 2001 (Mitchell et al. 2004). Murray et al. (2010) suggested that similar numbers were present in 2009 based on an incomplete survey that year. The feature was assessed as "Favourable Maintained" on 29/06/2009, presumably based on the incomplete survey and unclear numbers at site designation. There have been no population estimates since then (JNCC 2020).

St Kilda SPA: The SPA citation (site designated 31/08/1992) lists 850 pairs. Seabird2000 reported 1,121 AOS in 1999/2000 (Mitchell et al. 2004). The feature was assessed as "Favourable Maintained" on 31/07/2000, and JNCC suggests that there have been no population estimates since then (JNCC 2020).

Priest Island: The SPA citation lists 2,200 pairs in 1995. Seabird2000 reported 4,400 AOS in 1999 (Mitchell et al. 2004). The feature was assessed as "Favourable Maintained" on 15/07/1999. Hounsome et al. (2006) used tape playback to estimate numbers in 2004 and found 2,437 AOS. However, a mark-recapture analysis suggested a larger population size than indicated from tape playback. There were estimates of 2,168 AOS in 2009, 4,259 AOS in 2014 and 4,640 AOS in 2019 (JNCC 2020).

Treshnish Isles SPA (Mull): The SPA citation lists 5,040 AOS in 1996, which is also the only count included in Seabird2000 for this site (Mitchell et al. 2004). The feature was assessed as "Favourable Maintained" on 31/07/2018. An estimate of 8,675 AOS in 2018 suggests an increase in the population there (JNCC 2020).

Skomer, Skokholm and seas off Pembrokeshire SPA: This was designated in August 1982, with the Natura standard data form updated in January 2017. At designation there were 3,500 pairs of European storm-petrels. Mitchell et al. (2004) suggested that there may have been a long-term decline in European storm-petrel breeding numbers at Skokholm and Skomer. There were estimated to be 5,000 to 7,000 pairs at Skokholm in 1969 but only 2,450 pairs in 2001. At Skomer there were estimated to be 500 pairs in 1991 but only 110 pairs in 2000. There are no more recent counts in the JNCC SCM database (JNCC 2020).

Isles of Scilly SPA: This was designated in August 2001, with the Natura standard data form updated in December 2015. The data form lists 5,406 pairs of European storm-petrel. The citation lists 5,406-8,798 pairs in 1999, with the data derived from Robinson (1999) which is an unpublished report with the data not included in the JNCC SCM database. The main colony on the Isles of Scilly is at Annet, which was estimated to hold 938 pairs in 2000; elsewhere in the archipelago there were another 500 or so pairs across ten colonies (Mitchell et al. 2004), so it is not clear where the total of 5,406 pairs comes from. The most recent counts (AOS, or pairs) in the JNCC SCM database (for 2015-2019) are: Annet 778, Round Island 172, Rosevear 112, Melledgan 97, Illiswilgig 52, Rosevean 26, Scilly Rock 21, Gugh 11, Mincario 9, Castle Bryher 3. These sum to 1,281 pairs (JNCC 2020). Trends, if any, are uncertain as detailed quantitative data are lacking for those colonies before 2000, but it seems likely that numbers have been reduced by the presence of mammal predators in most of the archipelago.

8.6 Leach's petrel

Ramna Stacks and Gruney SPA (Shetland): This small colony was first found in 1980 (Fowler and Butler 1982). The SPA citation states 20 pairs at this SPA and Mitchell et al. (2004) list 20 AOS on Gruney in 2000. The feature was assessed as "Unfavourable Declining" on 29/06/2018, but JNCC presents no recent data for Leach's petrel at the SPA site (JNCC 2020). Miles et al. (2012) found only four occupied burrows with nonbreeders rather than breeders.

Foula SPA (Shetland): The SPA citation lists 50 pairs of Leach's petrel. Breeding was first confirmed in 1974 but numbers were not censused until 2001, when there were thought to be at least 3 to 30 AOS, but with much suitable habitat inaccessible (Mitchell et al. 2004). Latest assessed condition in SiteLink was "Unfavourable Declining" on 22/09/2001. However, this population is thought almost certain to be extinct at Foula now as a consequence of predation by feral cats (S. Gear, in litt.).

Sule Skerry and Sule Stack SPA (Orkney): Lloyd et al. (1991) reported more than 5 AOS in 1986. The SPA citation lists 5 pairs of Leach's petrel (the site was classified in March 1994). However, searches in 2001 failed to find any. Latest assessed condition in SiteLink was "Unfavourable Declining" on 17/07/2018. JNCC presents no recent data for Leach's petrel at this SPA (JNCC 2020).

North Rona and Sula Sgeir SPA: The SPA citation simply states >1% of GB in 1986 but does not cite the numbers present. Lloyd et al. (1991) suggested about 500 pairs at North Rona and 15 pairs at Sule Sgeir. Mitchell et al. (2004) reported 1,132 AOS on North Rona and 5 AOS on Sule Sgeir in 2001 surveyed using tape playback. Murray et al. (2010) also surveyed using tape playback to estimate 713 AOS on North Rona and none on Sule Sgeir. Latest assessed condition in SiteLink was "Unfavourable Declining" on 18/06/2012.

Flannan Isles SPA: The SPA citation lists 100 to 1,000 pairs of Leach's petrel (the site was classified in August 1992). Numbers have not been censused in detail before Seabird2000. In 2001 there were estimated to be 1,425 AOS on the six islands with suitable habitat (Mitchell et al. 2004). Latest assessed condition in SiteLink was "Favourable Maintained" on 31/07/2001. JNCC presents no more recent data for Leach's petrel at this SPA (JNCC 2020).

St Kilda SPA: The SPA citation lists 5,000 pairs of Leach's petrel (the site was classified in August 1992). Mitchell et al. (2004) list 45,433 AOS in 1999/2000, with the largest numbers on Dun (27,704) and Boreray (12,093). Latest assessed condition in SiteLink was "Favourable Maintained" on 31/07/2000. JNCC presents no more recent data for Leach's petrel at this SPA (JNCC 2020). However, surveys on Dun, St Kilda, which was considered to hold about 58% of the UK population of this species (JNCC 2020) found a decrease from 27,704 pairs in 1999 to 14,500 pairs in 2003 and 12,800 pairs in 2006 (Newson et al. 2008), attributed at least in part to predation by great skuas (Miles 2010).

8.7 Gannet

Ailsa Craig SPA; According to SiteLink, gannet on Ailsa Craig SPA was "Favourable Maintained" when most recently assessed (on 06/06/2004). The citation for Ailsa Craig SPA (25/04/1990) lists 23,000 pairs of gannets for this site. There were 33,226 pairs in 2014 (Murray et al. 2015).

Fair Isle SPA; According to SiteLink, gannet on Fair Isle SPA was "Favourable Maintained" when most recently assessed (on 01/06/2014). The citation for Fair Isle SPA (16/12/1994) lists 1,166 pairs of gannets for this site. There were 3,591 pairs in 2014 (Murray et al. 2015) and 4,211 pairs in 2019 (JNCC 2020).

Forth Islands SPA; According to SiteLink, gannet on Forth Islands SPA was "Favourable Maintained" when most recently assessed (on 23/06/2014). The citation for Forth Islands SPA (25/04/1990) lists 21,600 pairs of gannets for this site (all on Bass Rock). There were 75,259 pairs in 2014 (Murray et al. 2015).

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, gannet on Hermaness, Saxa Vord and Valla Field SPA was "Favourable Maintained" when most recently assessed (on 24/10/2014). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001) lists 16,400 pairs of gannets for this site in 1999. There were 25,580 pairs in 2014 (Murray et al. 2015).

North Rona and Sula Sgeir SPA; According to SiteLink, gannet on North Rona and Sula Sgeir SPA was "Favourable Maintained" when most recently assessed (on 18/06/2013). The citation for North Rona and Sula Sgeir SPA (30/10/2001) lists 10,400 pairs of gannets for this site. There were 11,230 pairs in 2013 (Murray et al. 2015).

Noss SPA; According to SiteLink, gannet on Noss SPA was "Favourable Maintained" when most recently assessed (on 01/06/2014). The citation for Noss SPA (16/08/1996) lists 6,860 pairs of gannets for this site. There were 11,786 pairs in 2014 (Murray et al. 2015), and 13,765 pairs in 2019 (JNCC 2020).

St Kilda SPA; According to SiteLink, gannet on St Kilda SPA was "Favourable Maintained" when most recently assessed (on 19/06/2013). The citation for St Kilda SPA (31/08/1992) lists 50,050 pairs of gannets for this site. There were 60,290 pairs in 2013 (Murray et al. 2015).

Sule Skerry and Sule Stack SPA; According to SiteLink, gannet on Sule Skerry and Sule Stack SPA was "Favourable Maintained" when most recently assessed (on 18/06/2013). The citation for Sule Skerry and Sule Stack SPA (29/03/1994) lists 5,900 pairs of gannets for this site. There were 6,420 pairs in 2013 (Murray et al. 2015).

Flamborough and Filey Coast SPA; The citation for Flamborough and Filey Coast SPA (August 2018) lists 8,469 pairs of gannets for this site in 2008-2012. There were 13,392 pairs in 2017 (JNCC 2020), so this feature is "Favourable Maintained".

Grassholm SPA; The citation for Grassholm SPA (November 1995) lists 26,300 pairs of gannets for this site in 1994. There were 36,011 pairs in 2015 (JNCC 2020), so this feature is "Favourable Maintained".

8.8 Arctic skua

Fetlar SPA: The SPA citation lists 130 pairs of Arctic skua (the site was classified in March 1994). The feature was assessed as "Unfavourable Declining" on 21/06/2017. There were 152 AOTs in 1992, 96 in 2001, 9 AOTs in 2017 (JNCC 2020).

Foula SPA: The SPA citation lists 133 pairs of Arctic skua (the site was classified in November 1995). The feature was assessed as “Unfavourable Declining” on 01/06/2015. There were 28 AOTs in 2015, 23 in 2017, 20 in 2018, declining further to 19 in 2019 (S. Gear, annual monitoring reports to SOTEAG).

Fair Isle SPA: The SPA citation lists 110 pairs of Arctic skua (the site was classified in December 1994). The feature was assessed as “Unfavourable Declining” on 01/06/2016. There were 30 AOTs in 2017, 28 in 2018 (Shetland Bird Club 2020).

West Westray SPA: The SPA citation lists 78 pairs of Arctic skua (the site was classified in August 1996). The feature was assessed as “Unfavourable Declining” on 21/06/2017. Perkins et al. (2018) list 17 pairs there in 2008.

Papa Westray SPA: The SPA citation lists 135 pairs of Arctic skua (the site was classified in March 1996). The feature was assessed as “Unfavourable Declining” on 24/07/2015, when there were 25 AOTs (Perkins et al. 2018).

Rousay SPA: The SPA citation lists 130 pairs of Arctic skua (the site was classified in February 2000). The feature was assessed as “Unfavourable No change” on 24/06/2015. There were 37 AOTs in 2010 (Perkins et al. 2018).

Hoy SPA: The SPA citation lists 59 pairs of Arctic skua (the site was classified in December 2000). The feature was assessed as “Unfavourable Declining” on 08/07/2019. There were 12 AOTs on Hoy in 2010 (Perkins et al. 2018).

8.9 Great skua

Fair Isle SPA: The SPA citation lists 110 pairs of great skua (the site was classified in 1994). The feature was assessed as “Favourable Maintained” on 01/06/2016. There were 490 AOTs in 2019 (JNCC 2020).

Fetlar SPA: The SPA citation lists 508 pairs of great skua (the site was classified in 1994). The feature was assessed as “Favourable Maintained” on 21/06/2017. There were 743 AOTs in 2017 (JNCC 2020).

Foula SPA: The SPA citation lists 2,270 pairs of great skua (the site was classified in 1995). The feature was assessed as “Favourable Recovered” on 05/06/2015. There were 1,846 AOTs in 2015 (JNCC 2020).

Handa SPA: The SPA citation lists 66 pairs of great skua (the site was classified in 1990). The feature was assessed as “Favourable Maintained” on 28/06/2013. There were 283 AOTs in 2018 (JNCC 2020).

Hermaness, Saxavord and Vallafield SPA: The SPA citation lists 788 pairs of great skua (the site was classified in 2001). The feature was assessed as “Favourable Maintained” on 25/06/2013. There were 955 AOTs in 2018 (JNCC 2020).

Hoy SPA: The SPA citation lists 1,900 pairs of great skua (the site was classified in 1997). The feature was assessed as “Unfavourable Declining” on 08/07/2019. There were 1,063 AOTs in 2019 (JNCC 2020).

Noss SPA: The SPA citation lists 420 pairs of great skua (the site was classified in 1996). The feature was assessed as "Favourable Maintained" on 13/08/2013. There were 476 AOTs in 2018 (JNCC 2020).

Ronas Hill SPA: The SPA citation lists 130 pairs of great skua (the site was classified in 1994). The feature was assessed as "Favourable Maintained" on 14/06/2017. There were 289 AOTs in 2017 (JNCC 2020).

St Kilda SPA: The SPA citation lists 270 pairs of great skua (the site was classified in 1992). The feature was assessed as "Favourable Maintained" on 31/07/2016. There were 179 AOTs in 2012-2019 (JNCC 2020).

8.10 Lesser black-backed gull

Forth Islands SPA: According to SiteLink, lesser black-backed gull on Forth Islands SPA was "Favourable Maintained" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (amended 25/05/2018) lists 1,500 pairs of lesser black-backed gulls for this site in 1986-1988. There were 2,051 pairs in 2018-2019, mostly at the Isle of May (JNCC 2020).

Ailsa Craig SPA; According to SiteLink, lesser black-backed gull on Ailsa Craig SPA was "Unfavourable No Change" when most recently assessed (on 03/06/2017). The citation for Ailsa Craig SPA (25/04/1990) lists 1,800 pairs of lesser black-backed gulls for this site. There were 189 pairs in 2019 (JNCC 2020).

Rathlin Island; The citation for Rathlin Island SPA (25/02/1999) lists 155 pairs of lesser black-backed gulls in the species assemblage for this site. There were 143 pairs in 2011 (JNCC 2020).

Lough Neagh and Lough Beg; The citation for Lough Neagh and Lough Beg SPA (10/04/1996) lists breeding lesser black-backed gulls in the species assemblage for this site without indicating the numbers of breeding pairs. There were at least 40 pairs breeding there in 2018 (JNCC 2020).

Morecambe Bay and Duddon Estuary SPA; The citation for Morecambe Bay and Duddon Estuary SPA lists 9,720 individuals of breeding lesser black-backed gulls for this site (2011-2015). Most breed at South Walney, where numbers fell from 12,200 pairs in 2004 to 8,130 pairs in 2011, 2,782 pairs in 2017 and 390 pairs in 2019 (JNCC 2020).

Bowland Fells SPA; The Bowland Fells SPA citation lists 4,575 pairs of lesser black-backed gulls for this site in 2009-2012. In 2018, there were 9,054 pairs at Tarnbrook Fell and 5,573 pairs at Langden Head, so at least 14,627 pairs in this SPA (JNCC 2020), implying "Favourable" conservation status.

Ribble and Alt Estuaries SPA; The citation for Ribble and Alt Estuaries SPA compiled in November 2002 lists 1,800 pairs of lesser black-backed gulls for this site in 1993. There were 7,022 pairs on the Ribble Estuary in 2016 (JNCC 2020), suggesting "Favourable" conservation status.

Skomer, Skokholm and seas off Pembrokeshire SPA; The citation for Skokholm and Skomer SPA lists 20,300 pairs of lesser black-backed gulls for this site (mean 1993-1997). JNCC (2020) give the most recent counts as 5,410 pairs on Skomer in 2018, 1,008 pairs on Skokholm in 2019 and 43 pairs on Middleholm in 2009. Those counts suggest that the numbers are now less than one-third the numbers present at designation, indicating "Unfavourable" conservation status.

Isles of Scilly SPA; The citation for Isles of Scilly SPA (revised November 2020) lists 4,922 individuals of breeding lesser black-backed gulls for this site in 2015/2016, or 3,608 pairs in 1999 in the June 2001 citation version 1.5. The most recent counts in the JNCC SCM database (mostly from 2015) sum to 2,465 pairs for the SPA (JNCC 2020).

Alde-Ore Estuary SPA; The Alde-Ore Estuary SPA Natura 2000 Standard Data Form lists 14,070 pairs of lesser black-backed gulls for this site (5-year mean 1994-1998). There were 97 pairs at Orfordness and 1,327 pairs at Havergate Island in 2018 so a total of 1,424 pairs for the SPA (JNCC 2020), indicating "Unfavourable" conservation status.

8.11 Herring gull

East Caithness Cliffs SPA; According to SiteLink, herring gull on East Caithness Cliffs SPA was "Unfavourable No Change" when most recently assessed (on 30/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996, amended March 2017) lists 9,400 pairs of herring gulls for this site. There were 3,267 pairs in 2015 (Swann 2016).

Troup, Pennan and Lion's Heads SPA; According to SiteLink, herring gull on Troup, Pennan and Lion's Heads SPA was "Unfavourable Declining" when most recently assessed (on 13/07/2017). The citation for Troup, Pennan and Lion's Heads SPA (14/03/1997) lists 4,200 pairs of herring gulls for this site. There were 546 pairs in 2017 (JNCC 2020).

Buchan Ness to Collieston Coast SPA; According to SiteLink, herring gull on Buchan Ness to Collieston Coast SPA was "Unfavourable No Change" when most recently assessed (on 16/06/2017). The citation for Buchan Ness to Collieston Coast SPA (30/03/1998) lists 4,292 pairs of herring gulls for this site. There were 2,077 pairs in 2019 (JNCC 2020).

Fowlsheugh SPA; According to SiteLink, herring gull on Fowlsheugh SPA was "Unfavourable Declining" when most recently assessed (on 18/06/1999). The citation for Fowlsheugh SPA (31/08/1992) lists 3,190 pairs of herring gulls for this site. There were 1,035 pairs in 2018 (JNCC 2020).

Forth Islands SPA; According to SiteLink, herring gull on Forth Islands SPA was "Favourable Maintained" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990, amended 25/05/2018) lists 6,600 pairs of herring gulls for this site. There were 5,962 pairs in 2018-2019 (JNCC 2020).

St Abb's Head to Fast Castle SPA; According to SiteLink, herring gull on St Abb's Head to Fast Castle SPA was "Unfavourable Declining" when most recently assessed (on 31/05/2014). The citation for St Abb's Head to Fast Castle SPA (11/08/1997) lists 1,160 pairs of herring gulls for this site. There were 283 pairs in 2018 (JNCC 2020).

Canna and Sanday SPA; According to SiteLink, herring gull on Canna and Sanday SPA was "Unfavourable Declining" when most recently assessed (on 17/02/2014). The citation for Canna and Sanday SPA (20/02/1998) lists 1,300 pairs of herring gulls for this site. There were 99 pairs in 2019 (JNCC 2020).

Ailsa Craig SPA; According to SiteLink, herring gull on Ailsa Craig SPA was "Unfavourable No Change" when most recently assessed (on 03/06/2017). The citation for Ailsa Craig SPA (25/04/1990) lists 2,250 pairs of herring gulls for this site. There were 213 pairs in 2019 (JNCC 2020).

Flamborough and Filey Coast SPA; The citation for Flamborough and Filey Coast SPA (August 2018) lists the seabird assemblage but does not specifically list herring gull. However, Stroud et al. (2016) list this site as qualifying for herring gull, with 1,110 pairs in the 1990s. JNCC (2020) gives the latest count as 351 pairs at Flamborough Head and Bempton Cliffs in 2017.

Alde-Ore Estuary SPA; The citation for Alde-Ore Estuary SPA (January 1996) lists breeding herring gull at this site representing over 1% of the British breeding population but does not give the numbers present. JNCC (2020) gives the latest counts as 523 pairs in Havergate Island in 2019 and 68 pairs in Orfordness in 2018.

Rathlin Island; The citation for Rathlin Island SPA (March 1999) lists 4,037 pairs of herring gulls for this site. JNCC (2020) gives the latest count as 28 pairs in 2011.

Morecambe Bay and Duddon Estuary SPA; The citation for Morecambe Bay and Duddon Estuary SPA lists 20,000 individuals of breeding herring gulls for this site in 1991. Stroud et al. (2016) list this site as qualifying for herring gull, with 11,000 pairs in the 1990s. JNCC (2020) gives the latest counts within this SPA as 444 pairs in South Walney in 2020, 1 pair at Hodbarrow in 2016 and 6 pairs at Chapel Island in 2018.

8.12 Great black-backed gull

Calf of Eday SPA; According to SiteLink, great black-backed gull on Calf of Eday SPA was “Unfavourable Declining” when most recently assessed (on 01/06/2016). The citation for Calf of Eday SPA (29/06/1998) lists 938 pairs of great black-backed gulls for this site. There were 60 pairs in 2018 (JNCC 2020).

Hoy SPA; According to SiteLink, great black-backed gull on Hoy SPA was “Unfavourable Declining” when most recently assessed (on 08/07/2019). The citation for Hoy SPA (07/12/2000) lists 570 pairs of great black-backed gulls for this site. There were 50 pairs in 2018 (JNCC 2020).

Copinsay SPA; According to SiteLink, great black-backed gull on Copinsay SPA was “Unfavourable Declining” when most recently assessed (on 11/06/2015). The citation for Copinsay SPA (29/03/1994) lists 490 pairs of great black-backed gulls for this site. There were 84 pairs in 2015 (JNCC 2020).

East Caithness Cliffs SPA; According to SiteLink, great black-backed gull on East Caithness Cliffs SPA was “Unfavourable No Change” when most recently assessed (on 30/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996) lists 800 pairs of great black-backed gulls for this site. There were 266 pairs in 2015 (Swann 2016).

North Rona and Sula Sgeir SPA; According to SiteLink, great black-backed gull on North Rona and Sula Sgeir SPA was “Unfavourable Declining” when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001) lists 730 pairs of great black-backed gulls for this site. There were 191 pairs in 2012 (JNCC 2020).

Isles of Scilly SPA; The original citation (June 2001 version 1.5) gives no numbers of breeding great black-backed gulls but names the species as a component of the seabird assemblage. The revised citation for Isles of Scilly SPA (compiled November 2020) lists 1,882 individuals of breeding great black-backed gulls for this site in 2015-2016, which approximately matches up with the JNCC SCM data indicating about 900 pairs in 2015-2016 (JNCC 2020). There were 999 pairs in 1985-1988 and 807 pairs in 1998-2002 (Mitchell et al. 2004), so breeding numbers here seem to have remained fairly similar across recent decades.

8.13 Kittiwake

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, kittiwake on Hermaness, Saxa Vord and Valla Field SPA was “Unfavourable Declining” when most recently assessed (on 11/06/2017). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001) lists 922 pairs of kittiwakes for this site. There were 200 pairs in 2016 (JNCC 2020).

Foula SPA; According to SiteLink, kittiwake on Foula SPA was “Unfavourable Declining” when most recently assessed (on 24/06/2015). The citation for Foula SPA (27/11/1995) lists 3,840 pairs of kittiwakes for this site. There were 259 pairs in 2019 and 308 pairs in 2020 (Sheila Gear, annual monitoring reports to SOTEAG).

Noss SPA; According to SiteLink, kittiwake on Noss SPA was “Unfavourable Declining” when most recently assessed (on 23/06/2015). The citation for Noss SPA (16/08/1996) lists 7,020 pairs of kittiwakes for this site. There were 76 pairs in 2019 (JNCC 2020).

Sumburgh Head SPA; According to SiteLink, kittiwake on Sumburgh Head SPA was “Unfavourable Declining” when most recently assessed (on 14/06/2017). The citation for Sumburgh Head SPA (27/03/1996) lists 1,366 pairs of kittiwakes for this site. There were 241 pairs in 2017 (JNCC 2020).

Fair Isle SPA; According to SiteLink, kittiwake on Fair Isle SPA was “Unfavourable Declining” when most recently assessed (on 01/06/2016). The citation for Fair Isle SPA (16/12/1994) lists 18,160 pairs of kittiwakes for this site. There were 859 pairs in 2015 (JNCC 2020).

West Westray SPA; According to SiteLink, kittiwake on West Westray SPA was “Unfavourable Declining” when most recently assessed (on 08/06/2017). The citation for West Westray SPA (16/08/1996) lists 23,900 pairs of kittiwakes for this site. There were 2,755 pairs in 2017 (JNCC 2020).

Rousay SPA; According to SiteLink, kittiwake on Rousay SPA was “Unfavourable Declining” when most recently assessed (on 24/06/2016). The citation for Rousay SPA (02/02/2000) lists 4,900 pairs of kittiwakes for this site. There were 330 pairs in 2016 (JNCC 2020).

Calf of Eday SPA; According to SiteLink, kittiwake on Calf of Eday SPA was “Unfavourable Declining” when most recently assessed (on 08/06/2016). The citation for Calf of Eday SPA (29/06/1998) lists 1,717 pairs of kittiwakes for this site. There were 142 pairs in 2018 (JNCC 2020).

Copinsay SPA; According to SiteLink, kittiwake on Copinsay SPA was “Unfavourable Declining” when most recently assessed (on 11/06/2015). The citation for Copinsay SPA (29/03/1994) lists 9,550 pairs of kittiwakes for this site. There were 955 pairs in 2015 (JNCC 2020).

Hoy SPA; According to SiteLink, kittiwake on Hoy SPA was “Unfavourable Declining” when most recently assessed (on 10/06/2017). The citation for Hoy SPA (07/12/2000) lists 3,000 pairs of kittiwakes for this site. JNCC (2020) lists no complete counts since 1999.

Marwick Head SPA; According to SiteLink, kittiwake on Marwick Head SPA was "Unfavourable Declining" when most recently assessed (on 15/06/2015). The citation for Marwick Head SPA (16/12/1994) lists 7,700 pairs of kittiwakes for this site. There were 906 pairs in 2018 (JNCC 2020).

North Caithness Cliffs SPA; According to SiteLink, kittiwake on North Caithness Cliffs SPA was "Unfavourable Declining" when most recently assessed (on 13/06/2016). The citation for North Caithness Cliffs SPA (16/08/1996) lists 13,100 pairs of kittiwakes for this site.

East Caithness Cliffs SPA; According to SiteLink, kittiwake on East Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 17/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996) lists 32,500 pairs of kittiwakes for this site. There were 24,460 pairs in 2015 (Swann 2016).

Buchan Ness to Collieston Coast SPA; According to SiteLink, kittiwake on Buchan Ness to Collieston Coast SPA was "Unfavourable No Change" when most recently assessed (on 16/06/2017). The citation for Buchan Ness to Collieston Coast SPA (30/03/1998) lists 30,452 pairs of kittiwakes for this site. There were 11,295 pairs in 2019 (JNCC 2020).

Troup, Pennan and Lion's Heads SPA; According to SiteLink, kittiwake on Troup, Pennan and Lion's Heads SPA was "Unfavourable No Change" when most recently assessed (on 03/07/2007). The citation for Troup, Pennan and Lion's Heads SPA (14/03/1997) lists 31,600 pairs of kittiwakes for this site. There were 10,503 pairs in 2017 (JNCC 2020).

Fowlsheugh SPA; According to SiteLink, kittiwake on Fowlsheugh SPA was "Favourable Maintained" when most recently assessed (on 11/06/1999). The citation for Fowlsheugh SPA (31/08/1992) lists 36,650 pairs of kittiwakes for this site. There were 9,444 pairs in 2018 (JNCC 2020) so this SPA feature should now be considered "Unfavourable Declining".

Forth Islands SPA; According to SiteLink, kittiwake on Forth Islands SPA was "Unfavourable Declining" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990) lists 8,400 pairs of kittiwakes for this site. There were 3,661 pairs in 2018-2019 (JNCC 2020).

St Abb's Head to Fast Castle SPA; According to SiteLink, kittiwake on St Abb's Head to Fast Castle SPA was "Unfavourable Declining" when most recently assessed (on 14/06/2014). The citation for St Abb's Head to Fast Castle SPA (11/08/1997) lists 21,170 pairs of kittiwakes for this site. There were 5,000 pairs in 2019 (JNCC 2020).

Farne Islands SPA; The citation for Farne Islands SPA (25/04/1990) lists 8,241 pairs of kittiwakes for this site in 2010-2014. There were 4,402 pairs in 2019 (JNCC 2020).

Flamborough and Filey Coast SPA; The citation for Flamborough and Filey Coast SPA (compiled August 2018) lists 44,520 pairs of kittiwakes for this site (2008-2011). There were 51,535 pairs in 2017 (Lloyd et al. 2019).

North Rona and Sula Sgeir SPA; According to SiteLink, kittiwake on North Rona and Sula Sgeir SPA was "Unfavourable Declining" when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001) lists 5,000 pairs of kittiwakes for this site. There were 1,253 pairs in 2012 (JNCC 2020).

Cape Wrath SPA; According to SiteLink, kittiwake on Cape Wrath SPA was “Unfavourable Declining” when most recently assessed (on 04/06/2017). The citation for Cape Wrath SPA (15/03/1996) lists 9,700 pairs of kittiwakes for this site. There were 3,622 pairs in 2017 (JNCC 2020).

Handa SPA; According to SiteLink, kittiwake on Handa SPA was “Unfavourable Declining” when most recently assessed (on 10/07/2013). The citation for Handa SPA (25/04/1990) lists 10,732 pairs of kittiwakes for this site. There were 3,749 pairs in 2018 (JNCC 2020).

Flannan Isles SPA; According to SiteLink, kittiwake on Flannan Isles SPA was “Unfavourable Declining” when most recently assessed (on 03/06/2013). The citation for Flannan Isles SPA (31/08/1992) lists 2,780 pairs of kittiwakes for this site. JNCC (2020) lists no counts for this site since 1998, when there were 1,052 pairs.

St Kilda SPA; According to SiteLink, kittiwake on St Kilda SPA was “Unfavourable Declining” when most recently assessed (on 08/06/2016). The citation for St Kilda SPA (31/08/1992) lists 7,830 pairs of kittiwakes for this site. There were 420 pairs in 2015 (JNCC 2020).

The Shiant Isles SPA; According to SiteLink, kittiwake on The Shiant Isles SPA was “Unfavourable No Change” when most recently assessed (on 18/06/2015). The citation for The Shiant Isles SPA (31/08/1992) lists 1,800 pairs of kittiwakes for this site. There were 1,075 pairs in 2015 (JNCC 2020).

Mingulay and Berneray SPA; According to SiteLink, kittiwake on Mingulay and Berneray SPA was “Unfavourable Recovering” when most recently assessed (on 04/06/2014). The citation for Mingulay and Berneray SPA (16/12/1994) lists 8,600 pairs of kittiwakes for this site. There were 2,878 pairs in 2014 (JNCC 2020).

Canna and Sanday SPA; According to SiteLink, kittiwake on Canna and Sanday SPA was “Favourable Maintained” when most recently assessed (on 17/02/2014). The citation for Canna and Sanday SPA (20/02/1998) lists 930 pairs of kittiwakes for this site. There were 1,457 pairs in 2019 (JNCC 2020).

Rum SPA; According to SiteLink, kittiwake on Rum SPA was “Unfavourable No Change” when most recently assessed (on 10/06/2015). The citation for Rum SPA (31/08/1992) lists 1,500 pairs of kittiwakes for this site. JNCC (2020) lists no complete counts for Rum since 2000, when there were 788 pairs.

North Colonsay and Western Cliffs SPA; According to SiteLink, kittiwake on North Colonsay and Western Cliffs SPA was “Favourable Maintained” when most recently assessed (on 12/06/2014). The citation for North Colonsay and Western Cliffs SPA (24/03/1997) lists 4,512 pairs of kittiwakes for this site. There were 3,380 pairs in 2016 (JNCC 2020).

Ailsa Craig SPA; According to SiteLink, kittiwake on Ailsa Craig SPA was “Unfavourable Declining” when most recently assessed (on 02/06/2017). The citation for Ailsa Craig SPA (25/04/1990) lists 3,100 pairs of kittiwakes for this site. There were 300 pairs in 2019 (JNCC 2020).

Rathlin Island; The citation for Rathlin Island SPA (March 1999) lists 6,822 pairs of kittiwakes for this site in 1985. There were 7,922 pairs in 2011 (JNCC 2020).

Skomer, Skokholm and seas off Pembrokeshire SPA; The updated citation for Skomer, Skokholm and seas off Pembrokeshire SPA lists 4,472 pairs of kittiwakes as a component of the seabird assemblage for this site in 2001 (NRW 2013). There were 1,236 pairs on Skomer in 2018 (JNCC 2020).

8.14 Sandwich tern

Loch of Strathbeg SPA; According to SiteLink, Sandwich tern on Loch of Strathbeg SPA was "Unfavourable No Change" when most recently assessed (on 31/07/2013). The citation for Loch of Strathbeg SPA (27/11/1995) lists 280 pairs of Sandwich terns for this site (1985-1990). Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 530 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs there in any year 2000-2015 (JNCC 2020).

Ythan Estuary, Sands of Forvie and Meikle Loch SPA; According to SiteLink, Sandwich tern on Ythan Estuary, Sands of Forvie and Meikle Loch SPA was "Favourable Maintained" when most recently assessed (on 01/08/2012). The citation for Ythan Estuary, Sands of Forvie and Meikle Loch SPA (30/03/1998, updated 03/12/2020) lists up to 1,125 pairs of Sandwich terns for this site (1989-1991). There were 849 pairs in 2016, 949 pairs in 2017, 852 pairs in 2018, and 1,010 pairs in 2019 (JNCC 2020).

Forth Islands SPA; According to SiteLink, Sandwich tern on Forth Islands SPA was "Unfavourable Declining" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990) lists 440 pairs of Sandwich terns for this site. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 22 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs in 2018 (JNCC 2020). However, 21 pairs nested nested on the Isle of May in purpose-built nest boxes in 2016, 4 pairs in 2017 and 10 pairs in 2019, the first to breed on this island since 2008 (Steele and Outram 2020).

Farne Islands SPA; The citation (updated 2017) lists 862 pairs (2010-2014). Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 2,070 pairs in the 1990s and 544 pairs in the 2000s. There were 750 pairs in 2015, 629 pairs in 2016, 556 pairs in 2017, 424 pairs in 2018, 417 pairs in 2019 (JNCC 2020).

Coquet Island SPA; At classification in 1985 there were 1,500 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 1,590 pairs in the 1990s and 1,717 pairs in the 2000s. There were 1,349 pairs in 2016, 1,573 pairs in 2017, 1,415 pairs in 2018, and 1,652 pairs in 2019 (JNCC 2020).

North Norfolk Coast SPA; The Natura standard data form (compiled January 1989, revised December 2015) lists 3,700 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 3,457 pairs in the 1990s and 2,980 pairs in the 2000s. At Blakeney Point there were 1,113 pairs in 2015, 451 pairs in 2016, 3 pairs in 2017, 165 pairs in 2018, 788 pairs in 2019, 2,425 pairs in 2020 (JNCC 2020). At Scolt Head Island there were 3,550 pairs in 2015, 3,365 pairs in 2016, 4,665 pairs in 2017, 4,685 pairs in 2018 (JNCC 2020).

Alde-Ore Estuary SPA; The citation for Alde-Ore Estuary SPA lists Sandwich tern as a breeding species for this site without giving numbers. The JNCC standard data form lists 170 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 169 pairs in the 1990s and 2 pairs in the 2000s. JNCC (2020) lists numbers breeding at Havergate Island as 3 in 2005, 0 in 2006, 0 in 2007, 0 in 2008, 2 in 2009, and 0 in 2018.

Foulness SPA; The Natura standard data form (compiled October 1996, revised December 2015) lists 320 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 320 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs in all years 2010 to 2019 (JNCC 2020).

Chichester and Langstone Harbours SPA; The Natura standard data form (compiled October 1987, revised December 2015) lists 31 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 158 pairs in the 1990s and 183 pairs in the 2000s. However, there were 0 pairs in 2015, 2016, 2017, 2018 or 2019 (JNCC 2020).

Solent and Southampton Water SPA; The Natura standard data form (compiled October 1998, revised December 2015) lists 231 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 231 pairs in the 1990s and 215 pairs in the 2000s. There were 0 pairs in 2015, 2016, 2017, 2018 and 2019 at North Solent NNR but 90 pairs in 2018 at Cockleshell (JNCC 2020).

Carlingford Lough; The Natura standard data form (compiled March 1998, revised December 2015) lists 717 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 575 pairs in the 1990s and 0 pairs in the 2000s. There were 250 pairs in 2015, 7 pairs in 2016, 71 pairs in 2017, 13 pairs in 2018, 24 pairs in 2019 (JNCC 2020).

Larne Lough; The Natura standard data form (compiled March 1997, revised September 2018) lists 189 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 165 pairs in the 1990s and 324 pairs in the 2000s. There were 694 pairs in 2015, 1,229 pairs in 2016, 1,141 pairs in 2017, 732 pairs in 2018, 1,010 pairs in 2019 (JNCC 2020).

Strangford Lough; The Natura standard data form (compiled March 1998, revised September 2018) lists 593 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 593 pairs in the 1990s and 978 pairs in the 2000s. There were 581 pairs in 2015, 337 pairs in 2016, 775 pairs in 2017, 776 pairs in 2018, 434 pairs in 2019, 252 pairs in 2020 (JNCC 2020).

Morecambe Bay and Duddon Estuary SPA; The Natura standard data form (compiled February 2017) lists 1,608 breeding individuals, so presumably equates to 804 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 500 pairs in the 1990s and 1 pair in the 2000s There were 20 pairs in 2014, 0 pairs in 2015, 22 pairs in 2016, 550 pairs in 2017, 1,950 pairs in 2018, 805 pairs in 2019 (JNCC 2020).

Anglesey terns SPA (also known as Ynys Feurig, Cemlyn Bay and The Skerries SPA); The Natura standard data form (compiled June 1992, revised January 2017) lists 460 pairs. Stroud et al. (2016) report breeding numbers of Sandwich terns in this SPA as 460 pairs in the 1990s and 1,700 pairs in the 2000s. There were 0 pairs in all years from 2000 to 2009 apart from 1 pair in 2006, and there were 0 pairs in 2019 (JNCC 2020).

8.15 Common tern

Cromarty Firth SPA; According to SiteLink, common tern on Cromarty Firth SPA was “Unfavourable Declining” when most recently assessed (on 10/06/2000). The citation for Cromarty Firth SPA (22/03/1999) lists 294 pairs of common terns for this site (1989-1993). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 294 pairs in the 1990s and 0 pairs in the 2000s. There were 7 pairs there in 2011 but 0 pairs there in each year 2012-2015 (JNCC 2020).

Inner Moray Firth SPA; According to SiteLink, common tern on Inner Moray Firth SPA was "Unfavourable No Change" when most recently assessed (on 30/06/2000). The citation for Inner Moray Firth SPA (amended 20/04/2018) lists 310 pairs of common terns for this site ("at the time of classification" which was 1999). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 310 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs there each year from 2010 to 2015 (JNCC 2020).

Ythan Estuary, Sands of Forvie and Meikle Loch SPA; According to SiteLink, common tern on Ythan Estuary, Sands of Forvie and Meikle Loch SPA was "Unfavourable No Change" when most recently assessed (on 01/08/2012). The citation for Ythan Estuary, Sands of Forvie and Meikle Loch SPA (amended 03/12/2020) lists up to 265 pairs of common terns for this site (1989-1993). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 265 pairs in the 1990s and 4 pairs in the 2000s. There were 97 pairs in 2013, 150 pairs in 2014, 172 pairs in 2015, 157 pairs in 2016, 140 pairs in 2017, 278 pairs in 2018 (JNCC 2020).

Forth Islands SPA; According to SiteLink, common tern on Forth Islands SPA was "Unfavourable Declining" when most recently assessed (on 23/06/2017). The citation for Forth Islands SPA (amended 25/05/2018) lists 334 pairs of common terns for this site (1997-2001). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 800 pairs in the 1990s and 26 pairs in the 2000s. On the Isle of May, there were 15 pairs in 2014, 13 pairs in 2015, 19 pairs in 2016, 29 pairs in 2017, while on Long Craig there were 73 pairs in 2013, 165 pairs in 2017, 78 pairs in 2018, 128 pairs in 2019 (JNCC 2020). The Isle of May held 17 pairs in 2018 and 51 pairs in 2019, the increase in numbers being attributed to provision of purpose-built nest boxes for terns in 2016-2019 (Steel and Outram 2020).

Imperial Dock Lock, Leith SPA; According to SiteLink, common tern on Imperial Dock Lock, Leith SPA was "Favourable Maintained" when most recently assessed (on 23/06/2017). The citation for Imperial Dock Lock, Leith SPA (September 2004) lists 558 pairs of common terns for this site (1997-2001). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 818 pairs in the 2000s. There were 818 pairs in 2013, 931 pairs in 2014, 636 pairs in 2015, 719 pairs in 2016, 985 pairs in 2017, 514 pairs in 2018, 246 pairs in 2019 (JNCC 2020), numbers in 2018 and especially 2019 being reduced by presence of mink as well as predation by crows.

Farne Islands SPA; Citation (1985) lists 183 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 230 pairs in the 1990s and 101 pairs in the 2000s. Farne Islands SPA held 65 to 112 pairs in 2010 to 2020 (JNCC 2020).

Coquet Island SPA; The Natura standard data form (compiled July 1985, revised January 2017) lists 740 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 740 pairs in the 1990s and 1,193 pairs in the 2000s. From 2010 to 2020 there were between 1,041 and 1,667 pairs (JNCC 2020).

The Wash SPA; The Natura standard data form (compiled March 1988, revised December 2015) lists 152 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 152 pairs in the 1990s and 169 pairs in the 2000s. At Snettisham there were 118 pairs in 2015, 174 pairs in 2016, 150 pairs in 2017, 125 pairs in 2018, 114 pairs in 2019 (JNCC 2020).

North Norfolk Coast SPA; The Natura standard data form (compiled January 1989, revised December 2015) lists 460 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 460 pairs in the 1990s and 347 pairs in the 2000s. There were at least 230 pairs in 2018 and 290 pairs in 2020 (JNCC 2020).

Dungeness to Pett Level SPA; The Natura standard data form (compiled August 1999, revised November 2017) lists 188 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 266 pairs in the 1990s and 235 pairs in the 2000s. Rye Harbour held about 100 pairs in most years 2010 to 2020 (JNCC 2020).

Foulness SPA; The Natura standard data form (compiled October 1996, revised December 2015) lists 220 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 220 pairs in the 1990s and 25 pairs in the 2000s.

Breydon Water SPA; The Natura standard data form (compiled March 1996, revised December 2015) lists 155 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 155 pairs in the 1990s and 158 pairs in the 2000s. From 2010 to 2020 there were between 6 and 173 pairs with about 100 pairs in most years (JNCC 2020).

Poole Harbour SPA; The Natura standard data form (compiled March 1999, revised November 2017) lists 178 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 155 pairs in the 1990s and 222 pairs in the 2000s. At Brownsea Island there were 222 pairs in 2011, 171 pairs in 2012, 163 pairs in 2013, 145 pairs in 2014, 200 pairs in 2015 (JNCC 2020).

Solent and Southampton Water SPA; The Natura standard data form (compiled October 1998, revised December 2015) lists 267 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 267 pairs in the 1990s and 280 pairs in the 2000s. There were 94 pairs in 2018 at Pylewell (JNCC 2020).

Glas Eileanan SPA; According to SiteLink, common tern on Glas Eileanan SPA was “Unfavourable Declining” when most recently assessed (on 15/06/2015). The citation for Glas Eileanan SPA (December 1997) lists 530 pairs of common terns for this site (1993-1997). Stroud et al. (2016) report breeding numbers of common terns in this SPA as 530 pairs in the 1990s and 97 pairs in the 2000s. There were 97 pairs there in 2011, 22 pairs in 2012, 98 pairs in 2014, 55 pairs in 2015 (JNCC 2020).

Carlingford Lough; The Natura standard data form (compiled March 1998, revised December 2015) lists 443 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 339 pairs in the 1990s and 69 pairs in the 2000s. There were 220 pairs in 2015, 123 pairs in 2016, 147 pairs in 2017, 70 pairs in 2018, 56 pairs in 2019 (JNCC 2020).

Larne Lough; The Natura standard data form (compiled March 1997, revised September 2018) lists 247 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 180 pairs in the 1990s and 317 pairs in the 2000s. There were between 228 and 380 pairs each year from 2010 to 2019 (JNCC 2020).

Lough Neagh and Lough Beg; The Natura standard data form (compiled April 1996, revised September 2018) lists 185 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 185 pairs in the 1990s and 93 pairs in the 2000s. There were around 100-200 pairs most years from 2010 to 2019 (JNCC 2020).

Strangford Lough; The Natura standard data form (compiled March 1998, revised September 2018) lists 603 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 603 pairs in the 1990s and 726 pairs in the 2000s. There were 80 to 730 pairs in 2010 to 2020, with about 200-300 pairs in most years (JNCC 2020).

Ribble and Alt Estuaries SPA; The citation (November 2002) lists 182 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 182 pairs in the 1990s and 111 pairs in the 2000s. There were 0 pairs in 2016, 6 pairs in 2018 (JNCC 2020).

Mersey Narrows and North Wirral Foreshore SPA; The Natura standard data form (compiled July 2013, revised December 2015) lists 177 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 177 pairs in the 2000s. Seaforth Nature Reserve held 156 to 202 pairs in 2010 to 2019 (JNCC 2020).

The Dee Estuary SPA; The Natura standard data form (compiled July 1985, revised December 2015) lists 392 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 277 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs in 2019 (JNCC 2020).

Anglesey terns SPA (also known as Ynys Feurig, Cemlyn Bay and The Skerries SPA); The Natura standard data form (compiled June 1992, revised January 2017) lists 189 pairs. Stroud et al. (2016) report breeding numbers of common terns in this SPA as 189 pairs in the 1990s and 592 pairs in the 2000s. There were 183 pairs in 2010, 178 pairs in 2011, 191 pairs in 2019 (JNCC 2020).

8.16 Arctic tern

Fetlar SPA; According to SiteLink, Arctic tern on Fetlar SPA was “Unfavourable Declining” when most recently assessed (on 21/06/2017). The citation for Fetlar SPA (29/03/1994, amended September 2009) lists 1,065 pairs of Arctic terns for this site at the time of classification. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 520 pairs in the 1990s and 2 pairs in the 2000s. There were about 200 adults present in May-June 2018, but only 16 nests noted (JNCC 2020).

Foula SPA; According to SiteLink, Arctic tern on Foula SPA was “Unfavourable Declining” when most recently assessed (on 22/06/2016). The citation for Foula SPA (27/11/1995, amended September 2009) lists up to 1,500 pairs of Arctic terns for this site at the time of classification. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,100 pairs in the 1990s and 95 pairs in the 2000s. There were about 19 nesting pairs in 2018 although about 200 adults were present, and 150 nesting pairs there in 2020 (Sheila Gear annual reports to SOTEAG).

Papa Stour SPA; According to SiteLink, Arctic tern on Papa Stour SPA was “Unfavourable No Change” when most recently assessed (on 19/06/2015). The citation for Papa Stour SPA (27/03/2000, amended 26/04/2018) lists 850 pairs of Arctic terns for this site (1991, 1994 and 1995). Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,000 pairs in the 1990s and 1,172 pairs in the 2000s. There were 94 adults present in 2018 but apparently no nests (JNCC 2020).

Mousa SPA; According to SiteLink, Arctic tern on Mousa SPA was “Unfavourable Declining” when most recently assessed (on 01/06/2015). The citation for Mousa SPA (October 1994) lists up to 1,000 pairs of Arctic terns for this site. Stroud et al. (2016) report breeding numbers of Arctic terns in this

SPA as 767 pairs in the 1990s and 0 pairs in the 2000s. There were 287 pairs in 2014, 56 pairs in 2016, 20 pairs in 2017, 109 pairs in 2019 (JNCC 2020).

Sumburgh Head SPA; According to SiteLink, Arctic tern on Sumburgh Head SPA was "Unfavourable No Change" when most recently assessed (on 11/06/2018). The citation for Sumburgh Head SPA (27/03/1996, revised September 2009) lists 700 pairs of Arctic terns for this site. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 700 pairs in the 1990s and 203 pairs in the 2000s. There were counts of up to 600 adults present in June 2018, 195 adults and 42 nesting pairs in 2019 (JNCC 2020).

Fair Isle SPA; According to SiteLink, Arctic tern on Fair Isle SPA was "Unfavourable Declining" when most recently assessed (on 01/06/2016). The citation for Fair Isle SPA (16/12/1994, amended September 2009) lists 1,100 pairs of Arctic terns for this site at the time of classification. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,120 pairs in the 1990s and 9 pairs in the 2000s. There were 30 pairs in 2015, 118 pairs in 2016, 322 pairs in 2017, 190 pairs in 2018, 286 pairs in 2019 (JNCC 2020).

Papa Westray SPA; According to SiteLink, Arctic tern on Papa Westray SPA was "Unfavourable No Change" when most recently assessed (on 01/08/2017). The citation for Papa Westray SPA (27/03/1996, amended 27/04/2018) lists 1,700 pairs of Arctic terns for this site in 1994. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,950 pairs in the 1990s and 176 pairs in the 2000s.

West Westray SPA; According to SiteLink, Arctic tern on West Westray SPA was "Unfavourable No Change" when most recently assessed (on 20/07/2017). The citation for West Westray SPA (16/08/1996, revised September 2009) lists 1,140 pairs of Arctic terns for this site. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,200 pairs in the 1990s and 1,067 pairs in the 2000s. The only count provided by JNCC (2020) for years since 2000 was of 1,048 individuals at Noup Head in 2009 (date not specified).

Rousay SPA; According to SiteLink, Arctic tern on Rousay SPA was "Unfavourable No Change" when most recently assessed (on 09/06/2018). The citation for Rousay SPA (02/02/2000, revised September 2009) lists 790 pairs of Arctic terns for this site (1991-1995). Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,000 pairs in the 1990s and 707 pairs in the 2000s. There were 9 pairs in 2018 (JNCC 2020).

Auskerry SPA; According to SiteLink, Arctic tern on Auskerry SPA was "Unfavourable Declining" when most recently assessed (on 21/06/2018). The citation for Auskerry SPA (August 1996) lists 780 pairs of Arctic terns for this site (1992-1995). Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 780 pairs in the 1990s and 0 pairs in the 2000s. There were 0 pairs in 2011, 8 pairs in 2016 (JNCC 2020). In June-July there were some counts of large numbers of Arctic terns at Auskerry (for example 1,500 birds on 15/07/2014) but no proof of breeding (JNCC 2020).

Pentland Firth Islands SPA; According to SiteLink, Arctic tern on Pentland Firth Islands SPA was "Unfavourable No Change" when most recently assessed (on 30/06/2018). The citation for Pentland Firth Islands SPA (June 1998) lists 1,000 pairs of Arctic terns for this site in 1995. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,200 pairs in the 1990s and 667 pairs in the 2000s. There were about 200 adults present in June 2011 and June 2016, and 789 adults in June 2018 but no counts of nests (JNCC 2020).

Forth Islands SPA; According to SiteLink, Arctic tern on Forth Islands SPA was "Favourable Declining" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990, amended 25/05/2018) lists 540 pairs of Arctic terns for this site (1992-1996). Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 520 pairs in the 1990s and 2 pairs in the 2000s. On the Isle of May there were 484 pairs in 2015, 527 pairs in 2016, 832 pairs in 2017 (JNCC 2020) and 490 pairs in 2019 with 31% of those using nest boxes (Steel and Outram 2020).

Farne Islands SPA; The citation and standard data form (compiled July 1985, revised September 2018) list 2,003 pairs in 2010-2014. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 2,840 pairs in the 1990s and 1,830 pairs in the 2000s. There were between 1,416 and 2,199 pairs in 2010 to 2019 (JNCC 2020).

Coquet Island SPA; The Natura standard data form (compiled July 1985, revised January 2017) lists 700 pairs. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 700 pairs in the 1990s and 1,140 pairs in the 2000s. There were between 1,046 and 1,579 pairs in 2010 to 2019 (JNCC 2020).

Copeland Islands; The Natura standard data form (compiled December 2009, revised December 2015) lists 566 pairs. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,025 pairs in the 2000s. There were 150 pairs in 2018 and in 2019 (JNCC 2020).

Outer Ards; The Natura standard data form (compiled December 2012, revised December 2015) lists 207 pairs. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 207 pairs in the 1990s and 775 pairs in the 2000s. There were 248 pairs at Green Island, 129 pairs at Cockle Island and 140 pairs at Bird Island in 2017, and 341 pairs at Cockle Island in 2018 (JNCC 2020).

Strangford Lough; The Natura standard data form (compiled March 1998, revised September 2018) lists 210 pairs. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 210 pairs in the 1990s and 229 pairs in the 2000s. There were between 55 and 373 pairs in 2010 to 2020, with around 200 pairs in most years (JNCC 2020).

Anglesey terns SPA (also known as Ynys Feurig, Cemlyn Bay and The Skerries SPA); The Natura standard data form (compiled June 1992, revised January 2017) lists 1,290 pairs. Stroud et al. (2016) report breeding numbers of Arctic terns in this SPA as 1,290 pairs in the 1990s and 3,620 pairs in the 2000s. There were 625 pairs in 2010, 550 pairs in 2011, and 382 pairs in 2019 (JNCC 2020).

8.17 Common guillemot

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, common guillemot on Hermaness, Saxa Vord and Valla Field SPA was "Unfavourable Declining" when most recently assessed (on 11/06/2017). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001, revised September 2009) lists 25,000 individuals (≈ 16,750 pairs) of common guillemots for this site in surveys in 1996 and 1999. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 11,363 pairs in the 1990s and 10,445 pairs in the 2000s. There were 6,109 individuals (≈ 4,091 pairs) in 2016 (JNCC 2020).

Noss SPA; According to SiteLink, common guillemot on Noss SPA was "Unfavourable No Change" when most recently assessed (on 23/06/2015). The citation for Noss SPA (16/08/1996, revised September 2009) lists 38,970 individuals (≈ 26,110 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 30,619 pairs in the 1990s and 14,784 pairs in the 2000s. There were 24,456 individuals (≈ 16,386 pairs) in 2015 (JNCC 2020).

Foula SPA; According to SiteLink, common guillemot on Foula SPA was "Unfavourable Declining" when most recently assessed (on 24/06/2015). The citation for Foula SPA (27/11/1995, updated September 2009) lists 37,500 individuals (≈ 25,125 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 25,125 pairs in the 1990s and 16,615 pairs in the 2000s. There were 24,799 individuals (≈ 16,615 pairs) in 2007 (JNCC 2020).

Sumburgh Head SPA; According to SiteLink, common guillemot on Sumburgh Head SPA was "Unfavourable Declining" when most recently assessed (on 14/06/2017). The citation for Sumburgh Head SPA (27/03/1996, revised September 2009) lists 16,000 individuals (≈ 10,720 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 10,752 pairs in the 1990s and 11,103 pairs in the 2000s. There were 6,957 individuals in 2015, 16,208 individuals (≈ 10,859 pairs) reported to be present in 2017 (JNCC 2020), but only 7,749 individuals in 2018 (JNCC 2020).

Fair Isle SPA; According to SiteLink, common guillemot on Fair Isle SPA was "Unfavourable Declining" when most recently assessed (on 01/06/2016). The citation for Fair Isle SPA (16/12/1994, revised September 2009) lists 32,300 individuals (≈ 21,641 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 25,165 pairs in the 1990s and 13,066 pairs in the 2000s. There were 20,924 individuals (≈ 14,019 pairs) in 2015 (JNCC 2020).

West Westray SPA; According to SiteLink, common guillemot on West Westray SPA was "Unfavourable Declining" when most recently assessed (on 08/06/2017). The citation for West Westray SPA (16/08/1996, revised September 2009) lists 42,150 individuals (≈ 28,240 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 28,274 pairs in the 1990s and 33,911 pairs in the 2000s. There were 28,697 individuals (≈ 19,227 pairs) in 2017 (JNCC 2020).

Calf of Eday SPA; According to SiteLink, common guillemot on Calf of Eday SPA was "Unfavourable Declining" when most recently assessed (on 08/06/2016). The citation for Calf of Eday SPA (29/06/1998, revised September 2009) lists 12,645 individuals (≈ 8,472 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 8,241 pairs in the 1990s and 1,715 pairs in the 2000s. There were 5,524 individuals (≈ 3,701 pairs) in 2018 (JNCC 2020).

Rousay SPA; According to SiteLink, common guillemot on Rousay SPA was "Unfavourable Declining" when most recently assessed (on 24/06/2016). The citation for Rousay SPA (02/02/2000, revised September 2009) lists 10,600 individuals (≈ 7,102 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 7,102 pairs in the 1990s and 4,120 pairs in the 2000s. There were about 6,500 individuals (≈ 4,355 pairs) in 2016 (JNCC 2020).

Hoy SPA; According to SiteLink, common guillemot on Hoy SPA was "Unfavourable No Change" when most recently assessed (on 10/06/2017). The citation for Hoy SPA (07/12/2000, revised September 2009) lists 13,400 pairs of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 13,400 pairs in the 1990s and 6,043 pairs in the 2000s. There were at least 12,198 individuals (≈ 8,173 pairs) in 2016-2017 (JNCC 2020).

Marwick Head SPA; According to SiteLink, common guillemot on Marwick Head SPA was "Unfavourable Declining" when most recently assessed (on 22/06/2017). The citation for Marwick Head SPA (16/12/1994, revised September 2009) lists 37,700 individuals (≈ 25,259 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 24,388 pairs in the 1990s and 11,267 pairs in the 2000s. There were 11,985 individuals (≈ 8,030 pairs) in 2018 (JNCC 2020).

Copinsay SPA; According to SiteLink, common guillemot on Copinsay SPA was "Unfavourable No Change" when most recently assessed (on 11/06/2015). The citation for Copinsay SPA (29/03/1994, revised September 2009) lists 29,450 individuals (≈ 19,732 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 13,333 pairs in the 1990s and 9,166 pairs in the 2000s. There were 8,369 individuals (≈ 5,607 pairs) in 2012 and 18,454 individuals (≈ 12,364 pairs) in 2015 (JNCC 2020).

North Caithness Cliffs SPA; According to SiteLink, common guillemot on North Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 13/06/2016). The citation for North Caithness Cliffs SPA (16/08/1996, amended 26/04/2018) lists 38,300 individuals (≈ 25,661 pairs) of common guillemots for this site in 1985-1987. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 26,994 pairs in the 1990s and 47,003 pairs in the 2000s. There were at least 25,000 individuals (≈ 16,750 pairs) in 2015-2016 (JNCC 2020).

East Caithness Cliffs SPA; According to SiteLink, common guillemot on East Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 30/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996, amended March 2017) lists 106,700 individuals (≈ 71,489 pairs) of common guillemots for this site in 1985-1987. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 71,509 pairs in the 1990s and 106,520 pairs in the 2000s. There were 149,228 individuals (≈ 99,983 pairs) in 2015 (Swann 2016).

Troup, Pennan and Lion's Heads SPA; According to SiteLink, common guillemot on Troup, Pennan and Lion's Heads SPA was "Unfavourable Declining" when most recently assessed (on 03/07/2007). The citation for Troup, Pennan and Lion's Heads SPA (14/03/1997, revised September 2009) lists 44,600 individuals (≈ 29,882 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 29,902 pairs in the 1990s and 10,938 pairs in the 2000s. There were at least 23,800 individuals (≈ 15,947 pairs) in 2017 (JNCC 2020).

Fowlsheugh SPA; According to SiteLink, common guillemot on Fowlsheugh SPA was "Favourable Maintained" when most recently assessed (on 11/06/1999). The citation for Fowlsheugh SPA (31/08/1992, revised September 2009) lists 56,450 individuals (≈ 37,822 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 40,140 pairs in the 1990s and 33,873 pairs in the 2000s. There were 69,828 individuals (≈ 46,785 pairs) in 2018 (JNCC 2020).

Buchan Ness to Collieston Coast SPA; According to SiteLink, common guillemot on Buchan Ness to Collieston Coast SPA was "Favourable Maintained" when most recently assessed (on 16/06/2017). The citation for Buchan Ness to Collieston Coast SPA (30/03/1998, revised September 2009) lists 8,640 pairs of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 8,640 pairs in the 1990s and 12,928 pairs in the 2000s. There were 29,187 individuals (≈ 19,555 pairs) in 2019 (JNCC 2020).

Forth Islands SPA; According to SiteLink, common guillemot on Forth Islands SPA was "Favourable Maintained" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990, amended 25/05/2018) lists 16,000 pairs of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 22,452 pairs in the 1990s and 16,804 pairs in the 2000s. There were 28,751 individuals (≈ 19,263 pairs) in 2017, 25,956 individuals (≈ 17,391 pairs) in 2018 (JNCC 2020).

St Abb's Head to Fast Castle SPA; According to SiteLink, common guillemot on St Abb's Head to Fast Castle SPA was "Favourable Maintained" when most recently assessed (on 22/06/2013). The citation for St Abb's Head to Fast Castle SPA (11/08/1997, revised September 2009) lists 31,750 individuals (≈ 21,273 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 20,971 pairs in the 1990s and 29,308 pairs in the 2000s. There were about 43,000 individuals (≈ 29,000 pairs) in 2018 (JNCC 2020).

Farne Islands SPA; The Natura standard data form (compiled July 1985, revised September 2018) lists 32,875 pairs. The citation lists 65,751 individuals in 2010-2014. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 23,499 pairs in the 1990s and 32,145 pairs in the 2000s. There were 64,042 individuals (≈ 42,908 pairs) in 2019 (JNCC 2020).

Flamborough and Filey Coast SPA; The citation (compiled August 2018 Version 2.0) lists 41,607 pairs based on a count of 62,100 individuals in 2008-2011. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 16,150 pairs in the 1990s and 39,641 pairs in the 2000s. There were 84,647 individuals (≈ 56,713 pairs) in 2017 (JNCC 2020).

Cape Wrath SPA; According to SiteLink, common guillemot on Cape Wrath SPA was "Favourable Maintained" when most recently assessed (on 04/06/2017). The citation for Cape Wrath SPA (15/03/1996, revised September 2009) lists 13,700 individuals (≈ 9,179 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 9,159 pairs in the 1990s and 27,359 pairs in the 2000s. There were 38,109 individuals (≈ 25,533 pairs) in 2017 (JNCC 2020).

North Rona and Sula Sgeir SPA; According to SiteLink, common guillemot on North Rona and Sula Sgeir SPA was "Unfavourable Declining" when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001, revised September 2009) lists 43,200 individuals (≈ 28,944 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 28,944 pairs in the 1990s and 21,021 pairs in the 2000s. There were 4,961 (≈ 3,324 pairs) individuals in 2012 (JNCC 2020).

Sule Skerry and Sule Stack SPA; According to SiteLink, common guillemot on Sule Skerry and Sule Stack SPA was "Favourable Maintained" when most recently assessed (on 10/07/2015). The citation for Sule Skerry and Sule Stack SPA (29/03/1994, revised September 2009) lists 6,298 (unit not specified in the citation document but assumed here to be individuals) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 6,298 pairs in the 1990s and 7,633 pairs in the 2000s. At Sule Skerry there were estimated to be 10,068 individuals (≈ 6,746 pairs) in 2018 using mark-recapture of adults (JNCC 2020).

Handa SPA; According to SiteLink, common guillemot on Handa SPA was "Unfavourable No Change" when most recently assessed (on 01/04/2012). The citation for Handa SPA (25/04/1990, revised September 2009) lists 98,686 individuals (≈ 66,120 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 76,105 pairs in the 1990s and 37,993 pairs in the 2000s. There were 56,706 individuals (≈ 37,993 pairs) in 2011, 54,664 individuals (≈ 36,625 pairs) in 2016 (JNCC 2020).

St Kilda SPA; According to SiteLink, common guillemot on St Kilda SPA was "Unfavourable Declining" when most recently assessed (on 11/06/2016). The citation for St Kilda SPA (31/08/1992, revised September 2009) lists 22,700 individuals (≈ 15,209 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 15,209 pairs in the 1990s and 15,673 pairs in the 2000s. There were about 10,300 individuals (≈ 6,900 pairs) in 2015-2016 (JNCC 2020).

Mingulay and Berneray SPA; According to SiteLink, common guillemot on Mingulay and Berneray SPA was "Favourable Recovered" when most recently assessed (on 21/06/2014). The citation for Mingulay and Berneray SPA (16/12/1994, revised September 2009) lists 30,900 individuals (≈ 20,703 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 20,703 pairs in the 1990s and 13,527 pairs in the 2000s. In 2014 there were 9,949 individuals on Berneray and 12,316 individuals on Mingulay; total 22,265 individuals (≈ 14,918 pairs) (JNCC 2020). In 2017 there were 19,384 individuals on Mingulay but numbers were not counted on Berneray.

Flannan Isles SPA; According to SiteLink, common guillemot on Flannan Isles SPA was "Unfavourable Declining" when most recently assessed (on 03/06/2013). The citation for Flannan Isles SPA (31/08/1992, amended September 2009) lists 21,930 individuals (≈ 14,931 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 14,693 pairs in the 1990s and 9,807 pairs in the 2000s. No counts since 2000 are available in JNCC (2020).

Canna and Sanday SPA; According to SiteLink, common guillemot on Canna and Sanday SPA was "Unfavourable Declining" when most recently assessed (on 17/02/2014). The citation for Canna and Sanday SPA (20/02/1998, revised September 2009) lists 5,800 individuals (≈ 3,886 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 3,858 pairs in the 1990s and 3,913 pairs in the 2000s. There were 2,850 individuals (≈ 1,910 pairs) in 2018 (JNCC 2020).

Rum SPA; According to SiteLink, common guillemot on Rum SPA was "Unfavourable No Change" when most recently assessed (on 10/06/2015). The citation for Rum SPA (31/08/1982, revised September 2009) lists 4,000 individuals (≈ 2,680 pairs) of common guillemots for this site at classification. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 2,680 pairs in the 1990s and 1,644 pairs in the 2000s. There were 2,454 individuals (≈ 1,644 pairs) in 2020 but no count published since then (JNCC 2020).

The Shiant Isles SPA; According to SiteLink, common guillemot on The Shiant Isles SPA was "Unfavourable No Change" when most recently assessed (on 18/06/2015). The citation for The Shiant Isles SPA (31/08/1992, revised September 2009) lists 18,380 individuals (12,315 pairs) of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 18,380 pairs in the 1990s and 5,148 pairs in the 2000s. There were 9,054 individuals (≈ 6,066 pairs) in 2015 (JNCC 2020).

North Colonsay and Western Cliffs SPA; According to SiteLink, common guillemot on North Colonsay and Western Cliffs SPA was "Favourable Maintained" when most recently assessed (on 12/06/2014). The citation for North Colonsay and Western Cliffs SPA (24/03/1997, revised September 2009) lists 6,656 pairs of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 6,656 pairs in the 1990s and 13,170 pairs in the 2000s. There were 18,724 individuals (≈ 12,545 pairs) in 2018 (JNCC 2020).

Ailsa Craig SPA; According to SiteLink, common guillemot on Ailsa Craig SPA was "Favourable Maintained" when most recently assessed (on 02/06/2017). The citation for Ailsa Craig SPA (25/04/1990, revised September 2009) lists 3,350 pairs of common guillemots for this site. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 3,350 pairs in the 1990s and 7,818 pairs in the 2000s. There were 8,940 individuals (≈ 5,990 pairs) in 2017, 7,040 individuals (≈ 4,717 pairs) in 2018, 6,180 individuals (≈ 4,141 pairs) in 2019 (JNCC 2020).

Rathlin Island; The citation, and the standard data form (complied February 1999, revised September 2018) lists 41,887 individuals in 1985. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 28,064 pairs in the 1990s and 87,398 pairs in the 2000s. There were 130,445 individuals (≈ 87,398 pairs) in 2011 (JNCC 2020).

Skomer, Skokholm and seas off Pembrokeshire SPA; The Natura standard data form (compiled August 1982, revised January 2017) lists common guillemot as a component of the seabird assemblage, with 15,262 individuals. Stroud et al. (2016) report breeding numbers of common guillemots in this SPA as 7,067 pairs in the 1990s and 16,375 pairs in the 2000s. In 2017 there were 13,547 individuals at Skomer, 4,038 individuals at Skokholm, and 310 individuals at Middleholm so 17,895 individuals in the SPA population estimate (≈ 11,990 pairs) (JNCC 2020).

8.18 Razorbill

Foula SPA; According to SiteLink, razorbill on Foula SPA was "Unfavourable Declining" when most recently assessed (on 24/06/2015). The citation for Foula SPA (27/11/1995, revised September 2009) lists 6,200 individuals (≈ 4,154 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 4,154 pairs in the 1990s and 2,814 pairs in the 2000s. There were 559 individuals (≈ 375 pairs) in 2007 (JNCC 2020).

Fair Isle SPA; According to SiteLink, razorbill on Fair Isle SPA was "Unfavourable Declining" when most recently assessed (on 01/06/2015). The citation for Fair Isle SPA (16/12/1994, revised September 2009) lists 3,400 individuals (≈ 2,278 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,044 pairs in the 1990s and 915 pairs in the 2000s. There were 1,930 individuals (≈ 1,293 pairs) in 2015 (JNCC 2020).

West Westray SPA; According to SiteLink, razorbill on West Westray SPA was "Favourable Recovered" when most recently assessed (on 08/06/2017). The citation for West Westray SPA

(16/08/1996, revised September 2009) lists 1,946 individuals (≈ 1,304 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 1,307 pairs in the 1990s and 545 pairs in the 2000s. There were 2,159 individuals (≈ 1,447 pairs) in 2017 (JNCC 2020).

North Caithness Cliffs SPA; According to SiteLink, razorbill on North Caithness Cliffs SPA was "Favourable Recovered" when most recently assessed (on 13/06/2016). The citation for North Caithness Cliffs SPA (16/08/1996, amended 26/04/2018) lists 4,000 individuals (≈ 2,680 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,212 pairs in the 1990s and 1,652 pairs in the 2000s. There were at least 2,800 individuals (≈ 1,876 pairs) in 2015-2016 (JNCC 2020).

East Caithness Cliffs SPA; According to SiteLink, razorbill on East Caithness Cliffs SPA was "Favourable Maintained" when most recently assessed (on 30/06/2015). The citation for East Caithness Cliffs SPA (27/03/1996, amended March 2017) lists 15,800 individuals (≈ 10,586 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 9,259 pairs in the 1990s and 11,972 pairs in the 2000s. There were 30,042 individuals (≈ 20,128 pairs) in 2015 (Swann 2016).

Troup, Pennan and Lion's Heads SPA; According to SiteLink, razorbill on Troup, Pennan and Lion's Heads SPA was "Unfavourable Declining" when most recently assessed (on 03/07/2007). The citation for Troup, Pennan and Lion's Heads SPA (14/03/1997, revised September 2009) lists 4,800 individuals (≈ 3,216 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 3,216 pairs in the 1990s and 2,011 pairs in the 2000s. There were 4,518 individuals (≈ 3,027 pairs) in 2017 (JNCC 2020).

Fowlsheugh SPA; According to SiteLink, razorbill on Fowlsheugh SPA was "Favourable Maintained" when most recently assessed (on 11/06/1999). The citation for Fowlsheugh SPA (31/08/1992, revised September 2009) lists 5,800 individuals (≈ 3,886 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 4,576 pairs in the 1990s and 3,103 pairs in the 2000s. There were 14,063 individuals (≈ 9,422 pairs) in 2018 (JNCC 2020).

Forth Islands SPA; According to SiteLink, razorbill on Forth Islands SPA was "Favourable Maintained" when most recently assessed (on 30/06/2016). The citation for Forth Islands SPA (25/04/1990, amended 25/05/2018) lists 1,400 pairs of razorbills for this site in 1986-1988. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,693 pairs in the 1990s and 2,402 pairs in the 2000s. In 2018 there were 4,867 individuals (≈ 3,261 pairs) on the Isle of May, 176 pairs on Bass Rock, 161 pairs on Fidra, 102 pairs on Craigleith, 76 pairs on The Lamb, so a total of 3,776 pairs for this SPA (JNCC 2020).

St Abb's Head to Fast Castle SPA; According to SiteLink, razorbill on St Abb's Head to Fast Castle SPA was "Favourable Maintained" when most recently assessed (on 22/06/2013). The citation for St Abb's Head to Fast Castle SPA (11/08/1997, revised September 2009) lists 2,180 individuals (≈ 1,461 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 1,407 pairs in the 1990s and 2,115 pairs in the 2000s. There were at least 2,761 individuals (≈ 1,850 pairs) in 2018 (JNCC 2020).

Flamborough and Filey Coast SPA; The citation (compiled August 2018 Version 2.0) lists 10,570 pairs in 2008-2011 based on count of 15,776 individuals. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 5,133 pairs in the 1990s and 10,001 pairs in the 2000s. There were 30,228 individuals (≈ 20,253 pairs) in 2017 (Aitken et al. 2017).

North Rona and Sula Sgeir SPA; According to SiteLink, razorbill on North Rona and Sula Sgeir SPA was “Unfavourable Declining” when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001, revised September 2009) lists 2,300 individuals (≈ 1,541 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 1,541 pairs in the 1990s and 1,089 pairs in the 2000s. There were 513 individuals (≈ 344 pairs) on North Rona in 2012 (JNCC 2020).

Cape Wrath SPA; According to SiteLink, razorbill on Cape Wrath SPA was “Favourable Maintained” when most recently assessed (on 04/06/2017). The citation for Cape Wrath SPA (15/03/1996, revised September 2009) lists 1,800 individuals (≈ 1,206 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 1,206 pairs in the 1990s and 2,005 pairs in the 2000s. There were 3,241 individuals (≈ 2,171 pairs) in 2017 (JNCC 2020).

Handa SPA; According to SiteLink, razorbill on Handa SPA was “Unfavourable Declining” when most recently assessed (on 09/06/2014). The citation for Handa SPA (25/04/1990, revised September 2009) lists 16,394 individuals (≈ 10,984 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 10,432 pairs in the 1990s and 5,165 pairs in the 2000s. There were 8,207 individuals (≈ 5,499 pairs) in 2019 (JNCC 2020).

Flannan Isles SPA; According to SiteLink, razorbill on Flannan Isles SPA was “Unfavourable Recovering” when most recently assessed (on 03/06/2013). The citation for Flannan Isles SPA (31/08/1992, revised September 2009) lists 3,160 individuals (≈ 2,117 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,117 pairs in the 1990s and 1,051 pairs in the 2000s. No counts for this SPA are listed by JNCC (2020) for any years since 1998.

Mingulay and Berneray SPA; According to SiteLink, razorbill on Mingulay and Berneray SPA was “Favourable Recovered” when most recently assessed (on 21/06/2014). The citation for Mingulay and Berneray SPA (16/12/1994, revised September 2009) lists 16,890 individuals (≈ 11,316 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 11,323 pairs in the 1990s and 10,111 pairs in the 2000s. There were 9,167 individuals on Berneray and 8,233 individuals on Mingulay in 2014, a total of 17,400 individuals (≈ 11,658 pairs), and 11,453 individuals on Mingulay alone (Berneray not counted) in 2017 (JNCC 2020).

St Kilda SPA; According to SiteLink, razorbill on St Kilda SPA was “Unfavourable Declining” when most recently assessed (on 11/06/2016). The citation for St Kilda SPA (31/08/1992, revised September 2009) lists 3,810 individuals (≈ 2,553 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,546 pairs in the 1990s and 1,689 pairs in the 2000s. Counts of individuals in 2015 or 2016 were: 494 on Hirta, 155 on Dun, 103 on Boreray and Stacs, 68 on Soay and Stacs, so a total of 820 individuals (≈ 549 pairs) (JNCC 2020).

The Shiant Isles SPA; According to SiteLink, razorbill on The Shiant Isles SPA was “Favourable Recovered” when most recently assessed (on 08/06/2015). The citation for The Shiant Isles SPA (31/08/1992, revised September 2009) lists 10,950 individuals (≈ 7,337 pairs) of razorbills for this site. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 7,337 pairs in the 1990s and 5,391 pairs in the 2000s. There were 8,029 individuals (≈ 5,379 pairs) in 2015 (JNCC 2020).

Rathlin Island; The citation, and the standard data form (complied February 1999, revised September 2018) lists 8,922 individuals in 1985. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 5,978 pairs in the 1990s and 15,393 pairs in the 2000s. There were 22,975 individuals (≈ 15,393 pairs) in 2011 (JNCC 2020).

Skomer, Skokholm and seas off Pembrokeshire SPA; The Natura standard data form (compiled August 1982, revised January 2017) lists razorbill as a component of the seabird assemblage, with 4,300 individuals. Stroud et al. (2016) report breeding numbers of razorbills in this SPA as 2,854 pairs in the 1990s and 4,425 pairs in the 2000s. In 2018 there were 2,585 individuals at Skokholm, 410 individuals at Middleholm and 4,668 individuals at Skomer, a total of 7,663 individuals (≈ 5,134 pairs) for the SPA (JNCC 2020).

8.19 Puffin

Hermaness, Saxa Vord and Valla Field SPA; According to SiteLink, puffin on Hermaness, Saxa Vord and Valla Field SPA was "Unfavourable Declining" when most recently assessed (on 28/06/2017). The citation for Hermaness, Saxa Vord and Valla Field SPA (31/12/2001, revised September 2009) lists 55,000 individuals of puffins for this site in 1999. It is unclear how many pairs this represents, as there is no general conversion factor from individuals to pairs in the case of puffins. Mitchell et al. (2004) list 25,094 puffin AOBs (≈ pairs) at Hermaness in 1998-2002. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 25,400 pairs in the 1990s and 27,297 pairs in the 2000s. There were 23,661 AONs (pairs) in 2002 at Hermaness (JNCC 2020). In contrast, only 1,757 individuals were counted there in 2017 (Heubeck et al. 2017).

Noss SPA; According to SiteLink, puffin on Noss SPA was "Unfavourable Declining" when most recently assessed (on 10/05/2017). The citation for Noss SPA (16/08/1996, revised September 2009) lists 2,348 individuals of puffins for this site at classification. It is unclear how many pairs this represents, as there is no general conversion factor from individuals to pairs in the case of puffins. Mitchell et al. (2004) list 1,891 puffin AOBs (≈ pairs) at Noss in 1985-1988 where the count had been converted 1:1 from a count of individual birds. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 2,348 pairs in the 1990s and 802 pairs in the 2000s. There were 1,174 individuals in 2017 (JNCC 2020).

Foula SPA, According to SiteLink, puffin on Foula SPA was "Unfavourable No Change" when most recently assessed (on 06/05/2016). The citation for Foula SPA (27/11/1995, revised September 2009) lists 48,000 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 48,000 pairs in the 1990s and 22,500 pairs in the 2000s. There were 6,351 individuals in 2016 (JNCC 2020).

Fair Isle SPA; According to SiteLink, puffin on Fair Isle SPA was "Unfavourable Declining" when most recently assessed (on 01/04/2015). The citation for Fair Isle SPA (16/12/1994, revised September 2009) lists 23,000 individuals of puffins for this site. It is unclear how many pairs this represents, as there is no general conversion factor from individuals to pairs in the case of puffins, but the count in 1985-1988 was converted directly on a 1:1 basis from individuals to AOBs, which are approximately the same as pairs (Mitchell et al. 2004). Stroud et al. (2016) report breeding numbers of puffins in this SPA as 8,700 pairs in the 1990s and 7,278 pairs in the 2000s. There were 10,706 individuals in 2012 and 6,666 individuals in 2015 (JNCC 2020).

Hoy SPA; According to SiteLink, puffin on Hoy SPA was “Unfavourable Declining” when most recently assessed (on 29/06/2004). The citation for Hoy SPA (07/12/2000, revised September 2009) lists 3,500 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 3,500 pairs in the 1990s and no counts made in the 2000s. There were 361 individuals in 2016-2017 (JNCC 2020).

North Caithness Cliffs SPA; According to SiteLink, puffin on North Caithness Cliffs SPA was “Favourable Maintained” when most recently assessed (on 13/06/2016). The citation for North Caithness Cliffs SPA (16/08/1996, amended 26/04/2018) lists 2,080 pairs of puffins for this site at classification. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 1,750 pairs in the 1990s and less than 7,045 pairs in the 2000s. There were 3,036 individuals in 2015-2016 (JNCC 2020).

Forth Islands SPA; According to SiteLink, puffin on Forth Islands SPA was “Favourable Declining” when most recently assessed (on 26/05/2017). The citation for Forth Islands SPA (25/04/1990, amended 25 May 2018) lists 14,000 pairs of puffins for this site at classification. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 21,000 pairs in the 1990s and 50,355 pairs in the 2000s. The most recent counts were all in 2018 except for Isle of May (2017) and were counts of Apparently Occupied Burrows (AOBs) at all the large colonies (individuals at Bass Rock and Inchmickery). Counts were 39,200 at Isle of May, 2,640 at Craigleith, 1,000 at Fidra, 685 at The Lamb, 58 at Inchmickery and 2 at Bass Rock, giving a total for the SPA of 43,585 pairs (JNCC 2020).

Farne Islands SPA; The citation (compiled August 2018 Version 2.0) lists counts in 2008 and 2013 of puffins contributing to the seabird assemblage as 76,798 breeding adults, based on twice the average number of counted AOBs in those two years (38,399). Stroud et al. (2016) report breeding numbers of puffins in this SPA as 34,710 pairs in the 1990s and 36,835 pairs in the 2000s. There were 39,962 AOBs in 2013, 43,956 AOBs in 2018, 43,752 AOBs in 2019 (JNCC 2020).

Coquet Island SPA; The citation lists puffins as contributing to the seabird assemblage, added as a feature in 2016. The estimate is of 31,686 breeding adults, based on twice the average number of counted AOBs (15,843). Stroud et al. (2016) report breeding numbers of puffins in this SPA as 11,400 pairs in the 1990s and 15,812 pairs in the 2000s. There were 12,344 AOBs in 2013, 32,309 AOBs in 2018, 25,029 AOBs in 2019 (JNCC 2020).

Flamborough and Filey Coast SPA; The citation (compiled August 2018 Version 2.0) does not mention puffin. It states that a seabird assemblage qualifies as a feature but does not list the component species. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 958 pairs in the 2000s and suggest that this species would qualify as a component of the seabird assemblage but that this species was not included as a feature at that time. There were 2,879 individuals in 2017, 4,279 individuals in 2018 (JNCC 2020).

North Rona and Sula Sgeir SPA; According to SiteLink, puffin on North Rona and Sula Sgeir SPA was “Unfavourable No Change” when most recently assessed (on 19/06/2012). The citation for North Rona and Sula Sgeir SPA (30/10/2001, revised September 2009) lists 5,300 AOS (≈ pairs) of puffins for this site in 1986. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 5,250 pairs in the 1990s and 5,442 pairs in the 2000s. There are no counts listed by JNCC since 2001 for this site (JNCC 2020).

Sule Skerry and Sule Stack SPA; According to SiteLink, puffin on Sule Skerry and Sule Stack SPA was “Favourable Declining” when most recently assessed (on 10/07/2015). The citation for Sule Skerry and Sule Stack SPA (29/03/1994, revised September 2009) lists 46,900 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 43,380 pairs in the 1990s and 59,471 pairs in the 2000s. There were 33,552 AOBs on Sule Skerry in 2015 and 47,742 AOBs there in 2018 (JNCC 2020).

Cape Wrath SPA; According to SiteLink, puffin on Cape Wrath SPA was “Unfavourable No Change” when most recently assessed (on 05/07/2018). The citation for Cape Wrath SPA (15/03/1996, revised September 2009) lists 5,900 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 5,900 pairs in the 1990s and 1,602 pairs in the 2000s. There were at least 1,520 individuals in 2017 (JNCC 2020).

Flannan Isles SPA; According to SiteLink, puffin on Flannan Isles SPA was “Favourable Maintained” when most recently assessed (on 03/06/2013). The citation for Flannan Isles SPA (31/08/1992, revised September 2009) lists 4,400 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 5,500 pairs in the 1990s and 15,761 pairs in the 2000s. There are no counts published by JNCC for this site since 2001 (JNCC 2020).

St Kilda SPA; According to SiteLink, puffin on St Kilda SPA was “Favourable Maintained” when most recently assessed (on 31/07/2000). The citation for St Kilda SPA (31/08/1992, revised September 2009) lists 155,000 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 155,000 pairs in the 1990s and 142,264 pairs in the 2000s. No complete counts at this site are listed by JNCC since 2000 (JNCC 2020).

Mingulay and Berneray SPA; According to SiteLink, puffin on Mingulay and Berneray SPA was “Favourable Maintained” when most recently assessed (on 24/06/2017). The citation for Mingulay and Berneray SPA (16/12/1994, revised September 2009) lists 4,000 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 4,000 pairs in the 1990s and 3,126 pairs in the 2000s. The most recent count at Berneray was 2,498 individuals in 2009, while the most recent count at Mingulay was 1,318 individuals in 2012 (JNCC 2020).

Canna and Sanday SPA; According to SiteLink, puffin on Canna and Sanday SPA was “Favourable Maintained” when most recently assessed (on 09/07/1999). The citation for Canna and Sanday SPA (20/02/1998, revised September 2009) lists 1,200 individuals of puffins for this site at classification. It is unclear how many pairs this represents, as there is no general conversion factor from individuals to pairs in the case of puffins, but in this colony a 1:1 conversion from individuals to pairs has been assumed (Mitchell et al. 2004). Stroud et al. (2016) report breeding numbers of puffins in this SPA as 1,225 pairs in the 1990s and 945 pairs in the 2000s. There were 2,050 individuals (so 2,050 pairs) in 2016 and 1,935 individuals (so 1,935 pairs) in 2019 (JNCC 2020).

Shiant Isles SPA; According to SiteLink, puffin on Shiant Isles SPA was “Favourable Maintained” when most recently assessed (on 18/06/2015). The citation for Shiant Isles SPA (31/08/1992, revised September 2009) lists 77,000 pairs of puffins for this site. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 77,000 pairs in the 1990s and 65,170 pairs in the 2000s. There were 64,695 AOBs (pairs) in 2015 (JNCC 2020).

Rathlin Island; The citation (site classified 25/02/1999, registered 01/03/1999) lists 2,398 individual puffins in 1985. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 2,398 pairs in the 1990s and 731 pairs in the 2000s. There were 731 AOBs in 2007 and 695 AOBs in 2011 (JNCC 2020).

Skomer, Skokholm and seas off Pembrokeshire SPA; The citation (compiled August 1982, updated January 2017) lists 9,500 pairs. Stroud et al. (2016) report breeding numbers of puffins in this SPA as 9,500 pairs in the 1990s and 16,721 pairs in the 2000s. The most recent counts listed by JNCC are 7,447 individuals at Skokholm in 2019 and 30,895 individuals at Skomer in 2018 (JNCC 2020).