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Climate change could disrupt migratory patterns for an Arctic seabird population

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ABSTRACT: Climate change is altering the marine environment at a global scale, and these changes could affect the distribution and migration patterns of marine species throughout their annual cycle. Arctic regions are already experiencing some of the most dramatic changes in marine climate, and there is a need for predictive models to understand how these changes could alter the spatio-temporal distributions of Arctic marine species. We used a species distribution model to predict potential future changes in the non-breeding distribution of thick-billed murres *Uria lomvia* from a colony in Hudson Bay, Canada, from 2021 to 2100 using 3 Coupled Model Intercomparison Project Phase 6 (CMIP6) climate scenarios: low (SSP1-2.6), intermediate (SSP2-4.5), and high (SSP5-8.5) emissions. Under the intermediate- and high-emissions scenarios, suitable habitat within Hudson Bay would become available year-round during the next century. This could lead to a portion of this migratory population becoming year-round residents within the next 80 yr. We predicted a significant northward shift in the winter range, such that little or no habitat would be available below 55° N by 2100. This shift would have significant implications for the murre harvest in Canada because the winter distribution would no longer include coastal Newfoundland where most harvesting occurs, particularly if murres from other colonies show a similar shift in distribution. Although there were projected changes in seasonal distributions under all 3 climate scenarios, dramatic re-distribution of non-breeding habitat could be avoided with policies that limit future emissions.

KEY WORDS: Thick-billed murre · *Uria lomvia* · Hudson Bay Complex · Northwest Atlantic · Species distribution model

1. INTRODUCTION

Arctic marine environments are experiencing significant effects of anthropogenic climate change in the form of increased ocean and air temperatures and declining sea ice cover (Meredith et al. 2019, Rantanen et al. 2022). Polar marine predators are already changing their distributions and migratory patterns in response to marine climate change (Cherry et al. 2013, Patterson et al. 2021, Shuert et al. 2022, Stafford et al. 2022, Green et al. 2023). Declining sea ice cover has direct impacts on physical habitat, especially for air-breathing species (seabirds and marine mammals) (Laidre et al. 2008, Tynan et al. 2009), and also drives ecosystem level changes because spring ice determines the phenology of the spring bloom and peak productivity to support upper trophic level species (Both et al. 2009, Post et al. 2013). Warming ocean temperatures have facilitated the movement of subarctic and temperate fish species into Arctic regions (Fossheim et al. 2015), with consequences for diet, energetics, and reproductive success of marine predators (Piatt et al. 2020, Will et al. 2020, Descamps & Ramírez 2021). Seabirds are important indictors of

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the effects of climate change on marine ecosystems (Grémillet & Boulinier 2009, Sydeman et al. 2012, Grémillet & Descamps 2023). For example, a prolonged marine heat wave in the North Pacific has been linked to mass mortality of 4 million common murres *Uria aalge*, through a combination of bottom-up and top-down ecosystem changes leading to mass starvation (Piatt et al. 2020, Renner et al. 2024). Although the effects of climate change on seabirds in polar regions are already being noted, including changes in breeding phenology (Moe et al. 2009, Whelan et al. 2022) and shifting distributions (Krüger et al. 2018, Kuletz et al. 2020, Patterson et al. 2021), we still lack considerable predictive capacity for estimating the long-term impacts on spatial distributions of species under various climate change scenarios.

The potential effects of climate change on migratory birds are complex. High mobility makes it possible to track changing environmental conditions; however, migratory species also require suitable habitat in multiple locations for different stages of their annual cycle (Robinson et al. 2009, Wauchope et al. 2017). Thus, changes in habitat quality or quantity within any part of the annual range could have consequences for survival and productivity. Declining sea ice cover within Arctic marine environments could dramatically alter migratory behaviour and phenology. In the most extreme case, species that currently migrate away from regions with seasonal ice cover could become year-round residents within their breeding ranges (Clairbaux et al. 2019, 2021a). We assessed climate change effects on spatio-temporal distributions of a seabird, the thick-billed murre *U. lomvia* (hereafter murre), considered an important indicator species for Arctic ecosystems (Circumpolar Seabird Working Group 1996). Specifically, we focused on a population that breeds within Hudson Bay, Nunavut, Canada, which is a sentinel region for ecosystem changes in the Arctic (Hoover et al. 2013a, Florko et al. 2021).

Thick-billed murres are a circumpolar species that breeds at dense colonies on sea-cliffs in the Arctic and sub-Arctic (Gaston & Hipfner 2020). Murres have life history traits that constrain their pre- and post-breeding distribution to regions close to existing colonies, which limits their ability to respond to changing environmental conditions. Murres are adapted to cold-water environments, but require open water for foraging (Gaston & Nettleship 1981, Eby et al. 2023). Therefore, sea-ice dynamics are an important driver of migration patterns and seasonal distributions in this species because seasonal ice-cover forces murres from most colonies to migrate to areas with open water over winter. Murres have high natal and breeding philopatry (Gaston et al. 1994, Steiner & Gaston 2005); therefore, it is unlikely that murres will be able to respond to rapid climate change by establishing new colonies, at least in the short term. Immediately after breeding, murres undergo a flightless moult in September and October, when successful males care for flightless young (Elliott & Gaston 2014, Burke et al. 2015, Elliott et al. 2017). Prior to breeding, murres must re-establish their nest site and reconnect with their mate, and females need to build body reserves to produce an egg (Gaston et al. 2005, Bennett et al. 2022a). Moreover, murres have the highest flight costs recorded among birds (Elliott et al. 2013), limiting their ability to migrate farther from central breeding colonies in response to climate change (Watanabe 2016). Population trends in thick-billed and common murres have been linked to environmental conditions in the non-breeding period (Gaston 2003, Descamps et al. 2013, Frederiksen et al. 2016, 2021).

Murres are hunted on their wintering grounds in Canada, Greenland, and Iceland (Falk & Durinck 1992, Gaston & Robertson 2010, Frederiksen et al. 2019). Changes in the non-breeding distribution of murres could impact the availability of birds for harvest within these regions. Hunting across both regions was estimated to cause annual population declines of 0.006–0.022% (Frederiksen et al. 2019, Cox et al. 2024); therefore, changes in distribution that increase or decrease exposure to hunting mortality could impact growth rates for affected populations. Population trends in thick-billed and common murres have already been linked to environmental conditions in the non-breeding period (Gaston 2003, Descamps et al. 2013, Frederiksen et al. 2016, 2021).

Hudson Bay is a large enclosed subarctic sea that is already experiencing physical and ecological effects of climate change (Hochheim & Barber 2010, Hochheim et al. 2011, Macdonald & Kuzyk 2011, Gaston et al. 2012b, Hoover et al. 2013a,b, Florko et al. 2021). While there is considerable work on the effect of climate change on seabirds and related communities in the North Atlantic (Sandvik et al. 2005, Amélineau et al. 2018, Clairbaux et al. 2019, 2021a), Hudson Bay is unique for understanding changes in migration as it is the only enclosed sea that is connected to other water bodies solely by a northern passage (Hudson Strait). Trends in declining sea ice cover within Hudson Bay could alter migration patterns of seabirds and marine mammals that currently migrate out of the Bay as sea ice forms each fall. The forage fish community is also undergoing 'Atlantification', with temperate species (e.g. capelin *Mallotus villosus*) replacing cold water species (e.g. Arctic cod *Boreogadus saida*) (Hop &

Gjøsæter 2013, Fossheim et al. 2015, Florko et al. 2021). These community changes have also been observed in the diet composition for murres breeding in Hudson Bay (Gaston et al. 2003, Provencher et al. 2012). This change in diet has been linked to sudden declines in ice cover within Hudson Bay in the mid-1990s and has also been associated with a decline in nestling mass at colony departure (Gaston et al. 2012b). A similar shift in diet has also occurred in ringed seals *Pusa hispida* within Hudson Bay (Young & Ferguson 2014).

A previous species distribution model (SDM) successfully identified environmental drivers of the non-breeding distribution of thick-billed murres from the Coats Island colony in Hudson Bay and predicted past distributions to understand ongoing impacts of global warming across the range of this population (Patterson et al. 2021). The Coats Island colony is one of the longest consistently monitored wildlife populations in Canada, and research at this site has contributed significantly to our understanding of ecological changes in Arctic marine ecosystems (Patterson et al. 2024). This colony is relatively small (30 000 breeding pairs); however, other nearby colonies within Hudson Bay and Hudson Strait account for almost 1 000 000 breeding pairs (Gaston et al. 2012a). In the current study, we examine future changes in the non-breeding distributions of this population under 3 Coupled Model Intercomparison Project Phase 6 (CMIP6) climate change scenarios: low (Shared Socio-economic Pathway [SSP] 1-2.6), intermediate (SSP2-4.5), and high (SSP5-8.5) emissions. Our aims were to determine if climate change is likely to alter the distribution of non-breeding habitat for this population and to identify which stages of the non-breeding cycle and portions of the range are most susceptible to habitat changes.

2. METHODS

2.1. SDM

We used a previously published SDM to predict non-breeding distributions of thick-billed murres under different climate change scenarios (Patterson et al. 2021; Fig. 1). The model was developed using geolocator tracks from 91 adult thick-billed murres tracked from a breeding colony on Coats Island, Nunavut (62.95° N, 82.01° W), collected over 4 annual cycles (2007–2009, 2017–2019) (Patterson et al. 2021). Full details on geolocator unit deployment and location estimates are outlined in Patterson et al. (2021). Location estimates were derived from light-level data using the R packages 'TwGeos' (Lisovski et al. 2016) and 'probGLS' (Merkel et al. 2016). Pseudo-absences representing habitat available to murres were randomly generated for ocean areas within a 200–1000 km envelope from all observed locations within the month of tracking at a 1:1 ratio of pseudo-absences to observations. Environmental conditions at observed locations were compared to environmental conditions at pseudo-absence locations paired by date. The model used daily time-varying environmental variables for sea surface temperature (SST; NOAA high-resolution SST, NOAA/OAR/ESRL PSL, https://psl.noaa.gov/), air temperature and wind speed (NOAA Physical Sciences Laboratory NCEP/NCAR Reanalysis 1, https://psl.noaa.gov/data/gridded/data.ncep.reanalysis.pressure.html), and sea ice concentration (NOAA high-resolution ice cover, NOAA/OAR/ESRL PSL, https://psl.noaa.gov/), as well as fixed predictors for bathymetry (ETOPO1

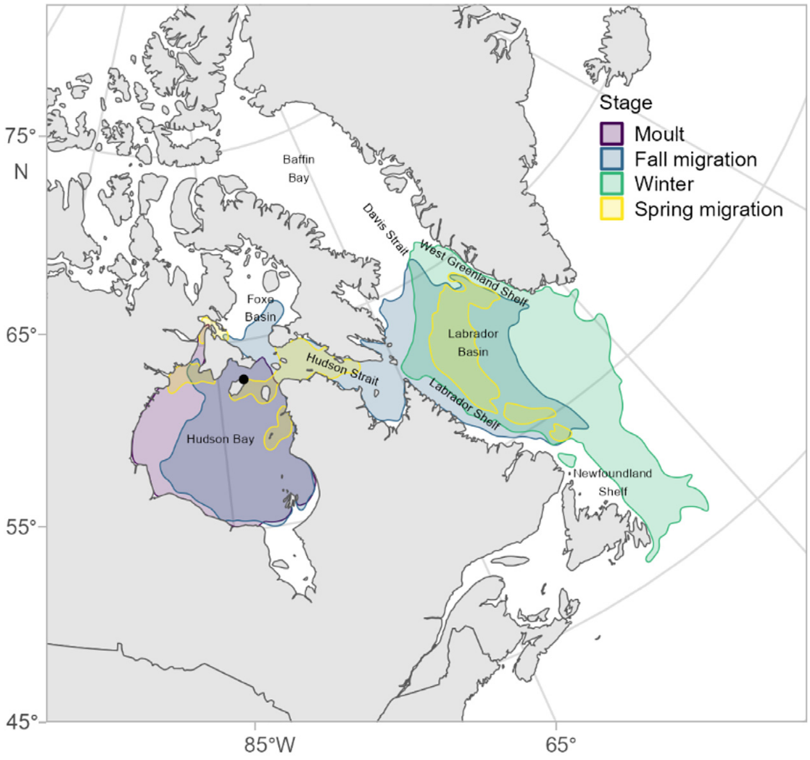

Fig. 1. Study area, showing the location of the thick-billed murre breeding colony at Coats Island, Nunavut, Canada (black point) and predicted current distribution (1982–2014) by stage of the annual cycle using the species distribution model from Patterson et al. (2021)

Global Relief Model, www.ngdc.noaa.gov/mgg/global/), slope, distance from colony, and day of year (DOY). All predictor variables were standardized to a common 0.25° × 0.25° raster resolution. The SDM was fit with a random forest, using the 'ranger' package in R (Wright & Ziegler 2017), to estimate probability of occurrence as a function of environmental conditions throughout the non-breeding season (1 September to 31 May). Distance from colony and SST had the greatest influence on murre distributions (see Fig. 2; Patterson et al. 2021). From this fitted model, we predicted expected distributions under different future environmental conditions (see Section 2.3). A complete description of original SDM development is provided in Patterson et al. (2021).

2.2. Identifying adequate climate change projection models

All models participating in CMIP6 run a historical experiment using observed values for greenhouse gasses and other climate drivers over the period 1850–2014 (Eyring et al. 2016). These historical simulations are used to assess how well models simulate climate, document model characteristics, and ensure continuity across phases of CMIP (Eyring et al. 2016, O'Neill et al. 2016). We used the historical simulations from 9 candidate models to determine which models gave credible historical distributions relative to our SDM predictions from the period for which remote sensing observations are available (1982–2014). We considered these 9 CMIP6 models: ACCESS-CM2 (Dix et al. 2019), CanESM5 (Swart et al. 2019), CMCC (Lovato & Peano 2020a), EC-Earth3 (EC-Earth Consortium 2019), MIROC6 (Tatebe & Watanabe 2018), MPI-ESM (Wieners et al. 2019d), MRI-ESM2-0 (Yukimoto et al. 2019a), NorESM2-LM (Seland et al. 2019), and NorESM2-MM (Bentsen et al. 2019). These models were chosen because each predicted 4 relevant environmental variables (SST, air temperature, sea ice concentration, windspeed) at a daily frequency at a nominal spatial resolution of 100 km. CMIP6 data were obtained from https://esgf-node.llnl.gov/search/cmip6/.

We used the SDM described in Section 2.1 to make predictions from 1982 to 2014 using the historical simulations from each model as inputs for SST, air temperature, sea ice concentration, and windspeed. SDM predictions were made at 7 d intervals for the non-breeding portion of the annual cycle (1 September to 31 May). Within each stage of the annual cycle (moult: DOY 245–307; fall migration: DOY 308–362; winter: DOY 363–88; spring migration: DOY 89–152) we calculated the median probability of use for each cell with predictions made from remote sensing observations and predictions made from the 9 climate models. We used Schoener's *D* (Schoener 1968, Warren et al. 2008) to compare the similarity between predictions based on remotely sensed data and each climate model. Schoener's *D* was calculated as:

$$D = 1 - 0.5 \times \Sigma(|P_1 - P_2|) \quad (1)$$

where P_1 is the predicted probability of use from remotely sensed historical data and P_2 is the predicted probability of use from the historical experiments of climate models. This statistic can take values between 0 (no overlap) and 1 (perfect overlap). Only raster cells with a value ≥0.5 for P_1 or P_2 were included in comparisons calculations. Cells with low predicted values in both the observed and predicted data were excluded to avoid having areas with a low probability of use strongly influencing the assessment; however, model rankings were not sensitive to this threshold (range tested: 0.3–0.7). For each CMIP6 model, we calculated mean Schoener's *D* across the 4 annual cycle stages. Models with mean Schoener's *D* greater than 0.90 were considered to adequately predict historical distributions. Predictions from models that met this criterion were averaged to create a composite model. Schoener's *D* from the composite model was compared to individual models to determine whether the composite model provided better consistency with predictions from remote sensing data.

2.3. Predicting future distributions under climate change scenarios

The CMIP6 provides predictions of environmental change under a range of predicted future climate scenarios, known as the Shared Socio-economic Pathways (SSPs) (O'Neill et al. 2016). SSP1-2.6 encompasses the low end of future emissions considered within CMIP6 scenarios (O'Neill et al. 2016); this scenario assumes low challenges to climate mitigation and adaptation (O'Neill et al. 2017, Riahi et al. 2017). SSP2-4.5 is the 'Middle of the Road' pathway assuming intermediate challenges to mitigation and adaptation, consistent with historical trends observed over the past century (O'Neill et al. 2017, Riahi et al. 2017). Finally, SSP5-8.5 encompasses the high end of future emissions (O'Neill et al. 2016). This scenario of high future emissions is only feasible under SSP5, the 'Fossil-fueled Development' pathway, which assumes rapid growth of the global economy coupled with intensive fossil fuel development, high challenges to climate mitigation, and low challenges to climate adaptation (O'Neill et al. 2017, Riahi et al. 2017). Data from the SSP1-2.6, SSP2-4.5, and SSP5-8.5 experiments were used to predict potential future distributions using the climate model with the highest Schoener's *D*. Predictions were made at 7 d intervals for the non-breeding portion of the annual cycle (1 September to 31 May) for every year from 2020 to 2100.

To visualize how habitat changes under different scenarios affect the expected spatial distribution of murres, we mapped the predicted probability of use under the 3 emission scenarios in 2020, 2060, and 2100 for the weeks of 1 October, 1 December, 15 February, and 1 May, respectively. These dates represent the middle of key stages of the murre annual cycle: moult, fall migration, winter, and spring migration (Patterson et al. 2021).

2.4. Predicted changes in habitat areas within marine regions and range extents

Following Patterson et al. (2021), we used a predicted probability of use ≥0.7 as the threshold for defining suitable habitat. This threshold corresponds to areas where 90% of observed murre locations fell in the original SDM. Murres from Coats Island primarily occur within 3 Canadian Marine Bioregions: Hudson Bay Complex (HBC), Eastern Arctic (EA), and Newfoundland—Labrador Shelves (NLS) (DFO 2009). Shapefiles of these marine bioregions were obtained from https://open.canada.ca/data/en/dataset/23eb8b56-dac8-4efc-be7c-b8fa11ba62e9. We calculated the area of suitable habitat within each of these regions for each week of the non-breeding period (September to May) from 2020 to 2100. Area calculations were made in the Canadian Albers projection using the 'raster' package (Hijmans & Van Etten 2016). We calculated the western and southern range extents based on the lower fifth quantile of longitude (°E) and latitude (°N) of all raster cells across the entire predicted distribution. Range extents were calculated weekly for each raster for the years 2020 to 2100.

2.5. Statistical analysis

We used general additive models (GAMs) to examine how suitable habitat areas and the western and southern range extents changed as a function of DOY, year, and climate scenario. DOY was centered on 1 January, such that dates from the preceding fall (e.g. 1 September to 31 December) had DOY <0 and dates after 1 January (1 January to 31 May) had values >0. All GAMs were fit with a Gaussian distribution. Models for suitable habitat area used a log-link function and models for range extents used an identity link function. Models included tensor smoothers fit with a cubic regression spline to model an interaction between DOY and year, grouped by climate scenario. We included autoregressive correlation structures to account for lack of independence among consecutive measurements; we considered autoregressive orders between 1 and 3 and used Akaike's information criterion to determine the best order for each model. We ran GAMs using the 'mgcv' package in R (Wood 2011). We used the 'gam.check' function to assess model fit and determine the minimum number of knots (*k*) required in model fits. We report decadal trends in habitat area, seasonal patterns, and range extents based on the mean ± SD of predicted values from the GAMs.

3. RESULTS

3.1. Identifying adequate climate change projection models

Predictions from 4 of the historical CMIP6 models considered were highly consistent (mean Schoener's *D* >0.90; Table 1) with predictions based on remote sensing observations. A composite model based on the average predictions from these 4 top models had comparable or better overlap with predictions modelled using remotely sensed observations within each non-breeding stage, and higher Schoener's *D* across all stages. We used predictions from this composite model in all subsequent analyses of the 3 climate change scenarios. The 4 climate models that contributed to the composite model used in climate scenario analysis were MPI-ESM1-2-LR (Wieners et al. 2019a,b,c), CMCC-CM2-SR5 (Lovato & Peano 2020b,c,d), MRI-ESM2-0 (Yukimoto et al. 2019b,c,d), and MIROC6 (Shiogama et al. 2019a,b,c).

3.2. Predicting future distributions under climate change

Predicted distributions are expected to change between 2020 and 2100 for all stages of the non-breeding period; changes under the low (SSP1-2.6) and intermediate (SSP2-4.5) scenarios tend to be

Table 1. Seasonal model comparisons for predicted non-breeding habitat for thick-billed murres from Coats Island, Nunavut, Canada, for the period 1982–2015. Schoener's *D* was used to estimate consistency between predictions based on remote sensing data and predictions based on the historical climate scenario for 9 different CMIP6 climate models. Model performance was ranked based on average Schoener's *D* across 4 non-breeding stages. A composite model (**bold**) was developed by averaging climate model predictions across the 4 CMIP6 models with the highest overlap (*italics*)

Climate model	Moult (Sep–Oct)	Fall migration (Nov–Dec)	Winter (Jan–Mar)	Spring migration (Apr–May)	Average Schoener's *D*
Composite model	**0.99**	**0.95**	**0.94**	**0.94**	**0.95**
MPI-ESM1-2-LR	*0.98*	*0.93*	*0.93*	*0.94*	*0.95*
CMCC-CM2-SR5	*0.99*	*0.94*	*0.93*	*0.92*	*0.94*
MRI-ESM2-0	*0.98*	*0.96*	*0.92*	*0.91*	*0.94*
MIROC6	*0.98*	*0.95*	*0.91*	*0.88*	*0.93*
NorESM2-LM	0.96	0.91	0.86	0.88	0.90
NorESM2-MM	0.96	0.88	0.86	0.89	0.90
EC-Earth3	0.98	0.95	0.82	0.83	0.89
ACCESS-CM2	0.90	0.97	0.88	0.81	0.89
CanESM5	0.98	0.95	0.72	0.64	0.82

more modest and restricted to certain times of year, while changes under the high-emission scenario are large-scale and occur across all stages. During the post-breeding moult (Fig. 2), when murres are flightless and thus unable to respond quickly to changes in habitat, their distribution from Coats Island is almost entirely confined to the HBC. Under all climate scenarios, regions of southern Hudson Bay are predicted to become less suitable for murres over the next 80 yr and the overall quantity of highly suitable habitat (probability of use ≥0.9) declines. This pattern is most pronounced under the high-emission scenario (SSP5-8.5). During fall migration (Fig. 3), when murres are migrating east into the Northwest Atlantic, the 2020 distribution includes Hudson Bay, Hudson Strait, Davis Strait, and the Northern Labrador Shelf. Under the intermediate (SSP2-4.5) and high-emission scenarios (SSP5-8.5), habitat suitability declines within Hudson Bay and predicted habitat in the Northwest Atlantic shifts north from Davis Strait into Baffin Bay. During winter, when murre distributions are most constrained by the presence of pack ice, 2020 distributions include Davis Strait, Labrador Basin, NLS, and the West Greenland Shelf (Fig. 4). Winter distributions are largely unchanged under the low and intermediate emission scenarios; however, under the high-emission scenario, the winter range changes substantially, including year-round occupation of Hudson Bay, northern expansion into Baffin Bay, and southern retraction from the Newfoundland and southern Labrador Shelves. During spring migration (Fig. 5), when murres are moving back towards breeding areas as sea ice recedes, the predicted distribution includes the NLS and any open water areas within Hudson Bay and Hudson Strait. The intermediate and high-emission scenarios both show increasing habitat within Hudson Bay over time, with declining use of the Labrador Shelf in the high-emission scenario.

3.3. Predicted changes in habitat areas within marine regions and range extents

3.3.1. HBC

Under current conditions, predicted suitable habitat area in the HBC is high from September through November and then declines rapidly in December. No suitable habitat is available in the HBC from February to the end of March (when the Bay is covered by ice); thereafter, suitable habitat area increases rapidly in April and May (Fig. 6; Fig. S1 in the Supplement at www.int-res.com/articles/suppl/m757p181_supp.pdf). Under SSP1-2.6 and SSP2-4.5 there are no major changes in this seasonal pattern; however, there is a decline in the amount of predicted habitat within the HBC during fall. Under SSP5-8.5 the seasonal pattern in suitable habitat availability within HBC is predicted to flip, with declining suitable habitat area in fall and spring concomitant with increasing suitable habitat area through winter. This switch in suitable habitat seasonality is most evident after 2080 (Fig. 6).

3.3.2. NLS

In 2020, predicted suitable habitat area within the NLS increases rapidly in late November and then

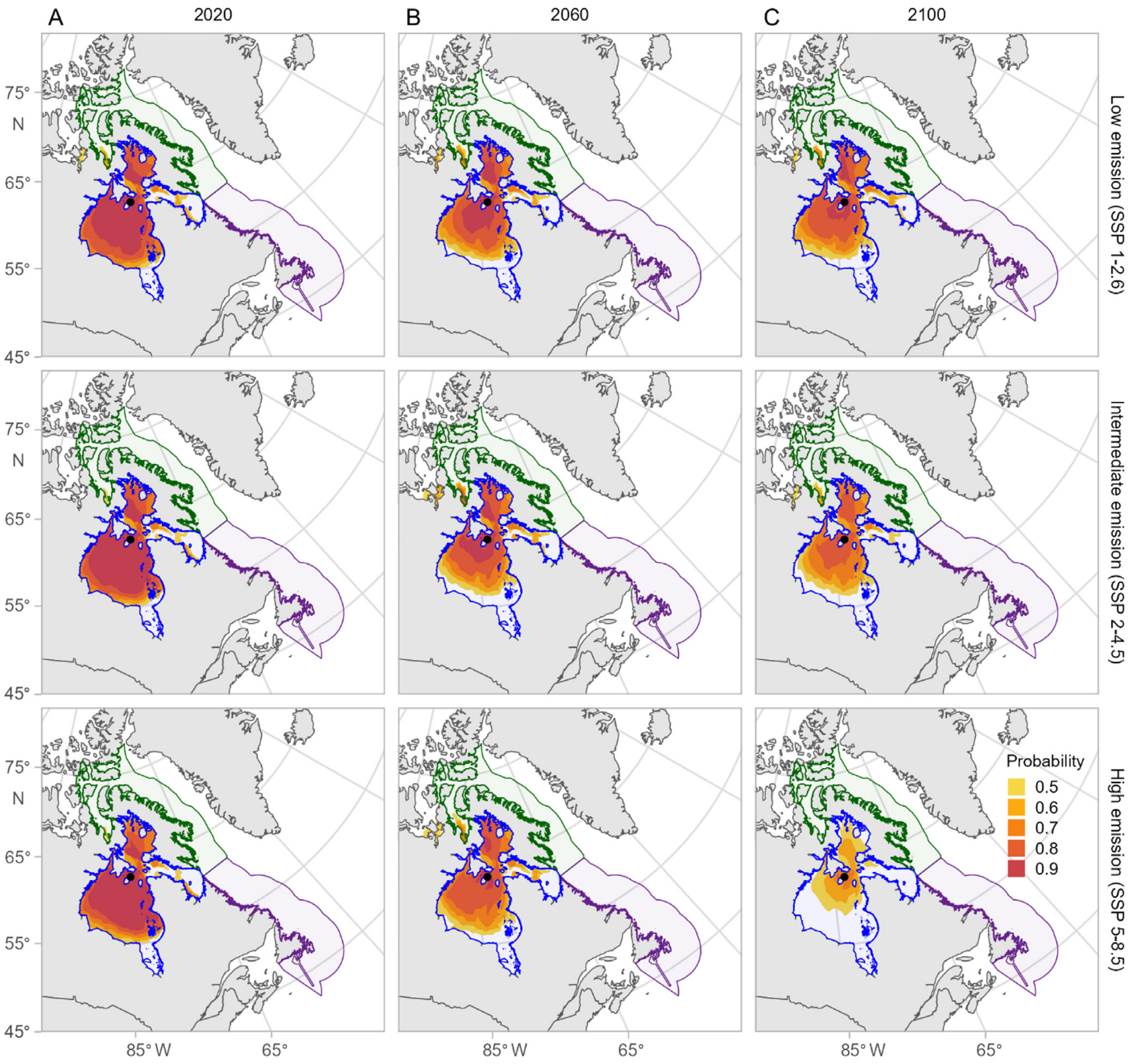

Fig. 2. Predicted distributions of thick-billed murres from the Coats Island, Nunavut, Canada, breeding colony, during post-breeding moult (1 October) in (A) 2020, (B) 2060, and (C) 2100 under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) scenarios. The location of the breeding colony is shown with a black point. Polygon outlines indicate the Hudson Bay Complex (HBC, blue), Newfoundland and Labrador Shelves (NLS, purple), and Eastern Arctic (EA, green) marine bioregions

declines again in late April (Fig. 6; Fig. S1). All 3 scenarios predicted declines in the amount of suitable habitat area available within the NLS. Declines under the SSP1-2.6 and SSP2-4.5 scenarios are relatively modest compared to the changes under SSP5-8.5. Under SSP5-8.5, predicted suitable habitat area becomes available in the NLS later in winter, declines earlier in spring, and the maximum suitable habitat area extent is lower overall.

3.3.3. EA

Predicted suitable habitat area within the EA, under current conditions, reaches a peak at the end of November and then declines to a minimum in April (Fig. 6; Fig. S1). Under SSP1-2.6 and SSP2-4.5, there are no major changes in suitable habitat seasonality or the peak in predicted suitable habitat area availability. Under SSP5-8.5, the seasonal increase in predicted suitable

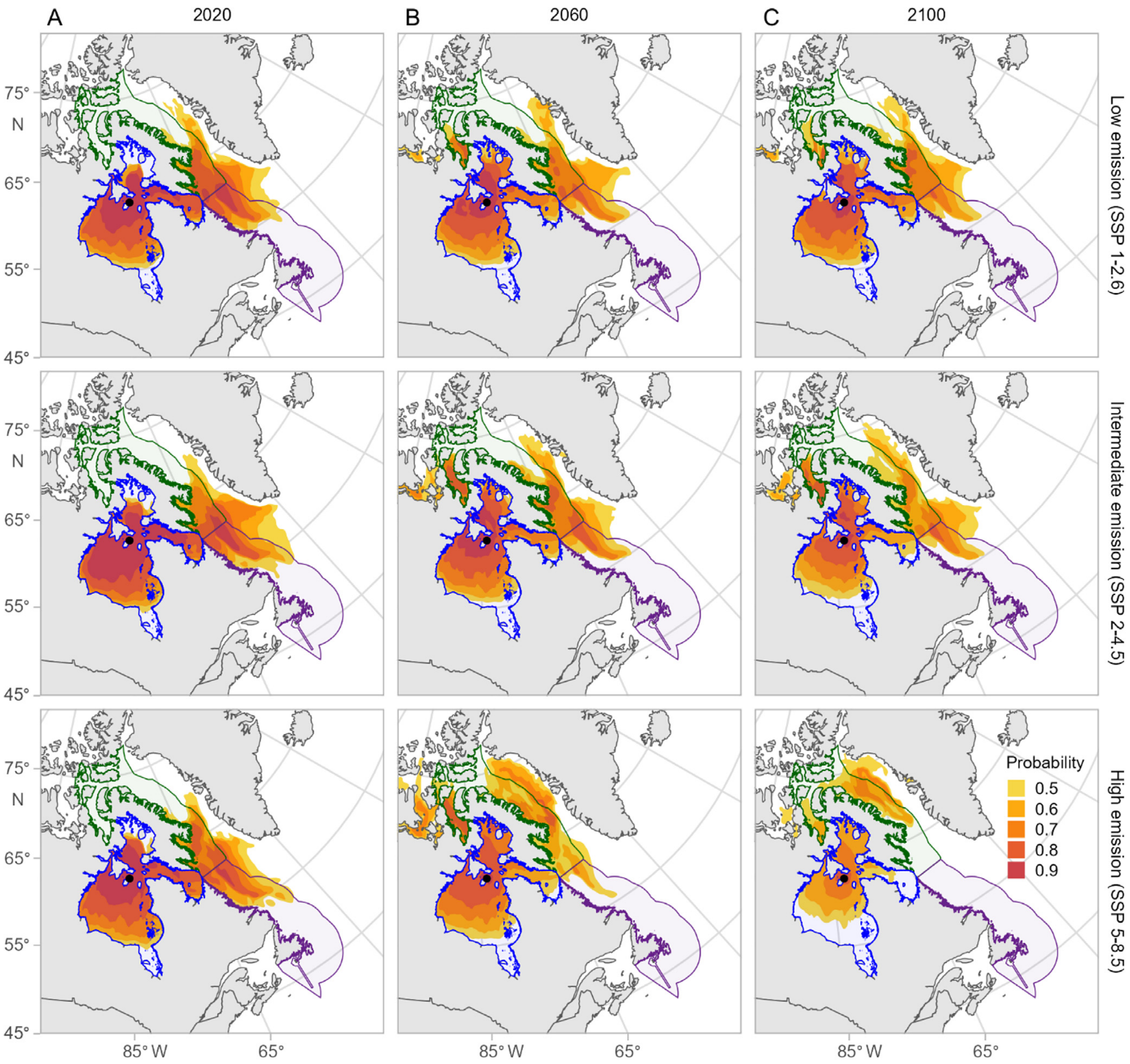

Fig. 3. Predicted distributions of thick-billed murres from the Coats Island, Nunavut, Canada, breeding colony, during fall migration (1 December) in (A) 2020, (B) 2060, and (C) 2100 under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) scenarios. Other details as in Fig. 2

habitat within the EA occurs later in winter and suitable habitat is available for longer, through late winter.

3.3.4. Western range extent

In 2020, the western range boundary is located around 90° W, at the western edge of Hudson Bay, from September to November (Fig. 7; Fig. S2). The boundary then shifts east as murres migrate to the northwest Atlantic for winter, reaching its easternmost position around 60° W, along the Labrador Shelf, in February and March. The western extent shifts back to 90° W in May as murres migrate back into Hudson Bay. Under SSP1-2.6 and SSP2-4.5, there is no change in the location of the western extent between 2020 and 2100, although under SSP2-4.5, the number of days when the range is east of 70° W declines (2020s: 96.5 ± 1.3 d; 2090s: 68.4 ± 1.0 d). Under SSP5-8.5, there is a strong decline in the eastward migration of the predicted range in winter, particularly after 2075. From 2020 to 2080, the number

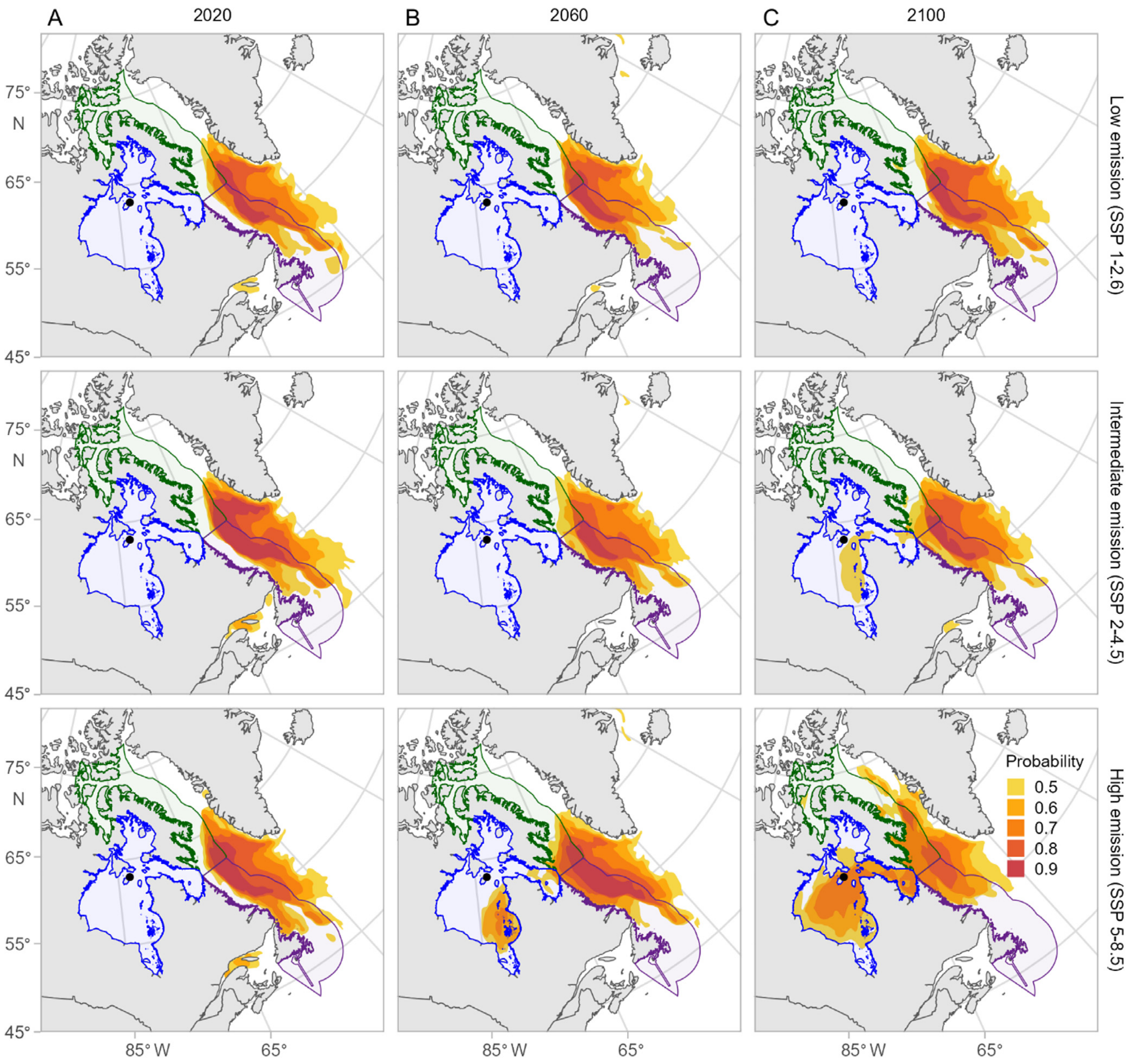

Fig. 4. Predicted distributions of thick-billed murres from the Coats Island, Nunavut, Canada, breeding colony, during winter (15 February) in (A) 2020, (B) 2060, and (C) 2100 under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) scenarios. Other details as in Fig. 2

of days when the range is east of 70° W declines markedly (2020s: 95.5 ± 2.3 d; 2080s: 12.7 ± 12.4 d). After 2085, the western extent no longer shifts east of 70° W because suitable habitat is predicted to be available within the HBC throughout winter.

3.3.5. Southern range extent

In 2020, the predicted southern range boundary is located at 57° N from September through November, then moves south to reach 48° N at the end of March before shifting back north to reach 58° N at the end of May (Fig. 7; Fig. S2). Under SSP1-2.6, there are no major changes in the seasonal trend or the location of the southern extent. Under SSP2-4.5 and SSP5-8.5, there is a consistent northward shift in the location of the southern boundary throughout the year. The magnitude of this shift is greater for SSP 5-8.5, with the minimum southern extent shifting north by 5.4° (2020s: 49.9 ± 0.3° N; 2090s: 55.3 ± 0.2° N).

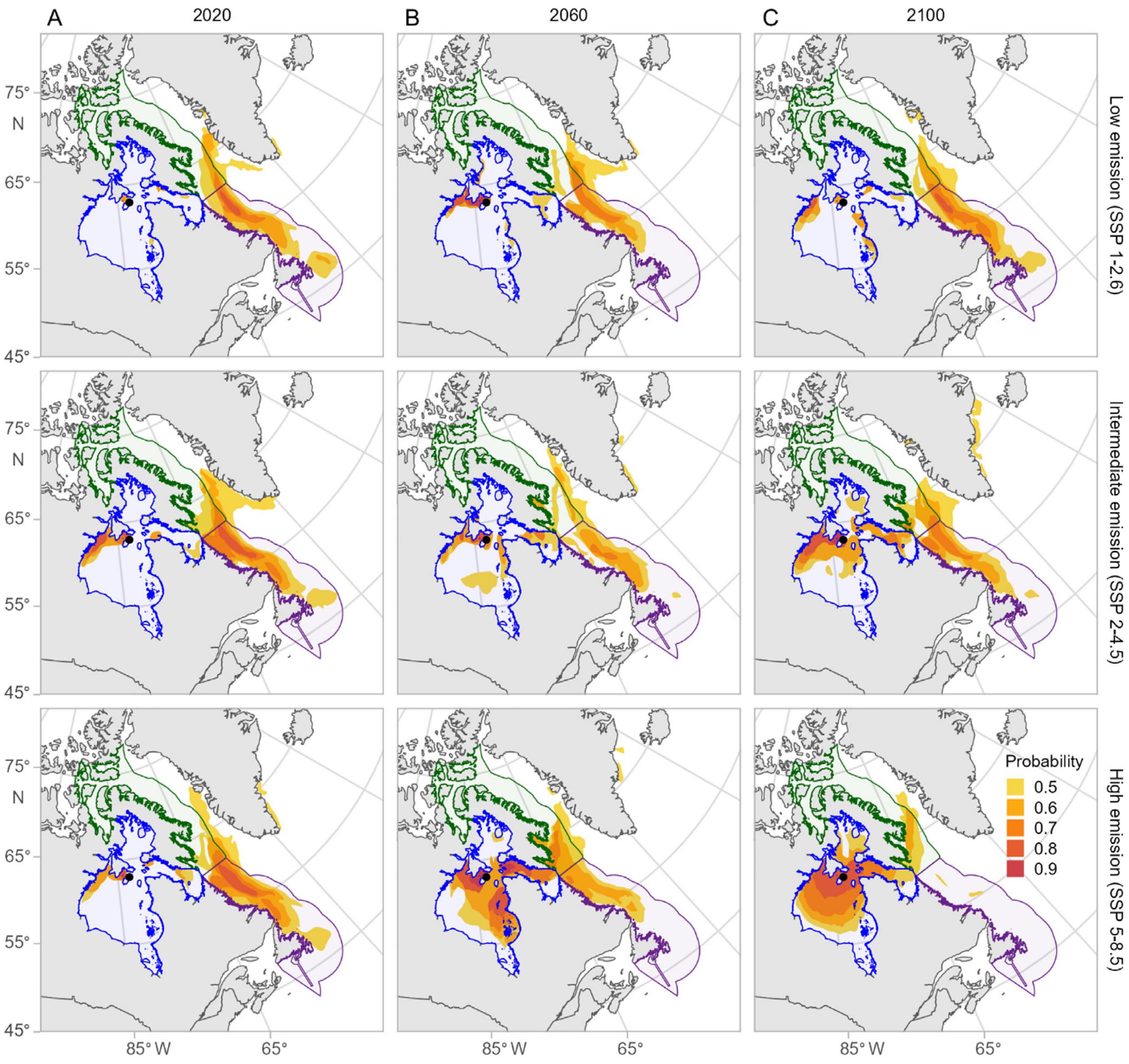

Fig. 5. Predicted distributions of thick-billed murres from the Coats Island, Nunavut, Canada, breeding colony, during spring migration (1 May) in (A) 2020, (B) 2060, and (C) 2100 under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) scenarios. Other details as in Fig. 2

4. DISCUSSION

We used a predictive modelling approach to determine how climate change is likely to alter the spatiotemporal distribution of non-breeding habitat for thick-billed murres from Coats Island, Nunavut, Canada. Among the models considered here, a high-emission future scenario would dramatically change habitat availability, distribution, and seasonal patterns for murres from this population. Increasing winter habitat within the HBC would enable murres from this population to forego their annual migration to the northwest Atlantic. The southern range boundary would shift north, so that murres from this population would no longer occur on the Newfoundland Shelf in winter, while occurrence within Baffin Bay would increase. Additionally, habitat suitability would decline during the pre- and post-breeding periods, when murres are constrained to areas close to the colony. These predicted large-scale changes in habitat and seasonal patterns would mostly be avoided in a low or moderate emission future scenario.

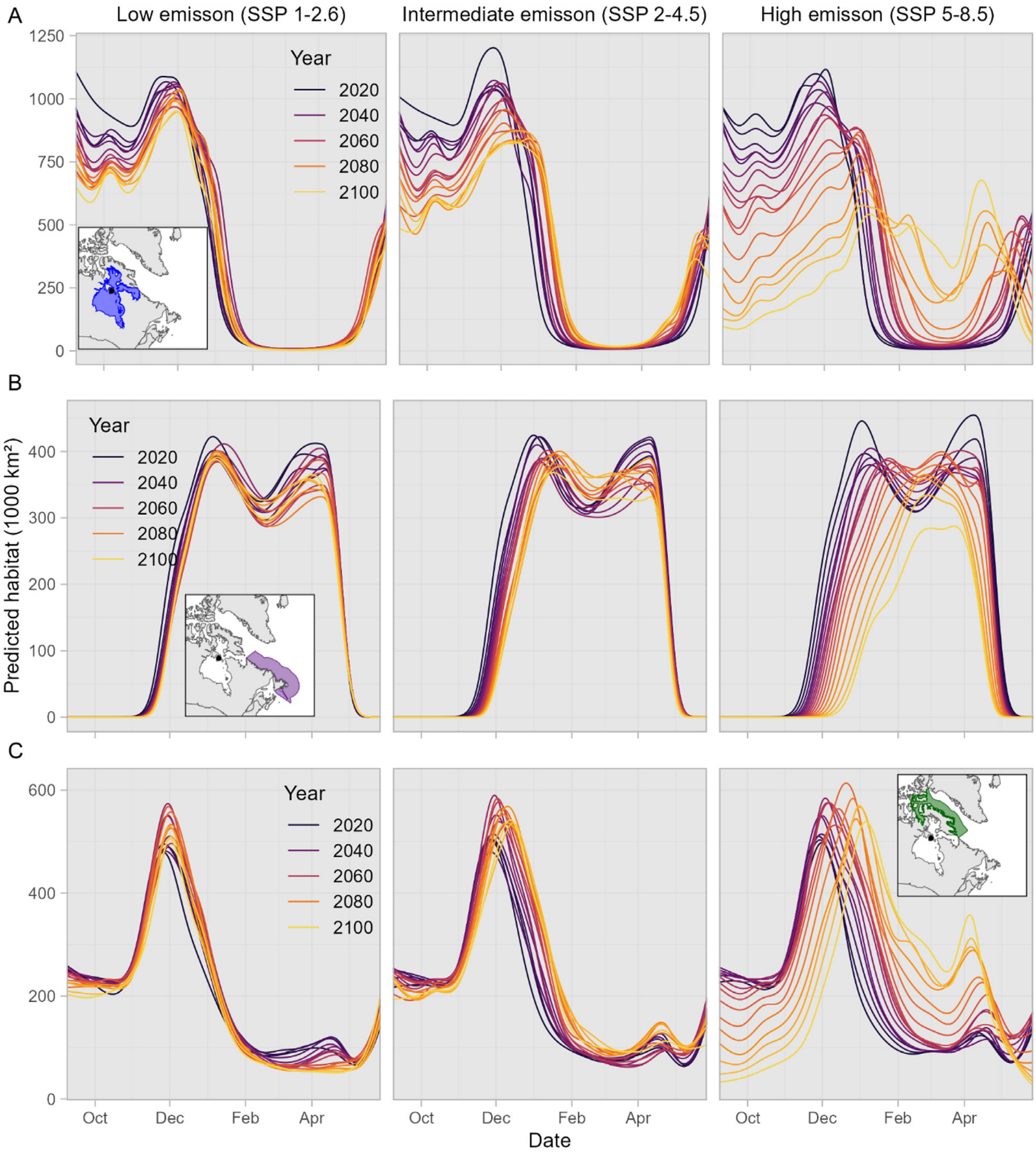

Fig. 6. Predicted suitable habitat area by day of year and year for thick-billed murres breeding at the Coats Island, Nunavut, Canada, breeding colony, within (A) the Hudson Bay Complex, (B) Newfoundland-Labrador Shelves, and (C) Eastern Arctic marine bioregions under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) climate scenarios. For visual clarity, only predictions for every fifth year are plotted; legend colours offer guide for colours at 20 yr intervals. The insets show the location of each marine bioregion

Although our model only applies to the relatively small breeding population of murres from Coats Island, similar patterns could be expected for other populations that share non-breeding ranges. In particular, the large colonies at Digges Sound (400 000 breeding pairs) in eastern Hudson Bay and Akpatok Island (800 000 breeding pairs) in Hudson Strait are the largest murre colonies in the Canadian Arctic

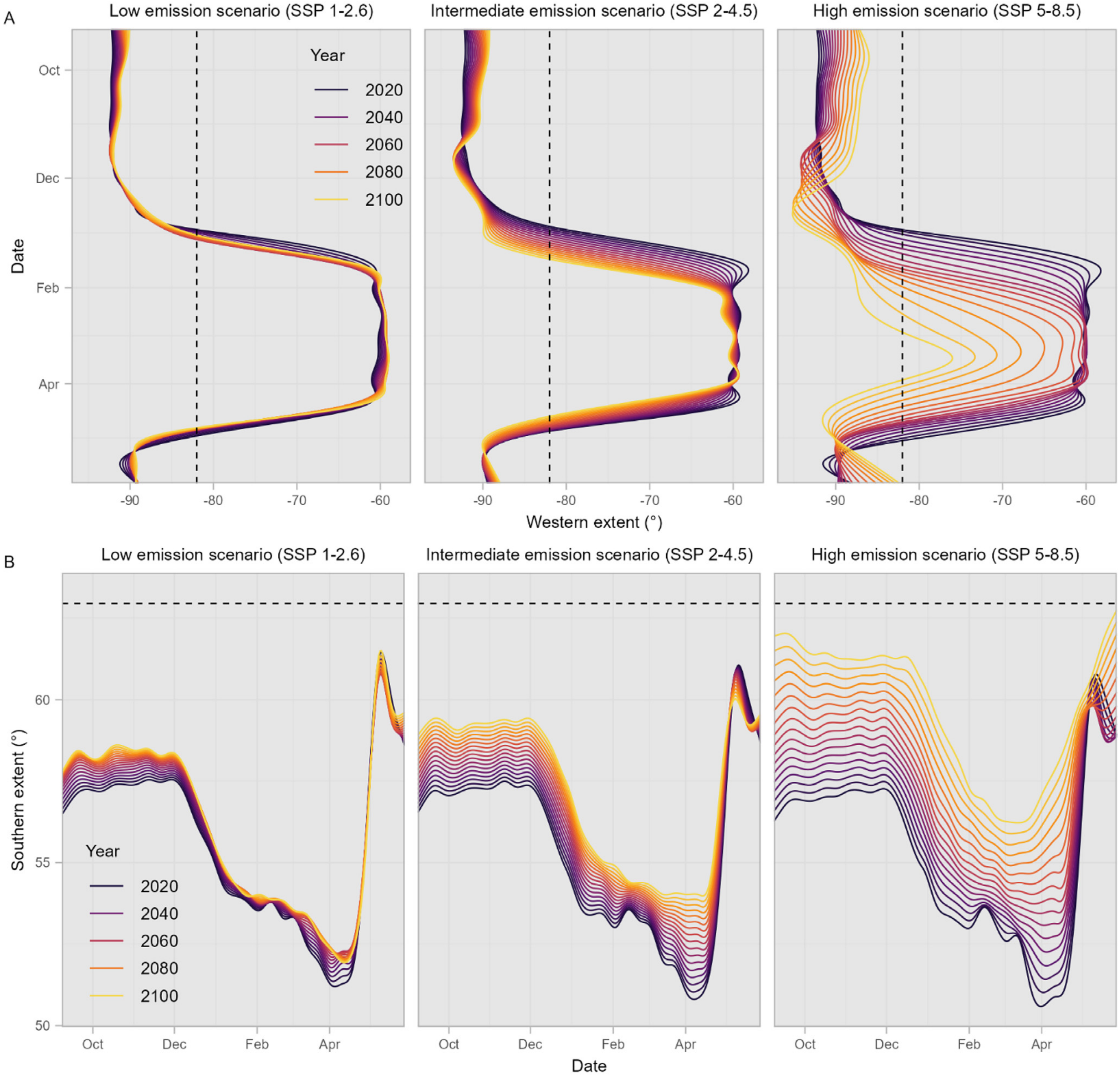

Fig. 7. Predicted (A) western and (B) southern range boundaries for thick-billed murres from the Coats Island, Nunavut, Canada, breeding colony under low (SSP 1-2.6), intermediate (SSP 2-4.5), and high emission (SSP 5-8.5) climate scenarios. For visual clarity, only predictions for every fifth year are plotted; legend colours offer guide for colours at 20 yr intervals. Dashed black lines show the longitude (top) and latitude (bottom) of the breeding colony

(Gaston et al. 2012a) and are among the largest murre colonies globally (Patterson et al. 2022a). Murres tracked from Coats Island had substantial winter range overlap with murres from Digges Sound (McFarlane Tranquilla et al. 2013, 2014), which supports our supposition that these colonies share non-breeding distributions. Indeed, winter range overlap is negatively associated with inter-colony distance, with closer colonies sharing more winter habitat than more distant colonies (McFarlane Tranquilla et al. 2014, Fauchald et al. 2021). Moreover, the Newfoundland Shelf supports wintering habitat for murre populations from Arctic and Atlantic Canada, as well as Greenland and Iceland (Frederiksen et al. 2019). As such, our finding that winter habitat in this region is likely to become less suitable for murres from Coats Island in a high-emission future scenario could also have significant impacts on the winter distribution of murres from other regions.

The most remarkable change predicted was the advent of year-round suitable habitat within Hudson

Bay, which we hypothesize could lead this population of murres to develop a partially migratory system with some members of the population remaining resident at the breeding site all year (Chapman et al. 2011). Thick-billed murres are largely migratory, but some populations in the Barents Sea are resident year-round (Fauchald et al. 2021). Closely related common murres are also resident to their breeding areas year-round (McFarlane Tranquilla et al. 2013, Dunn et al. 2020). There is significant variability in the timing of migration for murres from Coats Island (Patterson et al. 2021), with some birds remaining in Hudson Bay as late as possible in fall and returning as soon as open water appears in the spring. A portion of the population is therefore likely well positioned to respond to reduced ice cover. Adopting a resident lifestyle could have positive fitness consequences in spring through better breeding site retention ('prior residency') (Bennett et al. 2022b, Morinay et al. 2024) and earlier initiation of breeding ('arrival time') (Bennett et al. 2022a). Any benefits of remaining in the HBC would of course depend on the availability of suitable prey throughout the winter. However, remaining within an enclosed water body like Hudson Bay throughout winter could also make murres more susceptible to stochastic events, like storms and sudden ice movements. Murres can starve quickly when trapped by ice along the coast of Newfoundland (McFarlane Tranquilla et al. 2010) or exposed to winter cyclone events in the North Atlantic (Clairbaux et al. 2021b). A similar risk has been proposed for narwhal *Monodon monoceros* delaying their migration out of Eclipse Sound, Nunavut, Canada (Shuert et al. 2022). In Scotland, resident European shags *Gulosus aristotelis* had higher mortality associated with extreme climatic events than migratory individuals from the same population, reducing survival selection for resident phenotypes and demonstrating the complexity of demographic consequences of climate change responses (Acker et al. 2021).

Late spring and early fall predictions from the high-emission scenario predicted a dramatic decline in suitable habitat within Hudson Bay (e.g. close to the breeding colony), which could lead to this region becoming unable to support a breeding population. Murres have high breeding site fidelity, undertake a flightless moult from September to October, and males have post-fledgling care of flightless young (Gaston & Hipfner 2020); these life-history traits are likely to constrain murre ranges to the area immediately around existing breeding colonies during the pre- and post-breeding periods (e.g. May, September, and October). Our SDM included spatial and temporal covariates for distance from colony and DOY, which were important in the original parameterization of the SDM for accurately recreating the annual movement patterns for our study population (Patterson et al. 2021). Future projections from our model enforce this constraint, limiting predicted distributions in spring and fall to areas primarily within Hudson Bay. In a high-emission future scenario, the SDM predicted dramatic declines in suitable habitat for moult and pre-breeding (Fig. 5), as SST and air temperature in HBC increase outside of the historic range. Even under the low-emission scenario, there is a moderate decline in habitat in the HBC during moult. These results highlight potential vulnerability for murres and other colonial breeding marine species, during times of the year when they are closely tied to a fixed location, especially when fidelity to that location is high. Although our model did not include the breeding period (June to August), because it was explicitly fit to tracking data collected during non-breeding, the consistent trend of declining habitat suitability pre- and post-breeding point to declining habitat suitability through the breeding period as well. This trend is consistent with on-colony observations of adult mortality events associated with higher air temperatures (Gaston & Elliott 2013) and changes in diet associated with declining ice cover (Smith & Gaston 2012). Because murres are constrained to foraging within 200 km of their breeding colonies (Elliott et al. 2009, Patterson et al. 2022a), they would be unable to respond to declining habitat close to the colony by commuting farther to forage. Under a high-emission scenario, we hypothesize that declining habitat suitability around the breeding season could lead to greatly reduced reproductive success or even complete colony collapse. Understanding how future climate change could impact habitat suitability around breeding for colonial seabird species with restricted foraging ranges and high philopatry, like alcids, could help assess the range-wide population risks of climate change for these species.

How murres respond to changes in marine climate is likely to be mediated through changes in the distribution and abundance of their prey (Piatt et al. 2020). Within HBC, a high-emission scenario is expected to result in declines in the biomass and abundance of Arctic cod and increases in smaller pelagic prey species (e.g. capelin, sculpin, and sand lance) (Florko et al. 2021). A better understanding of the vulnerability of murres to future climate change would benefit from including modelling of climate change impacts on lower trophic levels. Murres are generalist predators that can exploit a variety of prey (Gaston et al. 2003, Woo et al. 2008); however, switching to lower-quality prey could have population consequences for productivity and survival (Piatt et al. 2020).

In a future high-emission scenario, our model predicts a northward shift in the southern range boundary, decline in suitable winter habitat for murres from Coats Island on the NLS, and increased winter habitat in the EA. This would mean that the winter distribution of adult murres from this population would no longer overlap with the murre harvest in Newfoundland. Delayed and reduced migration towards the east, as well as reduced use of the NLS during fall, could also alter the availability of adult murres for hunting in Labrador. However, increased habitat through winter along the Labrador Shelf could provide opportunities for murre harvest in this region outside of the existing hunting seasons. Anthropogenic sources of mortality, namely oiling and hunting, on the Newfoundland Shelf could negatively affect population growth rates of murres from colonies in Hudson Bay (Wiese et al. 2004, Frederiksen et al. 2019, Cox et al. 2024). Shifting distributions away from this relatively highly populated region could reduce winter mortality. However, anthropogenic sources of mortality within newly occupied regions could also increase if there are concomitant changes in hunting patterns, industrial development (e.g. mining, oil and gas development), commercial fishing, and shipping activity in response to reduced ice in Baffin Bay, Hudson Strait, and Hudson Bay.

Our habitat model was trained on data for adult murres only; however, juvenile murres from populations in Hudson Bay also use the NLS in winter. Unfortunately, knowledge of habitat use by juveniles is largely constrained to information from band recoveries in hunting and beach surveys (Frederiksen et al. 2019). Juveniles could be susceptible to changes in winter habitat quality because of their relative inexperience. Juveniles are particularly susceptible to harvest mortality (Frederiksen et al. 2019) and can be susceptible to winter wrecks (McFarlane Tranquilla et al. 2010). Additional studies tracking juvenile movements and modelling juvenile distributions are an important data gap in understanding climate change impacts on murres and other long-lived seabird species (Péron & Grémillet 2013, Carneiro et al. 2020, Lane et al. 2021).

The northward range shift and increased midwinter habitat in the Eastern Arctic predicted under a high-emission scenario would result in murres experiencing reduced daylengths during the non-breeding period. Murres are visual predators that primarily forage during daylight (Elliott & Gaston 2015); however, nocturnal foraging occurs in both the breeding and non-breeding periods (Elliott & Gaston 2015, Orben et al. 2015, Patterson et al. 2022b). In the North Pacific, smaller-bodied birds were more likely to forage at night (Orben et al. 2015). For birds from Coats Island, nocturnal foraging was more common in the Labrador Basin than on the Labrador Shelf, and murres tended to increase nocturnal foraging in response to increased moon illumination (Patterson et al. 2022b). Some thick-billed murres in Svalbard winter in Arctic regions with 24 h darkness (Berge et al. 2015, Ostaszewska et al. 2017). A northward shift in winter habitat towards regions with limited daylight could restrict foraging opportunities for murres and increase intra- and inter-specific competition for food. Relatively little is known about winter foraging behaviour and diet for this species; research on this subject would improve our ability to understand how northern range shifts will impact winter survival and population dynamics.

A multispecies, multi-population study looking at climate change impacts on winter distributions of North Atlantic seabirds, including murres, similarly concluded that a high-emission scenario would lead to changes in winter habitat for murres and that this could be avoided under a low-emission scenario (Clairbaux et al. 2021a). That analysis did not identify increased use of HBC under a high-emissions scenario. We attribute this to the focus on estimating and reporting predictions at an ocean basin scale, which likely masked effects specific to the Hudson Bay region. Large-scale multi-population studies are valuable for understanding broad-scale patterns of climate change. However, individual population responses and management implications will be driven by the unique environmental conditions specific to their own ranges. Our study highlights the value of also examining the effects of climate change at population and region-specific levels.

Our model assumes murres can shift non-breeding distributions to maintain association with analogous climate conditions to what they experience today. The model assumes that murres have the behavioural flexibility to adapt their migratory patterns in response to environmental change. Indeed, while murres have advanced breeding at Coats Island in response to earlier ice-off dates in Hudson Bay, the advance in lay date has not kept pace with the rate of change in the environment (Whelan et al. 2022). In winter, murres in the northwest Atlantic switch between available marine habitats (shelf and basin) and use different foraging strategies (diurnal and nocturnal foraging) within each habitat (Patterson et al. 2022b). This suggests that non-breeding murres may have significant flexibility to adapt foraging behaviour despite changes in local environmental conditions and the distribution of prey species within those regions. Biologging of migratory wildlife is necessary to detect shifting migration responses and behavioural adaptations to climate change (Chmura et al. 2018). Continued long-term tracking of Arctic migratory marine species is critical for understanding effects of climate change on these species.

Data availability. Predicted distributions from this study are available at https://doi.org/10.5061/dryad.xgxd254s6.

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