APP4SEA

ARCTIC PREPAREDNESS PLATFORM
FOR OIL SPILL AND OTHER ENVIRONMENTAL ACCIDENTS

APP4SEA
Oil Vulnerability & Seabirds
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Oil Vulnerability and Seabirds

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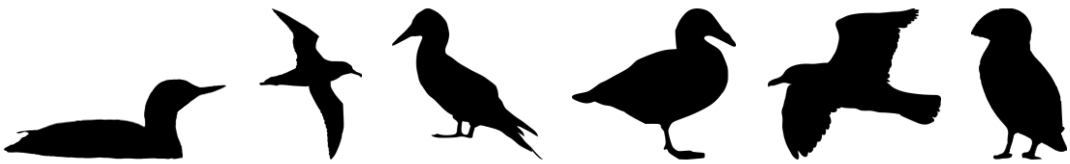

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Northern Periphery and Arctic Programme
2014–2020

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APP4SEA

The 21st century brought unprecedented interest in the Arctic resources, turning the region from the world's unknown periphery into the center of global attention.

Within the next 50 years, local coastal communities, their habitual environment and traditional lifestyle will undergo severe changes, starting from climatic perturbations and ending with petroleum industrial intervention and increased shipping presence.

The APP4SEA project, financed by the Northern Periphery and Arctic Programme will contribute to environmental protection of the Arctic waters and saving the habitual lifestyle of the local communities. It will improve oil spill preparedness of local authorities and public awareness about potential oil tanker accidents at sea.

Introduction

In recent decades, climate change has resulted in a reduction of ice cover and thickness in the Arctic, especially during the summer months, with increased melting of sea-ice predicted as global annual surface temperatures continue to rise (IPCC, 2013). This reduction in sea-ice has led to increasing political and commercial interest in the Arctic's resources as opportunities arise for new shipping trade routes, such as the Northern Sea Route, and access to unexploited hydrocarbon resources, especially oil (Miller and Ruiz, 2014; Wilkinson et al., 2017). However, an increase in shipping activity and hydrocarbon extraction also increases the risk of negative ecological impacts on marine habitats and organisms, for example, through shipping or oil extraction accidents, pipeline leaks, sub-surface well blowouts and accidental or deliberate discharge of oil during transportation (Clark, 2001; Wilkinson et al., 2017).

Seabirds are among the most threatened group of bird species, with 28% of the world's seabird species categorised as globally threatened (BirdLife International, 2019), and populations facing a multitude of threats, including pollution (Croxall et al., 2012; Dias et al., 2019). Seabirds are particularly vulnerable to oil pollution, which can affect species directly, through lethal and sub-lethal affects, and indirectly (Munilla et al., 2011; Piatt and Glenn Ford, 1996; Votier et al., 2005). Although, large oil spills and disasters can result in high mortality of individuals, persistent chronic oil pollution is thought to have the greatest impact on seabirds (O'Hara and Morgan, 2006; Ronconi et al., 2015; Wiese and Robertson, 2004). Furthermore, seabirds can be indirectly impacted by oil spills through displacement from foraging habitats and reduced food availability where prey are affected (Peterson et al., 2003; Velando et al., 2005). At colder latitudes, the vulnerability of seabirds to oil pollution may also be exacerbated (Fraser and Racine, 2016). Certain oil types will persist for longer at the sea surface in cold waters, although this is generally in a more viscous, solidified form (Brandvik and Faksness, 2008; Buist et al., 2000). Furthermore, seabirds may be more vulnerable due to potentially already being at a higher thermal stress, meaning that only small amounts of oil may cause hypothermia and therefore increase mortality risk (Hartung, 1967; Jenssen et al., 1985; Wiese and Ryan, 2003).

Given the predicted increase in shipping activity and hydrocarbon extraction in these northern waters there is a need to assess the vulnerability of seabirds to oil pollution, and to highlight locations of high risk. The most common approach to assess the vulnerability of seabirds to oil pollution, and other anthropogenic activities, is through calculating an index for the sensitivity of seabirds to oil, based on species-specific behaviours and life history traits, to create an Oil Vulnerability Index (OVI) (King and Sanger, 1979; Williams et al., 1994). These species-specific indexes can be combined with data on species distributions and densities to create a spatial OVI to identify specific locations where seabirds are most vulnerable to anthropogenic activities such as oil pollution (Webb et al., 2016).

The eastern North Atlantic holds internationally important numbers of seabirds due to cool sea temperatures and high productivity (BirdLife International, 2017; Wong et al., 2014). Within this region, spatial OVI data for seabirds exists for some locations within this region to varying degrees (summarised in Camphuysen, 2007; also for the UK Continental Waters: Webb et al., 2016; Faeroese Waters: Skov et al., 2002; west Greenland: Clausen et al., 2016). However, not all jurisdictions have methods for assessing risks to seabirds from oil, and there is no region wide assessment. One of the main reasons for a lack of seabird vulnerability assessments to oil pollution in this region is the scarcity of data for under-studied seabird species, particularly those that breed in the arctic, and limited year-round information on seabird distributions and densities at sea from vessel and aerial surveys.

A lack of data on seabird, but also sea mammal, demography, how species behaviour may influence their sensitivity to oil and at-sea distributions and densities is an issue for many data-deficient locations globally. However, it is important to understand how anthropogenic activities might affect species in these locations. It is critical therefore to highlight what type of data, and level of understanding, is required to robustly assess potential negative impacts of incidents, such as oil spills, in regions where data is lacking. It is also important to establish which methods can be used when existing empirical data is not available.

In this study, we explored the practicalities of creating a spatial OVI for seabirds within a region of the eastern North Atlantic Ocean that may be increasingly affected by oil incidents. We highlight the challenges and difficulties incurred due to data-deficiency in seabird demography and seabird at-sea information. Secondly, to establish where seabird in the eastern North Atlantic Ocean may be most at risk to oil pollution, we mapped marine shipping routes and hydrocarbon exploration/extraction sites. We highlight how this approach can be used to identify hotspots where seabirds may be most at risk to oil pollution and therefore where data collection should be a priority.

Methods

We focused on an area of the eastern North Atlantic Ocean that included the sea regions of the following non-continental European countries and autonomous territories: Denmark, the Faroe Islands, east Greenland, Iceland, Republic of Ireland, Norway, Svalbard (including Bjørnøya and Jan Mayan), and the UK. We included species categorised as seabirds following Gaston (2004), which included the tubenoses (Procellariidae, Hydrobatidae), cormorants (Phalacrocoracidae), gannets (Sulidae), phalaropes (Charadriidae: Phalaropus spp.), skuas, gulls, and, terns (Laridae), and auks (Alcidae). We also included loons (Gaviidae), sea ducks, mergansers (Anatidae: Mergini), and grebes (Podicipedidae), as these species spend a large proportion of the year at sea (Gaston, 2004). All seabird species known to breed within the listed northeastern Atlantic countries were included (del Hoyo et al. 2016). Throughout, we followed the taxonomic treatment of The Hand-book of the Birds of the World (HBW) and BirdLife International (del Hoyo and Collar, 2014). We identified 62 seabird species that commonly occur within the eastern North Atlantic (Table 1).

Table 1. OVI scores of widespread migrant and breeding seabird species present in the eastern North Atlantic.

Common name	Scientific name	2019 IUCN Red List	Status	OVI Score
Red-throated Loon	*Gavia stellata*	LC	Breeding	0.511
Arctic Loon	*Gavia arctica*	LC	Breeding	0.538
Common Loon	*Gavia immer*	LC	Breeding	0.563
Yellow-billed Loon	*Gavia adamsii*	NT	Breeding	0.703
Red-necked Grebe	*Podiceps grisegena*	LC	Breeding	0.300
Great Crested Grebe	*Podiceps cristatus*	LC	Breeding	0.300
Horned Grebe	*Podiceps auritus*	VU	Breeding	0.570
Black-necked Grebe	*Podiceps nigricollis*	LC	Breeding	0.336
Northern Fulmar	*Fulmarus glacialis*	LC	Breeding	0.282
Cory's Shearwater	*Calonectris borealis*	LC	Migrant	0.203
Great Shearwater	*Ardenna gravis*	LC	Migrant	0.211
Sooty Shearwater	*Ardenna grisea*	NT	Migrant	0.266
Manx Shearwater	*Puffinus puffinus*	LC	Breeding	0.333
Balearic Shearwater	*Puffinus mauretanicus*	CR	Migrant	0.592
European Storm-petrel	*Hydrobates pelagicus*	LC	Breeding	0.089
Leach's Storm-petrel	*Hydrobates leucorhous*	VU	Breeding	0.133
Northern Gannet	*Morus bassanus*	LC	Breeding	0.282
Great Cormorant	*Phalacrocorax carbo*	LC	Breeding	0.345
European Shag	*Phalacrocorax aristotelis*	LC	Breeding	0.435
Common Eider	*Somateria mollissima*	NT	Breeding	0.542
King Eider	*Somateria spectabilis*	LC	Breeding	0.420

Steller's Eider	*Polysticta stelleri*	VU	Breeding	0.570
Harlequin Duck	*Histrionicus histrionicus*	LC	Breeding	0.336
Long-tailed Duck	*Clangula hyemalis*	VU	Breeding	0.570
Common Scoter	*Melanitta nigra*	LC	Breeding	0.336
Velvet Scoter	*Melanitta fusca*	VU	Breeding	0.657
Goldeneye	*Bucephala clangula*	LC	Breeding	0.300
Goosander	*Mergus merganser*	LC	Breeding	0.260
Red-breasted Merganser	*Mergus serrator*	LC	Breeding	0.270
Greater Scaup	*Aythya marila*	LC	Breeding	0.287
Red-necked Phalarope	*Phalaropus lobatus*	LC	Breeding	0.041
Red Phalarope	*Phalaropus fulicarius*	LC	Breeding	0.048
Pomarine Jaeger	*Stercorarius pomarinus*	LC	Breeding	0.203
Arctic Jaeger	*Stercorarius parasiticus*	LC	Breeding	0.255
Long-tailed Jaeger	*Stercorarius longicaudus*	LC	Breeding	0.255
Great Skua	*Catharacta skua*	LC	Breeding	0.319
Mediterranean Gull	*Larus melanocephalus*	LC	Breeding	0.231
Little Gull	*Hydrocoloeus minutus*	LC	Breeding	0.161
Sabine's Gull	*Xema sabini*	LC	Migrant	0.194
Black-headed Gull	*Larus ridibundus*	LC	Breeding	0.255
Mew Gull	*Larus canus*	LC	Breeding	0.272
Lesser Black-backed Gull	*Larus fuscus*	LC	Breeding	0.239
European Herring Gull	*Larus argentatus*	LC	Breeding	0.227
Yellow-legged Gull	*Larus michahellis*	LC	Migrant[1]	0.227
Iceland Gull	*Larus glaucoides*	LC	Breeding	0.138
Glaucous Gull	*Larus hyperboreus*	LC	Breeding	0.138
Great Black-backed Gull	*Larus marinus*	LC	Breeding	0.299
Ross's Gull	*Rhodostethia rosea*	LC	Breeding	0.121
Black-legged Kittiwake	*Rissa tridactyla*	VU	Breeding	0.436
Ivory Gull	*Pagophila eburnea*	NT	Breeding	0.254
Sandwich Tern	*Thalasseus sandvicensis*	LC	Breeding	0.171
Roseate Tern	*Sterna dougallii*	LC	Breeding	0.195
Common Tern	*Sterna hirundo*	LC	Breeding	0.205
Arctic Tern	*Sterna paradisaea*	LC	Breeding	0.162
Little Tern	*Sternula albifrons*	LC	Breeding	0.198
Black Tern	*Chlidonias niger*	LC	Breeding	0.084
Common Murre	*Uria aalge*	LC	Breeding	0.585
Thick-billed Murre	*Uria lomvia*	LC	Breeding	0.585
Razorbill	*Alca torda*	NT	Breeding	0.721
Black Guillemot	*Cepphus grylle*	LC	Breeding	0.563
Little Auk	*Alle alle*	LC	Breeding	0.563
Atlantic Puffin	*Fratercula arctica*	VU	Breeding	0.843

Calculating species-specific OVI scores

To calculate specific-species OVI scores for seabirds in the eastern North Atlantic region, we adapted the approach taken by the updated OVI for the UK continental shelf, the Seabird Oil Sensitivity Index (SOSI; Webb et al., 2016), as the basis for developing a regional OVI. A full rationale of why we selected this approach is given in O'Hanlon et al., (2020), where we also discussed the relevance of the factors used to assess oil sensitivity in the UK to the larger eastern North Atlantic region of interest.

The SOSI for the UK continental shelf incorporates eight factors that represent three principals to assess the sensitivity of seabird species to oil incidents: 1) how likely individuals are to be affected by oil due on their behaviour (SOSI factors 1-3); 2) how vulnerable a population/species is (SOSI factors 4-6); and how quickly a population/species might recover from an oil incident (SOSI factors 7-8) (Webb et al., 2016). The eight factors are scored on a scale of 0.2 to 1.0, from low to high sensitivity (Webb et al., 2016).

1. **Proportion of time spent sitting on the water**
2. **Percentage of tideline corpses contaminated with oil**
3. **Habitat flexibility**
4. **Percentage of biogeographical population within the UK Continental Shelf**
5. **Listing in Birds of Conservation Concern**
6. **Presence on EU Birds Directive Annexes**
7. **Potential annual productivity (maximum and mean clutch size & age at first breeding**
8. **Adult survival rate**

Sensitivity analysis

Data on seabird species within the region of interest is variable, especially for less studied species, therefore data was not available to score all factors for all species. To determine the relative importance of the SOSI factors, and identify which are the most influential for calculating species-specific OVI scores, we performed a sensitivity analysis on the eight SOSI factors. The sensitivity analysis also allowed us to assess whether spatial variability in parameter values, especially for factor 1 (proportion of time spent on water), 2 (proportion of tideline corpses contaminated with oil), 7 (potential annual productivity), and 8 (annual adult survival rate), need to be taken into account given the large geographical area of interest.

Analyses were undertaken in R 3.5.1 (R Core Team, 2018). To determine the relative importance of the eight SOSI factors on the SOSI calculation we carried out a sensitivity analysis with Latin Hypercube Sampling (Blower and Dowlatabadi, 1994; McKay, 1992), in the R package *pse* (Chalom and Prado, 2017). Full details on the sensitivity analysis are described in O'Hanlon et al., (2020). In brief, the relative importance of each factor was established by calculating partial rank correlation coefficients (PRCC), with values closer to 1 having a stronger influence on the final SOSI scores (Blower and Dowlatabadi, 1994). The three factors relating to how likely individuals are to be affected by oil pollution based on their behaviour had the greatest influence on the species-specific SOSI scores (factor 1 - proportion of time spent on water, factor 2 - proportion of tideline corpses contaminated with oil, and factor 3 - habitat flexibility), all having high PRCC values > 0.50 (Thomas and Taylor, 1990). In comparison, factors 7 and 8, which reflect how quickly a population/species might recover from an oil incident based on the species demography (potential annual productivity, and adult annual survival rate), had the least influence on the final SOSI scores. Therefore, to use the SOSI approach accurate data, ideally considering temporal and spatial variability, is required to score factor 1- 3, whilst there is more flexibility in the confidence of data used to score factors 7-8, with factors 4-6 being intermediate in importance.

Calculating species-specific OVI scores for the eastern North Atlantic

To calculate OVI scores for the widespread seabird species found within the eastern North Atlantic we used the following sources to obtain data on each factor.

For species included within the UK SOSI, values for the percentage of time species spent sitting on the water (factor 1) and percentage of tideline corpses contaminated with oil (factor 2) were taken from Webb et al., (2016), which were obtained from European Seabird at Sea (ESAS) data within the UK continental shelf area between 1995 and 2015. For species (n = 9) not included in this study we used values from similar species. Given the influence of these two factors on the final species-specific SOSI scores revealed by the sensitivity analysis this is suboptimal. However, for these more northerly and Arctic breeding seabird species, data does not easily exist to provide accurate parameter values for these two factors. This is an important consideration when using this approach to assess oil vulnerability in data-deficient locations. Data from beached bird surveys, which are used to determine the proportion of tideline corpses contaminated with oil are not available for more inaccessible coastlines (e.g., Greenland, Iceland or Svalbard). We also do not account for potential spatial or temporal variation in these two factors. However, as the proportion of time individuals spend on the sea surface is largely driven by behaviour, this is likely to remain relatively consistent within a species across time and space. This is likely also to be the case for factor 2 as the proportion of time individuals spend on the sea surface is positively correlated to the proportion of tideline corpses contaminated with oil, with species that spend more time on the sea surface having higher oiling rates, (Camphuysen, 1998; O'Hanlon et al., 2020). Although the proportion of tideline corpses contaminated with oil for species across different locations differ, the ranking of oiling rates for species are generally similar (Camphuysen, 1998).

Values for habitat flexibility (factor 3), for species included within the UK SOSI, were taken from Webb et al. 2016, which were based on the scores used by Furness et al., (2013). For the species not represented by these studies we determined their habitat flexibility scores based on their habitat use and foraging ecology as described in the literature and from the Handbook of the Birds of the World Alive (del Hoyo et al., 2018).

The three factors within the SOSI that reflect how vulnerable a population/species is are specifically related to the UK (factors 4 and 5) or the European Union (factor 6) (Webb et al., 2016). To consider how vulnerable a population or species is over a larger geographical scale, and to be useful globally, we replaced the SOSI factors 4-6 with a single factor to reflect the global conservation status of each species as categorised on the IUCN Red List (BirdLife International, 2019). By using a global measure of conservation status this modified OVI can be used anywhere in the world, with species-specific scores being comparable. However, this does mean we do not consider any spatial or temporal variation in a species' conservation status. Factor 4 is the OVI used here is therefore the IUCN Red List category.

Across broad geographical scales, such as the region of interest, there can be considerable spatial and temporal variation in seabird demography, including in maximum and mean clutch size, age at first breeding and adult survival rate (Horswill and Robinson, 2015), which the SOSI approach does not take into consideration. However, the relatively low influence of factors 7 and 8, regarding a species demography, on the final species-specific scores indicates that it may not be necessary to account for this variation in traits. This also means that it may not be essential to account for uncertainty in values for understudied species and populations, where values are estimated using data from similar species, expert opinion, or local ecological knowledge. Furthermore, as parameter values are assigned to bins of 0.2 increments, small variation in demographic values will not alter the final species-specific scores.

For species the occur in the UK, data to score factors 7 (potential annual productivity) and 8 (adult survival) were obtained from Horswill and Robinson (2015). For the remaining species, data was acquired from the literature and *Handbook of the Birds of the World Alive* (del Hoyo et al., 2018). Where data could not be found for species to score factors 7 and 8 (factor 5 and 6 is our OVI) we used the scores of similar species.

Equation 1.

$$OVI_i = (F_1 \times F_2)^{1-\frac{F_3}{F_3+0.5}} \times (F_4)^{1-\frac{\left(\frac{F_5+F_6}{2}\right)}{\left(\frac{F_5+F_6}{2}\right)+0.5}}$$

Given the uncertainty around the values for Factors 1 and 2 for many areas of the region of interest we also created spatial OVI maps using OVI scores using the above equation but 1) removing Factor 1 and 2) replacing Factor 2 with Factor 1 (so using F1 x F1).

Equation 2.

$$OVI_i = (F_1)^{1-\frac{F_3}{F_3+0.5}} \times (F_4)^{1-\frac{\left(\frac{F_5+F_6}{2}\right)}{\left(\frac{F_5+F_6}{2}\right)+0.5}}$$

Equation 3.

$$OVI_i = (F_1 \times F_1)^{1-\frac{F_3}{F_3+0.5}} \times (F_4)^{1-\frac{\left(\frac{F_5+F_6}{2}\right)}{\left(\frac{F_5+F_6}{2}\right)+0.5}}$$

Seabird at sea distribution and density data for the Spatial OVI

To identify locations where seabirds are most vulnerable to oil pollution, it is necessary to combine the calculated species-specific OVI scores to data on seabird densities and distributions at sea. In some geographical areas there is relatively good data on seabird-at-sea densities and distributions from aerial and vessel surveys (Webb et al., 2016). However, for the majority of the region of interest data from aerial and vessel surveys are lacking (Camphuysen, 2007). An alternative method of estimating at-sea densities and distributions of seabirds during the breeding season is using a foraging radius approach (Critchley et al., 2018; Grecian et al., 2012; Soanes et al., 2016). This method uses a predictive radius approach, which uses the maximum foraging distance and colony sizes from all known seabird colonies for each species within an area of interest (Critchley et al., 2018; Grecian et al., 2012).

Seabird colony locations and populations sizes were obtained for each country bordering the eastern North Atlantic region of interest. Data was collected from national seabird censuses for Demark, Greenland, Ireland, Norway and Svalbard, and the UK and Ireland. Except for Denmark, we used the most recent colony count available for each colony. For Denmark, for each colony, the maximum colony size was used between 2005 and 2017 for all species except for Common Murre *Uria aalge* and Razorbill *Alca torda* (maximum colony size between 2010 and 2015) and Mew Gull *Sterna hirundo*, Great Cormorant *Phalacrocorax carbo* and Sandwich Tern *Thalasseus sandvicensis* (maximum colony size between 2010 and 2015).

Collated data from national census were not available for the Faroes and Iceland, therefore seabird colony data were obtained from the most recent literature available. For the Faroes Islands we obtained initial colony size and location data for the country's most important seabird colonies from published Faroese Important Bird Area (IBA) information (Grimmett and Jones, 1989). For each species we compared the total Faroese population estimates from these IBAs to national population estimates published in 2004 (Jensen et al., 2004). Both the Arctic Jaeger *Stercorarius parasiticus* and Great Skua *Catharacta skua* populations had increased over this period, whilst populations of Black-legged Kittiwake *Rissa tridactyla*, Common Murre and Razorbill had declined. Therefore, for these species we altered the colony sizes of the IBA colonies by the same scale of decline or increase to reflect the more recent population estimates. For the remaining species whose populations had remained relatively stable we used the colony sizes as provided in Grimmett and Jones (1989). As the Arctic Tern *Sterna paradisaea* colony sizes were provided as a range (i.e. 10-100 AONs), we used the mid-point of the range as an estimate of the colony size. The colony sizes given for the Faroese Important Bird Area were at the island level. However, there for each island a map was provided where the seabird colonies were predominantly located (Grimmett and Jones, 1989). To create the predicted at-sea densities for each species, we created points at 100m intervals along the stretches of coastline identified, which we divided the island-wide seabird population estimates into. This provides a more realistic scenario of where the seabirds were breeding to centre the foraging ranges from.

We were also able to obtain seabird population estimates for IBAs in Iceland (Skarphéðinsson et al., 2016). For most species, this publication also had population estimates for other important sites across Iceland. Although there was only a single population estimate covering all these other important sites for each species, maps were included showing the location of these other sites. We therefore, used these maps to estimate colony sizes for these other important locations. Due to the large size of many of the Iceland IBAs, we also used these maps to help determine where to locate information of colony size within these IBAs. Where ranges were given for colony size estimates, the midpoints were used. For many species the population sizes given for these IBAs and other important areas contribute all, or a high percentage, of Iceland's population estates for these species (Skarphéðinsson et al. 2016). However, for some species additional information was required to estimate additional colony locations and sizes. The majority of Mew Gulls breed in the Eyjafjörður region in north-east Iceland, therefore we used this location for this species (Thorstensen and Petersen, 2013). For the Glaucous Gull *Larus hyperboreus* additional, more recent, colony information was obtained from Petersen et al., (2015). For many Black-headed Gulls, Herring Gulls *Larus argentatus* and Black Guillemots *Cepphus grylle* information on colony locations and sizes were unavailable. Although, we did find published colony information for an approximately 11% of Black-headed Gulls *Larus ridibundus* colonies (Petersen and Thorstensen, 2005, 1993; Thorstensen and Petersen, 2013). For these species we made the assumption that the remaining breeding population, outside of the IBAs and other important areas, were distributed evenly around Iceland, and created 100 random points around the coastline of Iceland at equal intervals to plot the remaining population. This is unlikely to be realistic, however it was the best option given we had no alternative information on where these colonies may be located. We also took this approach for the Arctic Jaegar, which breeds across much of Iceland, both near the coast and further inland.

We managed to obtain spatial data for 31 of the 62 seabird species for which we calculated OVI scores across the eastern North Atlantic region of interest (Table 2). Suitable breeding location and population size data on seaducks, divers, grebes and phalaropes were not available for all countries in the region therefore these species were not included in the spatial

OVI. As the foraging radius approach focuses on the breeding season most of the species that could not be included breed inland during this period and therefore are expected to spend less time in the marine environment. However, it is important that these species are included in any maps covering the non-breeding season.

To generate species-specific predicted at sea distributions applying the predictive foraging radius approach we used R script provide by Critchley et al., (2018). This approach predicts the total number of individuals in every grid square across the region. The maximum foraging ranges that we used for each species are provided in Table 2.

For morphological similar species, some colony information was not identified to specific species: Common *Sterna hirundo* and Arctic Terns (16 colonies from the UK); Glaucous and Iceland Gulls *Larus glaucoides* (340 colonies from Greenland): Herring and Lesser Black-backed Gulls *Larus fuscus* (6 colonies from the UK). For these Common and Arctic Tern colonies, we used the higher OVI score of the Common Tern. For mixed Herring and Lesser Black-backed Gull colonies we used the lower maximum foraging range of the Herring Gulls and the higher OVI score of the Lesser Black-backed Gull. We used the higher OVI scores as a precautionary measure.

To create a breeding season spatial OVI based on the species-specific predicted at sea distributions and densities we used the following equation used by Webb et al., (2018), which puts greater emphasis on species that are more vulnerable to oil pollution (Certain et al., 2015; Webb et al., 2016).

Equation 4.

$$OVI_j = \sum_i^S \frac{\widehat{D}_{ij}}{1 - OVI_j}$$

OVI_j = overall OVI score at location j, $\widehat{D}_{ij}$ = density of species i (number/5 km²) at location j, OVI_i = OVI score for species i.

As we did not obtain seabird colony data from all countries surrounding the eastern North Atlantic we clipped the area of interest to ensure that we had accounted for all breeding birds that may forage within this area. This meant that we excluded west Greenland, as we did not include data for Canada or the United States. We also only included Scotland and the north-west coastline of Northern Ireland and the excluded the south of Norway as data was not obtained from the rest of mainland Europe.

Seabird phenology

Given the large geographical scale of the region of interest, seabird breeding phenology is likely to vary considerably over space, as well as between breeding season (Descamps et al., 2019; Keogan et al., 2018). To incorporate a temporal element into the spatial OVI one option is to create monthly maps to take into account the variability in breeding season phenology across species within the region of interest. Taking the Atlantic Puffin as an example, we collated information on mean hatching date from colonies across the region, from multiple years. The Atlantic Puffin was selected as it is a well studied species that breeds across the region of interest. We estimated the mean laying date by subtracting 40.5 days, the mean length of the incubation period, from the mean hatching date (Harris and Wanless, 2011). There was a significant positive relationship between latitude and mean laying date across colonies and years (GLMM with latitude and year as fixed effects and colony as a random effect: $t = 2.90$, $p < 0.001$, $R2 = 0.19$). Year and an interaction between year and latitude were not found to be significantly related to mean laying date ($p = 0.35$). However, there was much variability in the mean laying date among colonies and years, within and between colonies (Figure 1), with latitude only explaining part of this variation. Due to this variability, it is difficult to correct the spatial OVI for breeding phenology across the region. Furthermore, as we used fixed values for colony size, location and maximum foraging range for each species there is no variability in at sea distributions and density between months of the breeding season for each species at a given location. We therefore created a single spatial OVI map to cover the whole breeding season of seabirds in this region.

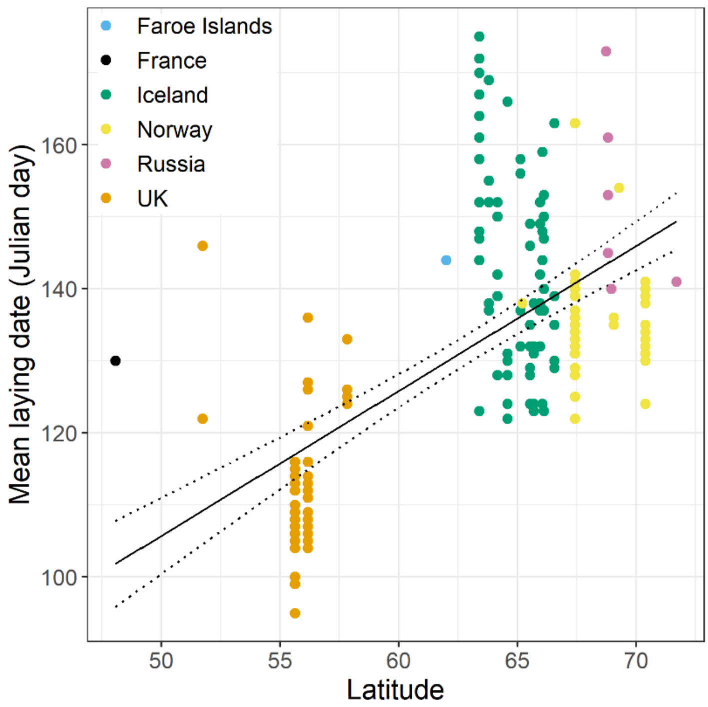

Figure 1.

The mean laying date of Atlantic Puffin across the eastern North Atlantic was positively related to latitude. However, there is a lot of variation among locations and years.

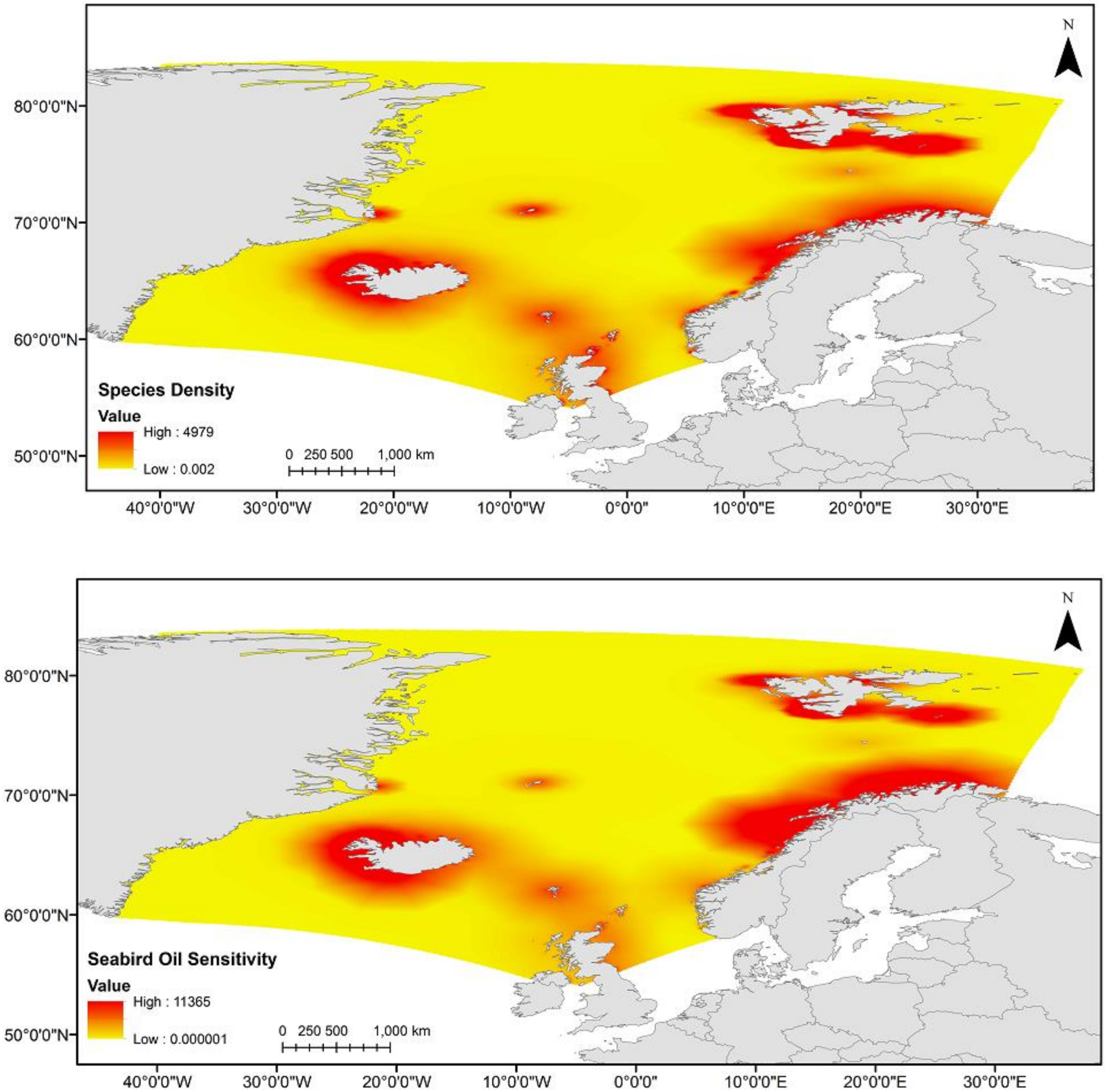

Figure 2. Eastern North Atlantic region of interest showing a) the predicted density of 31 seabird species and b) seabird vulnerability calculated from the predicted seabird densities and the oil vulnerability scores.

Areas of risk to seabirds from oil pollution in the eastern North Atlantic

The spatial OVI highlights areas across the eastern North Atlantic where seabirds are most vulnerable to oil pollution incidences during the breeding season, based on currently available data. It is important to note here that for some species and countries the data used to create the spatial OVI is not up to date. Due to these limitations associated with the data, our confidence in how accurately this map reflects the current situation of seabird at sea distribution across this region is low, especially surrounding the Faroes and Iceland. However, although the map does not reflect current, absolute distribution, and vulnerability to oil, information for seabirds across this region, it does provide an indication of which areas seabirds are most likely to be vulnerable to anthropogenic threats at sea, such as oil pollution. To highlight areas of potential high risk to seabirds from oil pollution, the resulting spatial OVI map can be overlayed with spatial information on activities that may result in an oil pollution incident, such as hydrocarbon extraction sites and shipping intensity. Locations where areas of high risk overlap with areas of high seabird sensitivity can then be identified as areas where seabirds are most at risk to oil pollution.

Monthly data on shipping intensity across the region of interest was downloaded from https://www.emodnet-humanactivities.eu/view-data.php /
As we only had seabird at sea density estimates for the breeding season, we used mean vessel density monthly data for March to September 2018, with density measured as the total ship time in hours spent in each 1 km^2 cell over a month. We selected 2018 to highlight areas where seabirds may currently be at risk from oil pollution associated with at-sea vessels. Using the Raster package in R we calculated the mean vessel density for each cell by stacking the seven monthly rasters and creating a new raster with the new monthly values for the period which reflects the seabird breeding season. We clipped the vessel density raster to the same extent as our region of interest used for the seabird OVI map (vessel density data was not available for the corner of southeast Greenland, therefore this area is missing from the map). To calculate the mean shipping intensity within each seabird OVI grid square we carried out a spatial join in R.

Following the methodology set out by Renner and Kuletz (2014), we estimated the potential risk to seabirds of oil pollution from sea vessels using the following calculation in the Raster Calculator function in ArcMap:

Equation 5.

$$\text{Risk} = \frac{\text{Vessel Density}}{\sigma(\text{Vessel Density})} \times \frac{\text{Seabird OVI}}{\sigma(\text{Seabird OVI})}$$

We downloaded data on hydrocarbon licensing blocks for the UK (http://itportal.decc.gov.uk/web_files/gis/kml/DECC_OFF_Hydrocarbon_Fields.kmz) and Norway (https://kartkatalog.geonorge.no/metadata/oljedirektoratet/npd-factmapsdata-3-0/). We included all identified hydrocarbon extraction fields, including those which are no longer active and those which may become active in future, as we were interested in whether hydrocarbon extraction sites were located in areas with high seabird OVI scores, and therefore areas where seabirds may be vulnerable to oil pollution. For each grid cell in the Seabird OVI layer we identified any overlap with a hydrocarbon field using the zonal statistics function in ArcMap.

Table 2. Oil Vulnerability Index (OVI) scores for the widespread breeding seabirds in the eastern North Atlantic included in the spatial OVI, with breeding colony information and maximum foraging ranges.

Common name	Scientific name	Number of colonies	Number of breeding pairs	Max. foraging range (km)	OVI Score
Northern Fulmar	*Fulmarus glacialis*	3166	2760424	664 [1]	0.282
Manx Shearwater	*Puffinus puffinus*	383	209922	1219 [1]	0.333
European Storm-petrel	*Hydrobates pelagicus*	442	418344	365 [1]	0.089
Leach's Storm-petrel	*Hydrobates leucorhous*	35	247427	1154 [1]	0.133
Northern Gannet	*Morus bassanus*	44	331410	709 [2]	0.282
Great Cormorant	*Phalacrocorax carbo*	943	198111	50 [2]	0.345
European Shag	*Phalacrocorax aristotelis*	1916	77416	24 [1]	0.435
Arctic Jaeger	*Stercorarius parasiticus*	1900	19759	75 [2]	0.255
Great Skua	*Catharacta skua*	771	14225	219 [2]	0.319
Mediterranean Gull	*Larus melanocephalus*	31	122	20 [2]	0.231
Sabine's Gull	*Xema sabini*	48	1338	92 [3]	0.194
Black-headed Gull	*Larus ridibundus*	1351	351739	40 [2]	0.255
Common Gull	*Larus canus*	4498	177013	50 [2]	0.272
Lesser Black-backed Gull	*Larus fuscus*	1928	209738	181 [2]	0.239
European Herring Gull	*Larus argentatus*	5205	423866	92 [2]	0.227
Iceland Gull	*Larus glaucoides*	529	36244	92 [3]	0.138
Glaucous Gull	*Larus hyperboreus*	1356	27182	92 [3]	0.138
Great Black-backed Gull	*Larus marinus*	5547	113946	60 [2]	0.299
Black-legged Kittiwake	*Rissa tridactyla*	1521	2356774	229 [1]	0.436
Ivory Gull	*Pagophila eburnea*	47	2488	92 [3]	0.254
Sandwich Tern	*Thalasseus sandvicensis*	73	29461	54 [2]	0.171
Roseate Tern	*Sterna dougallii*	10	789	30 [2]	0.195
Common Tern	*Sterna hirundo*	1200	34610	30 [2]	0.205
Arctic Tern	*Sterna paradisaea*	3000	450376	30 [2]	0.162
Little Tern	*Sternula albifrons*	292	3778	11 [2]	0.198
Common Murre	*Uria aalge*	850	2047882	339 [1]	0.585
Thick-billed Murre	*Uria lomvia*	203	3026668	168 [4]	0.585
Razorbill	*Alca torda*	1199	554538	314 [1]	0.721
Black Guillemot	*Cepphus grylle*	3004	120295	15 [2]	0.563
Little Auk	*Alle alle*	235	6159104	110 [4]	0.563
Atlantic Puffin	*Fratercula arctica*	991	6625150	383 [1]	0.843

[1] BirdLife International, 2019; LC Least Concern, NT Near Threatened, VU Vulnerable. [2] Oppel et al., 2018. [3] Critchley et al., 2018; [4] Foraging range not available therefore we used the maximum froaging range of the European Herring Gull; [5] Jovani et al., 2015

Results

Calculating species-specific OVI scores for the eastern North Atlantic

We calculated OVI scores for 62 seabird species that are widespread as breeding or migrants in the eastern North Atlantic (Table 1). As described above, and in O'Hanlon et al. (2020) in more detail, we modified the approach taken by Webb et al., (2016) to calculate species-specific OVI scores that were appropriate to a larger geographical region. The main change we made was replacing the three factors covering how vulnerable a population/species is (Factors 4-6 in the SOSI) with a single factor, a species IUCN Red List status (Birdlife International 2019).

For less well studied species, particularly those breeding in the Arctic data was not always available to score the five remaining OVI factors. Where we could not use species-specific information to score these factors, we instead used values from taxonomically similar species, with similar behaviour and demography. The extent to which using values from substitute species effected the overall OVI score depended on the influence of that factor on the overall OVI score, as shown from the results of the sensitivity analysis. Factor 1 and 2, and to a lesser extent factor 3, had the largest influence on the OVI score (O'Hanlon et al., 2020). One concern for expanding the SOSI approach to a larger area was the potential vast variability in demography from seabird populations in different areas of the eastern North Atlantic. However, unless the variability is particularly extreme, it is unlikely that any spatial or temporal variation would have a marked effect on the overall species-specific OVI scores given the small influence the factors relating to species demography (factors 5 and 6 within our OVI calculation, factors 7 and 8 within the SOSI) have on the final score, especially as the values are assigned to bins of 0.2 increments.

Seabird at sea distribution and density data for the Spatial OVI

The greatest challenge of creating an OVI for the larger region of the eastern North Atlantic was obtaining suitable spatial data on the distribution and density of all the selected widespread seabird species in the region of interest. Currently available data on year-round seabird distributions from vessel and aerial surveys is variable across the region, with some areas having much better coverage than others, with a similar pattern for seabird tracking data. It was therefore not possible to use seabird at-sea or tracking data to create satisfactory year-round estimates of seabird densities and distributions for the eastern North Atlantic. Instead we opted for a simpler approach to predict estimated at sea densities and distributions, during the breeding season, using the foraging radius approach (Critchley et al., 2018; Grecian et al., 2012; Soanes et al., 2016). We therefore did not attempt to estimate at sea densities and distributions for the non-breeding season.

We collated colony size and location information for 31 seabird species that breed across the eastern North Atlantic region of interest, obtained from six countries. Following the foraging radius approach, we created a predicted at-sea density distribution map of all 31 breeding species for which we had colony data for (Figure 2a). We then used this at-sea density distribution map to create the spatial OVI (Figure 2b). Although the calculation used to create the spatial OVI values puts greater emphasis on those species which are most vulnerable to oil pollution (species density divided by (1 – species-specific OVI score); Webb et al., 2016), the two maps are very similar with the highest density seabird areas being the areas where seabirds are most vulnerable to oil. There was a significant positive correlation the predicted at-sea seabird density raster and the spatial OVI raster (r = 0.91).

Areas of risk to seabirds from oil pollution in the eastern North Atlantic

We mapped two sources of potential oil pollution that may result in risk to seabirds at sea to highlight where seabirds may be most at risk to oil, 1) vessel density (in 2018) and 2) hydrocarbon extraction sites (for the UK and Norway). Vessel density was particularly high along the coasts of Scotland and Norway (Supplementary Figure S1). To establish the spatial distribution of risk to oil incidents associated with shipping activities, we rescaled vessel density and the spatial OVI to highlight where areas of higher than average vessel density match with areas of higher than average seabird vulnerability. On the linear scale, there are few locations where seabirds are potentially at high risk from oil pollution from shipping activities (Figure 3a), with these areas largely being located along the coast of Norway and off west Iceland. The log scaled map reveals the lower and intermediate risk areas, which includes most coastal regions where many seabirds will likely be present during the breeding season (Figure 3b).

Spatial OVI values of all UK and Norwegian hydrocarbon extraction fields showing that seabirds may be at greater risk from oil incidences associated with extraction sites in north Norway than in the North Sea (Figure 4). However, the maximum seabird OVI value in the hydrocarbon extraction fields was 3352, which is much lower than the overall maximum OVI value of 11000 in the eastern North Atlantic region of interest.

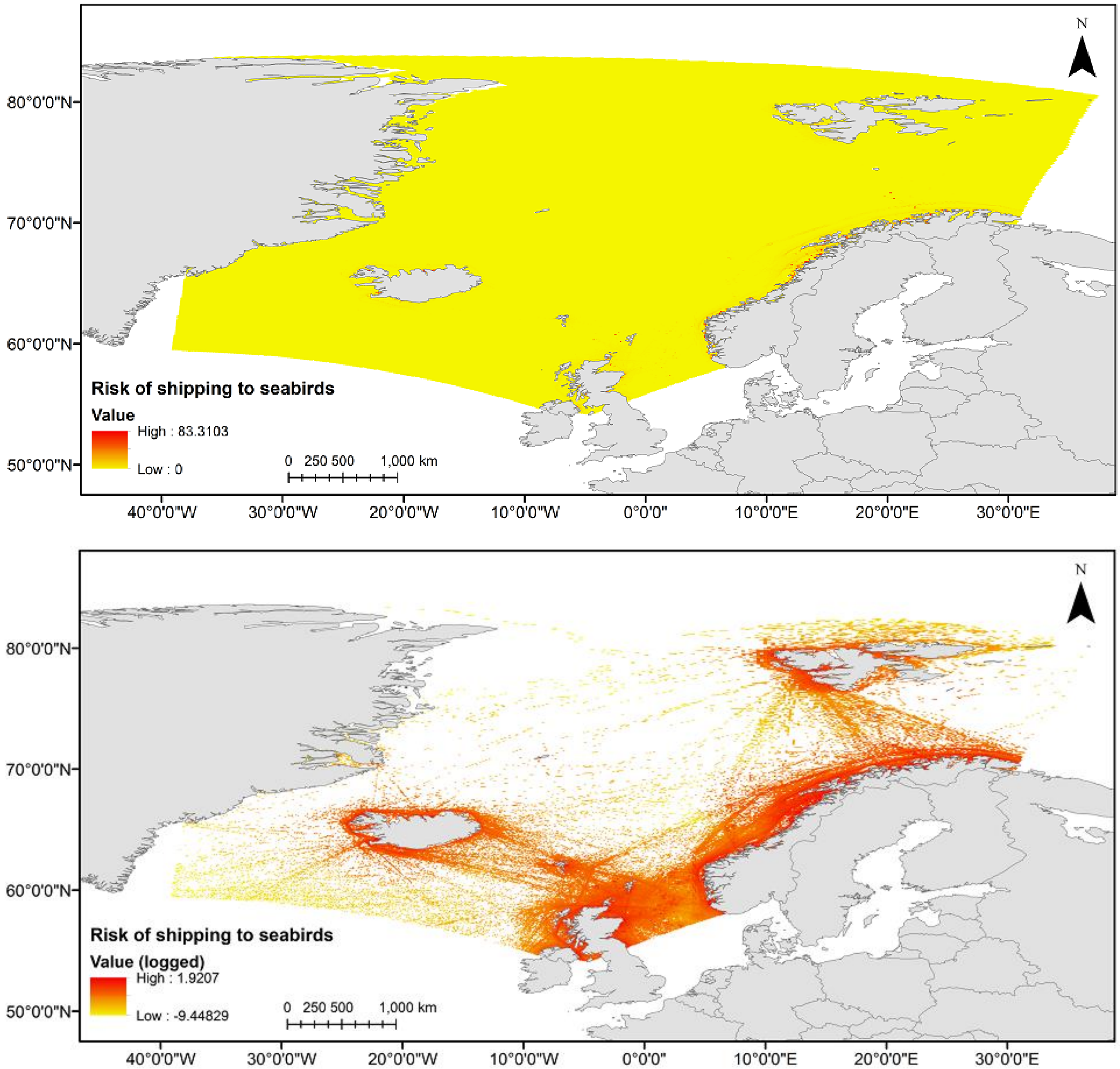

Figure 3. Risk of shipping to seabirds during the breeding season, calculated from vessel density (March to September 2018) and the spatial OVI, shown on a linear scale (a) and log scale (b).

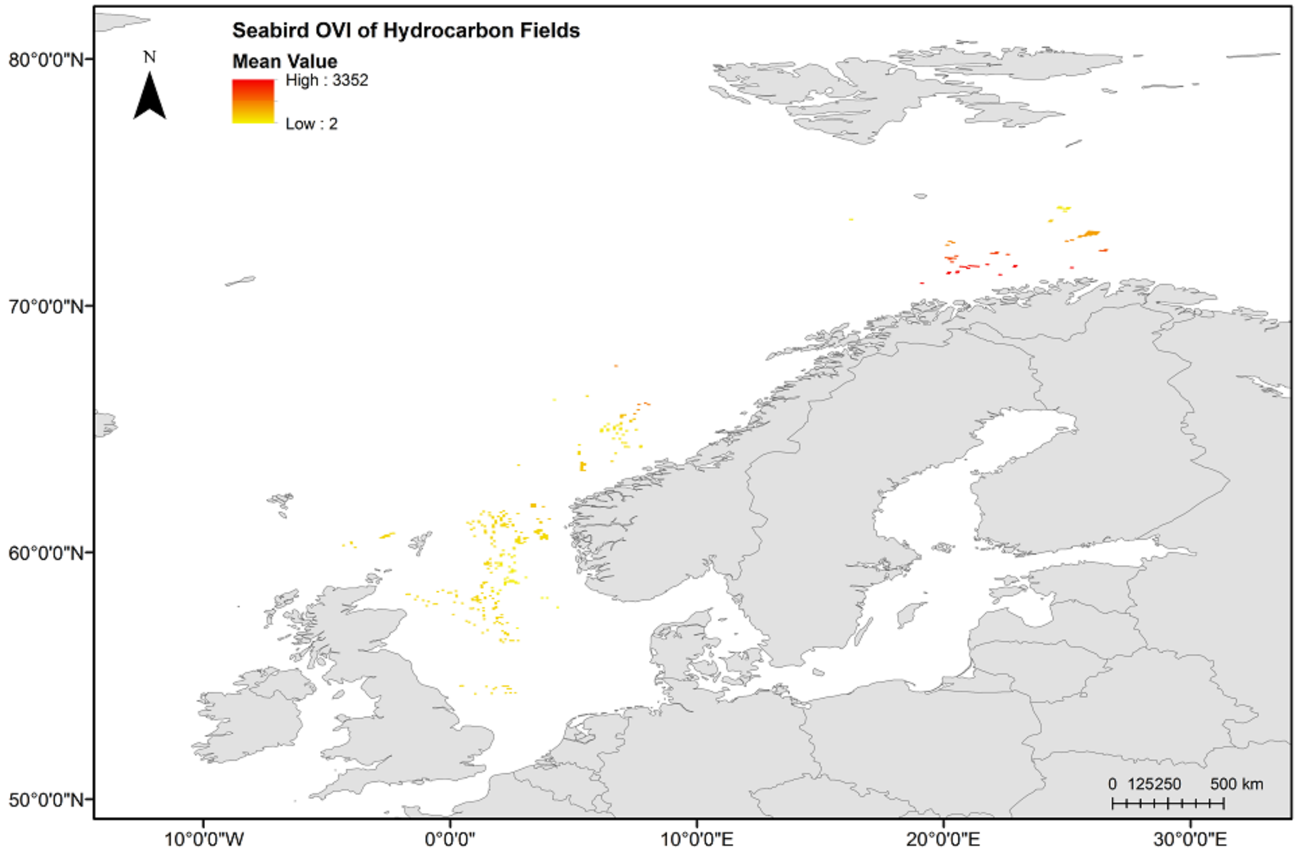

Figure 4. Spatial OVI values of all UK and Norwegian hydrocarbon extraction fields showing that seabirds may be at greater risk from oil incidences associated with extraction sites in north Norway.

Discussion

The eastern North Atlantic is an important region for seabirds, however, particularly in the northern seas of this region, increased activity associated with shipping activity and hydrocarbon extraction may result in a higher risk to these seabird populations from oil pollution. However, there are limited data for many species in this region on demography, behaviour and at-sea distribution, required to adequately assess seabird vulnerability to oil across this region. An issue shared by other data-deficient regions globally where oil pollution maybe a threat.

The first step in assessing seabird vulnerability to oil is to establish how sensitive different species are to oil pollution through calculating species-specific oil vulnerability (OVI) scores. As discussed, we opted for modifying the UK's SOSI approach to create an OVI for the eastern North Atlantic as this approach has recently been reviewed and updated to assess its usefulness in a UK context (Webb et al., 2016; O'Hanlon et al., 2020). This approach is relatively simple with eight factors contributing to species-specific OVI scores. There are numerous additional factors that could be included to assess seabird sensitivity to oil, for example: ability to withstand oiling (Burger and Gochfield, 2002), foraging behaviour (Schreiber and Burger, 2002), and at-sea aggregation behaviour (Reid et al., 2001; Stone et al., 1995). However, although incorporating all aspects of behaviour and demography that influences a species sensitivity to oil may allow an OVI to be more representative, it is unlikely that adequate data is available for many species and locations to score these factors accurately, with the danger of creating a false sense of precision in these index values. Furthermore, using a simpler approach is more straightforward to apply consistently to different regions around the world.

Even with the eight factors included in the SOSI, data to score these factors were not available for all species. The results of the sensitivity analysis was therefore useful to evaluate the quality of data needed to score these factors for less-studied species (O'Hanlon et al., 2020). The most important results from this analysis were that 1) the proportion of time spent on the sea and proportion of oiled beached corpses had a high influence on the overall OVI score therefore adequate data to score these factors is required; and 2) demography data to determine how quickly a species might recover from an oil incident had a low influence on the OVI score. However, once the spatial seabird data is incorporated into the OVI, the greatest influence on assessing where seabirds are most at risk to oil pollution is seabird density (see below). The limiting factor in assessing oil vulnerability to seabirds using this approach for any region is therefore adequate, up to date, seabird-at-sea information. Therefore, where accurate information on the proportion of time spent on the sea and proportion of oiled beached corpses is not available for a given species or location, using the best data available from substitute species or locations may be acceptable. Changing the equation to calculate the species-specific OVI score to remove proportion of oiled beached corpses (as this data is unavailable for many coastlines), had little effect on the final OVI map. To make the SOSI approach applicable to other regions globally, the main change we made was replacing the three SOSI factors relating to how vulnerable a population/species is to a single factor – a species IUCN red list category. Although this means that local conservation importance is not considered, it does mean that this OVI calculation can be used for any region globally, with OVI scores being comparable across locations.

Obtaining up to date seabird-at sea data was the biggest challenge in creating an OVI for the eastern North Atlantic. Across this region, seabird-at-sea data from vessel and aerial surveys were limited (Camphuysen, 2007; Dunn, 2012). However, even for relatively well surveyed regions such as UK waters, the coverage of vessel and aerial surveys to obtain seabird-at-sea data is variable, with limited or no data for certain locations and times of year (Webb et al., 2016). An alternative approach to only using seabird data from vessel and aerial surveys to create seabird density and distribution maps is to use species distribution models (SDM) to overcome uneven coverage in data collection (Waggitt et al., 2020). However, this approach still relies on some level of seabird-at-sea data to inform the SDMs. Tracking data can also be used to create distribution maps and provide data to SDMs (Carneiro et al., 2020), however despite the large, and increasing, amount of tracking data available, there is still limited tracking data for many species and locations, including for the eastern North Atlantic region of interest. Therefore, we opted for the foraging radius approach (Critchley et al., 2018; Grecian et al., 2012; Soanes et al., 2016) as a simpler method to estimate breeding seabird distributions for regions with limited at-sea data. Although there are limitations of using this approach it is useful where alternative seabird at-sea data are not available (Critchley et al., 2019, 2018).

We used the foraging radius approach to create predicted seabird distribution maps for 31 breeding seabird species that are widespread across the eastern North Atlantic. The species-specific OVI scores were then applied to these predicted distributions to create the final spatial OVI map, highlighting where seabirds are predicted to be most vulnerable to oil pollution across the region of interest (based on the colony size and location data obtained for the region). The strong positive correlation between the overall species density map and the spatial OVI highlights the strong influence of seabird density on identifying hotspots where seabirds are likely highly vulnerable to oil pollution.

When assessing seabird vulnerability to oil, spatial OVI maps at a monthly resolution are useful to account for temporal variability in seabird densities and distributions, and of high risks activities that may result in oil incidents. Using the foraging radius approach, we were unable to create monthly maps, as information was not available on how colony attendance and numbers may change throughout the breeding season, so instead we created a single map to cover the breeding season. One concern with creating a single map covering a breeding season is that it does not account for any spatial or temporal variation in phenology of the breeding season for different species breeding across such a large geographical region. To explore whether it was possible to account for seabirds breeding later in the north of the region (Burr et al., 2016) we obtained data on Atlantic Puffin laying dates for several colonies from across the eastern North Atlantic. We found a significant positive relationship between latitude and laying date. However, there was a lot of variation in hatching date both within colonies and among years as breeding phenology can vary, even at a local level, depending on extrinsic factors such as weather, ice extent and food availability (Burr et al., 2016). It was therefore unfeasible to try and predict how these may influence breeding phenology for any given colony or year in order to create maps at a higher temporal resolution.

There are a number of additional limitations that should be acknowledged with using the foraging radius approach to create a spatial OVI. Firstly, this approach only considers breeding adults, with no information included on the distributions of juveniles, immatures or non-breeding adults, which in some species might make up to 50% of the population (Carneiro et al., 2020). Secondly, the foraging radius approach requires data on colony size for all seabirds of interest within the region, as well as on maximum foraging distance. Therefore, the accurateness of the output from this method is influenced by the quality and confidence in the inputted seabird populations and foraging distances. Some countries have relatively good data on seabird colonies from national censuses, for example, Norway and the UK and Ireland. However, even for the UK and Ireland, current seabird data is not up to date, with the last national census carried out between 1998 and 2001. One benefit of the foraging radius approach is that it can easily be updated when more recent seabird census data is available. In this study, the colony data for Iceland and the Faroes was estimated from national population estimates and estimated colony sizes of the largest / most important colonies. Therefore, our confidence in the predicted seabird distributions around these countries is lower, than for example the UK and Norway. For many species, we have relatively good information on foraging ranges during the breeding season from tracking data (Critchley et al., 2018; Jovani et al., 2015; Oppel et al., 2018; Thaxter et al., 2012). However, for other species such as Iceland, Ivory *Pagophila eburnea* and Ross's Gulls *Rhodostethia rosea*, where we know little about their foraging strategies, we had to use the maximum foraging ranges of similar species.

The resulting spatial OVI for the 31 breeding seabird species at the large scale of the eastern North Atlantic highlights seabird hotspots (high density of seabird species with high vulnerability to oil) during the breeding season along the north coast of Norway, west Iceland and Svalbard as well as around Jan Mayan, the Faroes and northeast Scotland.

We considered two sources of oil pollution that seabirds may be at risk from in the eastern North Atlantic, hydrocarbon extraction sites and vessel density, as a proxy for the probability of oil spills from a shipping accident and intensity of chronic oil pollution (Renner and Kuletz, 2014). Combining the seabird spatial OVI data with the vessel density data revealed high risk areas, those with above-average seabird OVI scores and vessel density, at certain locations along the coasts of Norway and Iceland, as well as to a lesser extent northeast Scotland and Svalbard. Log scaling vessel density risk map revels larger areas where seabirds are at intermediate and low risk to shipping related oil pollution at this regional scale. This includes most coastal areas across the region where many seabirds will likely be present during the breeding season. Such maps can be useful to identity areas were mitigation should be put in place to ensure incidents do not occur at these locations during the breeding season, minimising any risk to seabirds. We used vessel density from 2018 to explore where areas of high vessel density and high seabird vulnerability to oil overlap. Although there is likely some level of spatial variation across years, vessel density was particularly high off the coasts of North Scotland and Norway which are main marine shipping routes and therefore are likely to be areas with high vessel density across years (Rodrigue, 2020).

We focused here on the breeding season, however, it is important to also consider the non-breeding season, when individuals are no longer constrained to their breeding colonies, and many species are more pelagic in their distribution (Fayet et al., 2017; Frederiksen et al., 2012). Certain species / individuals may also be at greater risk to oil pollution during the non-breeding season, for example, auk species where chicks fledge before being able to fly, or have periods of flightless moult (Harris and Wanless, 1990). For some seabird species in the eastern North Atlantic, tracking data collected through projects such as SEATRACK (http://www.seapop.no/en/seatrack/) will be very useful in future to assess non-breeding distributions.

Within this study we have focused on highlighting the feasibility of creating an assessment of seabird vulnerability to oil pollution during the breeding season across a large geographical area, where data availability of seabird demography and behaviour is variable, and where at-sea distribution data is particularly lacking. The resulting output provides an indication of where seabirds are likely to be most vulnerable to oil pollution during the breeding season at the scale of the eastern Northern North Atlantic, an area where shipping and hydroextraction activity is likely to increase in future. Caution should be employed when using this map given the caveats discussed, especially given the age of much of the colony data used to create the maps, especially given recent declines in seabird populations in recent years in the North Atlantic. However, despite these limitations, the map reflects the relative vulnerability to oil of seabirds across this region, given what data is currently available. This approach could be used for other locations where data on at-sea distributions are limiting, but where some level of colony size and location information exists. Given the influence of the factors used to score the OVI, this approach can also be used to calculate OVI scores for less studied species, instead using information from similar species or expert opinion to score each factor. We used this approach to establish where seabirds are most vulnerable to oil pollution but it could be used to explore other potential anthropogenic threats such as over-fishing and marine renewable installations (Certain et al., 2015; Garthe and Hüppop, 2004).

The most important priority for assessing the risk of any anthropogenic threat to seabird is high quality seabird-at-sea data, whether obtained from vessel and arieal surveys or from tracking data, used in isolation or in combination with SDMs. This data should include good coverage of different stages through the annual cycle and ideally include information on juveniles, immatures and non-breeding adults. Without this type of data it is impossible to accurately predict where, and when, seabirds will be most at risk from anthropogneic threats, to ensure adequate mitigation is put in place.

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