

Species Status Assessment for the Spectacled Eider

Central Yup-ik: *Quageq*

Inupiaq: *Qavaasuk*

Siberian Yup-ik: *Iyegaatelek*

Scientific: *Somateria fischeri*



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Executive Summary

This species status assessment (SSA) was developed to inform a 5-year status review of spectacled eiders. The spectacled eider was listed under the Endangered Species Act (ESA) as a threatened species throughout its range in Alaska and Russia on May 10, 1993 (58 FR 27474) because the species had declined by as much as 96 percent in western Alaska, which, at the time, was thought to be its primary breeding range. The factors that caused the decline were unknown at the time of listing and remain unknown today. Critical habitat was designated on February 6, 2001 to protect molting areas in Norton Sound and Ledyard Bay, nesting areas on the Yukon-Kuskokwim Delta (Y-K Delta), and the wintering area south of St. Lawrence Island (66 FR 9146). A recovery plan finalized on August 12, 1996 included recovery criteria that required each of the three breeding populations (western Alaska, northern Alaska, and Arctic Russia; referred to here as the Y-K Delta, Arctic Coastal Plain [ACP], and Arctic Russia breeding populations, respectively) to meet a threshold for abundance and trend and that threats be ameliorated.

The SSA synthesizes available information on spectacled eiders and provides an assessment of the species' current and future viability. Viability is defined as the likelihood that a species will persist over time and is a product of resilience, representation, and redundancy, otherwise known as "the three Rs" (Shaffer and Stein 2000). **Resilience** is defined as the ability of a population to withstand the natural range of favorable and unfavorable conditions, referred to as demographic and environmental stochasticity. For this SSA, each of the three breeding populations of spectacled eiders (Arctic Russia, Y-K Delta, and ACP) are referred to as populations. The degree of resilience of a population is positively related to its size and growth rate, and can also be characterized using information about demographic rates such as survival, reproduction, and immigration, as well as habitat quantity and quality.

Representation is the ability of a species to adapt to near-term and long-term changes in its physical (climate, habitat) and biological (disease, predators, competition) environment. Representation is characterized by the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas. **Redundancy** is the ability of a species to withstand catastrophic events and is characterized by the number and distribution of resilient populations within the species' ecological settings and historical range. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely.

Spectacled eiders inhabit marine waters for the majority of their annual cycle, and migrate to coastal tundra habitat in Alaska and Russia to nest, which occurs from approximately mid-May to early September. In Alaska, two breeding populations representing the highest nesting densities of spectacled eiders are recognized: the Y-K Delta and ACP breeding populations. On the Y-K Delta, spectacled eiders are mostly concentrated within 15 km of the coast from Kigigak Island north to Kokechik Bay. The coastal fringe of the Y-K Delta is the only subarctic breeding habitat where spectacled eiders occur at high densities. Spectacled eiders occur at lower densities on the ACP, which lies above the Arctic Circle, and their current distribution stretches from approximately Pt. Lay to the Canning

River. In Arctic Russia, spectacled eiders nest discontinuously from the Kolyma River to the Yana River Delta in Siberia, with highest nesting densities in the Chaun, Kolyma, Yana, and Indigirka River Deltas. There is some uncertainty about specific habitat requirements for nesting spectacled eiders, but based on the available information, factors that may influence demographic rates include: quality and quantity of freshwater invertebrates and vegetation; availability of nesting (wet coastal tundra within 3m of shallow ponds or lakes) and brood-rearing (vegetated areas close to ponds, emergent vegetation, and adequate pond depth) habitat; and, duckling access to sources of freshwater.

After nesting, spectacled eiders migrate and molt in shallow coastal water with sandy-gravelly substrates, typically less than 18 m deep. Migration corridors between nesting, molting, and wintering areas occur over marine waters up to 60 km offshore of Alaska and northeastern Russia. Use of spring migration corridors and wintering areas may be limited by openings in sea ice. In winter (November – April), spectacled eiders use areas in the open ocean, polynyas, or open leads in the sea ice of the northern Bering Sea and southern Chukchi Sea at water depths of less than 80 m. In the marine environment, factors that may affect demographic rates include: quantity and quality of marine benthic invertebrates, availability of shallow coastal water and coastal habitat up to 60km offshore, availability of marine waters with depth less than 80m, and sea ice concentration.

Several factors may affect individual spectacled eiders, and some may rise to the level that they affect population resilience. For many factors, we have limited information with which to quantify their effects to spectacled eiders, and therefore determine if they are significant population-level threats. However, two factors, exposure to lead and climate change (specifically the effect of both extreme and low sea ice conditions) are of the most concern for spectacled eiders and have been shown to affect demographic rates. All of the factors identified in this analysis are expected to continue, at some level, in the future. Some, such as collisions, exposure to contaminants, and salinization of freshwater wetlands, are expected to increase in severity in the future. With adequate management actions, harvest and lead exposure may decrease over time. Climate change has altered, and is expected to continue to alter, both the tundra breeding habitat and the marine non-breeding habitats of spectacled eiders. The northern Bering Sea in particular is expected to undergo drastic changes in the next several decades due to a reduction in sea ice. This may have consequences for spectacled eider food availability and thermoregulation in winter, resulting in reductions in breeding propensity and survival; however, whether spectacled eiders can behaviorally adapt and find adequate foraging conditions is unknown.

To evaluate the current resiliency we relied primarily on estimates of abundance and population growth rate, as these should reflect the effects of influential factors and habitat changes on the population resilience. The cause(s) of the population decline on the Y-K Delta, which was the primary factor influencing the listing of the species, are unknown. In addition, stressors influencing current population growth and potential habitat limitations are not well-understood. We have information on population abundance and growth rate for ACP and Y-K Delta breeding populations, and therefore based condition categories described in (Section 3.3.1) on these parameters. Few studies have been conducted on the Arctic Russia breeding population, and it is not regularly monitored. The only measure of resilience available for the Arctic Russia population is an estimate of the minimum global population size obtained

from occasional surveys during winter, from which a minimum size of the Arctic Russia population can be inferred.

Based on the Bayesian state-space model, the Y-K Delta breeding population has increased since the species was listed in 1993, and there are likely more than 12,000 spectacled eiders in the population. The estimated mean population growth rate from 2007 – 2019 is positive, but the lower 95 percent confidence interval bound is not greater than one percent per year, suggesting that the population is stable. However, when considering the entire time frame from 1988 to 2019, and using an IPM, the average annual growth rate is 0.051 (95 percent CI 0.036 – 0.067), indicating that the population increased over the long-term. The population appears to have leveled, or is growing at a slower rate, in recent years. The current abundance in the ACP breeding population is likely more than 3,500 individuals. We have limited information on the historical abundance of spectacled eiders in the ACP breeding population; however, the available information suggests that the breeding population has been stable since listing. The Arctic Russia breeding population abundance and trend are unknown, other than what can be inferred from estimates of the wintering population, which comprises all three breeding populations of spectacled eiders. Because abundance information on the Arctic Russia breeding population is not available, we subtracted the combined estimates of the Alaska populations (Y-K Delta and ACP) from the count of the global population (Y-K Delta, ACP and Arctic Russia combined) to calculate the minimum size of the Arctic Russia population. Following this methodology, the Arctic Russia population in March 2020 was comprised of *at least* 45,734 spectacled eiders, recognizing that this is an underestimate of the true population size.

In summary, by comparing the best available information on population abundance and growth rate with the condition categories described in Section 3.3.1, we consider the current resilience of the Y-K Delta breeding population to be moderate-high, and the current resilience of the ACP breeding population to be low-moderate. The estimated minimum size of the Arctic Russia population suggests high resilience, but without concurrent estimates of population growth rate, we do not have sufficient information to adequately characterize its current level of resilience.

In addition to resiliency, we considered representation and redundancy of spectacled eiders. Representation is the ability of a species to adapt to changes in its environment. We evaluated the representation of spectacled eiders by determining the number of ecotypes used throughout the annual cycle. We also discuss the possible range of behavioral flexibility in response to environmental conditions and the genetic diversity of the species. Overall, the current representation of spectacled eiders can be characterized by the following points: 1) spectacled eiders are widely distributed in at least three ecotypes during the breeding season (Y-K Delta, ACP, and Arctic Russia); 2) they use four molting areas with benthic communities that likely differ in composition; 3) they may be able to shift their distribution in response to changing sea ice conditions in the Bering Sea in winter; 4) evidence supports that they shift their diet depending on availability of benthic bivalve species; and, 5) there is no indication that genetic diversity is limiting their adaptive capacity.

Redundancy is the ability of a species to withstand catastrophic events, and is described by the number and distribution of resilient populations. The global population of spectacled eiders continues to be comprised of three breeding populations that use at least four different molting areas spanning a relatively large and geographically diverse area. However, the high degree of female breeding area fidelity and natal philopatry (Section 2.2.3) makes it unlikely for spectacled eiders to re-colonize an area should the population become extirpated. In addition, historically, the entire global population used one wintering area in the Bering Sea where birds were concentrated in openings in the sea ice, which may make them susceptible to catastrophic events such as a disease outbreak. In the last decade there have been fewer years with extreme sea ice conditions and wintering flocks appear to be more dispersed in milder years. While extreme sea ice conditions are predicted to still occur in some years, a wider winter distribution increases the ability of the species to withstand a catastrophic event.

We expect significant changes to the marine and terrestrial habitats used by spectacled eiders in the future due to climate change (Section 3.1.10). In particular, climate models predict that sea ice extent will decrease significantly and the Bering Sea ecosystem will change due to changes to primary productivity, species distributions, and ocean acidification. However, it is uncertain how spectacled eiders will respond to such changes. We also expect some factors that negatively affect spectacled eiders to increase in the future, such as collision risk, contaminant exposure, and disturbance due to increasing vessel traffic in the Bering and Chukchi seas, and habitat loss due to industrial and community development. Others, such as lead exposure and harvest, may decrease with concerted management efforts in Alaska. However, we have no indication that such management efforts are planned or possible in Arctic Russia.

We acknowledge that several factors may influence resilience of spectacled eider populations, including those described in the above paragraph. Given the results of simulations based on plausible scenarios of sea ice conditions and lead exposure rates, factors that have been shown to affect demographic rates and of which we have data for future projections, the probability of QE remains relatively low through 2060, but increases over time for both Alaska-breeding populations in all scenarios. We do not have data to project future scenarios for the Arctic Russia population but suspect that the same factors will act on the resilience of that breeding population. Therefore, we predict that the resilience of all three populations will decrease over time, but to an unknown degree. However, because all three populations are likely to persist beyond 2100 based on model results, changes in redundancy are not expected. Both tundra and marine environments are likely to change significantly in the future. We do not have data with which to predict how genetically and/or behaviorally flexible the species will be in regards to these changes; however, we expect the species to be forced to alter its distribution, diet, and timing of key life history events (nest initiation and migration) in response. Therefore, we predict that the representation of spectacled eiders may be reduced as their habitats shift, particularly past 2060 and closer to 2100.

1. Introduction

The spectacled eider (*Somateria fischeri*) was listed under the Endangered Species Act (ESA) as a threatened species throughout its range in Alaska and Russia on May 10, 1993 (58 FR 27474) because the species had declined by as much as 96 percent in western Alaska, which, at the time, was thought to be its primary breeding range. The factors that caused the decline were unknown at the time of listing and remain unknown today. Critical habitat was designated on February 6, 2001 to protect molting areas in Norton Sound and Ledyard Bay, nesting areas on the Yukon-Kuskokwim Delta (Y-K Delta), and the wintering area south of St. Lawrence Island (66 FR 9146). A recovery plan finalized on August 12, 1996 included recovery criteria that required each of the three breeding populations (western Alaska, northern Alaska, and Arctic Russia; referred to here as the Y-K Delta, Arctic Coastal Plain [ACP], and Arctic Russia breeding populations, respectively) to meet a threshold for abundance and trend and that threats be ameliorated (USFWS 1996; Figure 1). While the recovery plan has not been revised, the Spectacled Eider Recovery Team (Team) and the U.S. Fish and Wildlife Service (USFWS) review and update a list of spectacled eider recovery tasks periodically.

A 5-year status review of spectacled eiders was completed in June 2010 (USFWS 2010). Using the best available information, the review concluded that two of the three breeding populations were likely approaching or had exceeded delisting criteria (Y-K Delta and Arctic Russia), but the ACP breeding population had likely not met recovery criteria. The review also concluded that potential impacts from climate change in terrestrial and marine habitats, as well as other threats such as lead contamination, may continue to negatively impact population resilience. In conclusion, the 2010 5-year review recommended the species remain classified as threatened.

This species status assessment (SSA) was developed to inform a 5-year status review to be completed in 2021. The SSA synthesizes available information on spectacled eiders and provides an assessment of the species' current and future viability. To improve transparency, the SSA process was developed by the USFWS to separate evaluations of the best available science from the decision-making process that necessarily combines scientific evaluations with policy interpretation and risk assessment. While we limited this analysis to assessing the available scientific information and traditional ecological knowledge (TEK), prior to conducting the SSA, we found it useful to identify the upcoming policy decisions and management needs to focus our analysis on the topics pertinent to those decisions. This SSA is intended to be the basis for future recovery plan revisions, consultations with Federal agencies under ESA Section 7(a) (1) and 7(a) (2), prioritizing recovery actions, and other recovery program needs. However, the most pressing management task is the completion of a 5-year review of the spectacled eider by August 2021. The primary purpose of a 5-year review is to ensure that listed species have the appropriate level of protection under the ESA. Additionally, distinct population segment designations are reviewed in light of new information on the species biology and current interpretation of policy. We will also consider whether the recovery plan warrants revision.

Our analysis follows the SSA guidelines developed by the USFWS (USFWS 2016), in which viability is defined as the likelihood that a species will persist over time and is a product of resilience, representation, and redundancy, otherwise known as “the three Rs”(Shaffer and Stein 2000). **Resilience** is defined as the ability of a population to withstand the natural range of favorable and unfavorable conditions, referred to as demographic and environmental stochasticity. For this SSA, each of the three breeding populations of spectacled eiders (Arctic Russia, Y-K Delta, and ACP) are referred to as populations. The degree of resilience of a population is positively related to its size and growth rate, and can also be characterized using information about demographic rates such as survival, reproduction, and immigration, as well as habitat quantity and quality. **Representation** is the ability of a species to adapt to near-term and long-term changes in its physical (climate, habitat) and biological (disease, predators, competition) environment. Representation is characterized by the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas. **Redundancy** is the ability of a species to withstand catastrophic events and is characterized by the number and distribution of resilient populations within the species’ ecological settings and historical range. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely.

In addition to using the available information in scientific literature and agency reports, we considered Indigenous knowledge (IK) gained through community visits, meetings of management councils such as the Alaska Migratory Bird Co-Management Council, and other conversations over the past few decades. We are not aware of specific TEK reports or quantitative information resulting from TEK surveys on spectacled eiders. However, these conversations and meetings have informed our thought process in a general way; for example, we’ve learned through years of working on the ACP and Y-K Delta that while occasionally harvested, spectacled eiders are not considered an important subsistence resource (See Section 3.1.2).

We begin this assessment by describing taxonomy and physical characteristics, and the circumstances and resources that influence the ability of individuals to successfully complete each life stage. We then evaluate the demographic factors and stressors that may affect the current and future resiliency of the three breeding populations, and assess the current and future representation and redundancy of the species.



Figure 1. Spectacled eider distribution. Satellite telemetry data collected during a year with low sea ice in the Bering Sea suggest that spectacled eiders disperse once sea ice retreats in late winter, and use known migration, staging and molting areas north of the traditional wintering area until moving to breeding areas.

2. Species Biology and Habitat Requirements

2.1 Species Physical Description and Taxonomy

Spectacled eiders are medium-sized sea ducks, averaging about 1,500 grams (g; 3.3 pounds) in weight and 50 centimeters (cm; 19.7 inches) in total length (Petersen et al. 2000 p. 2). Males in breeding plumage have a white back, black breast, and pale green head with large white, black-rimmed “spectacles” around the eyes. In late summer and autumn adult males molt into a mottled brown plumage that lasts until late fall, when they re-acquire breeding plumage. Females are mottled brown year round, with pale tan spectacles. Juveniles attain breeding plumage in their second (female) or third (male) year; until then females are mottled brown and males mottled brown and white. Both males and females have sloped foreheads and bills.

Spectacled eiders are classified as a sea duck in the subfamily *Anatinae* and the tribe *Mergini*; cladistics analysis of morphological characters of the tribe *Mergini* further places spectacled eiders in the monophyletic subtribe *Somatereae* with Steller's eiders (*Polysticta stelleri*), common eiders (*Somateria mollissima*), and king eiders (*Somateria spectabilis*; (Livezey 1995 p. 236). Spectacled eiders are basal to the other congeners in the genus *Somateria*, king and common eiders (Livezey 1995 p. 236). No subspecies of spectacled eiders have been identified, and no geographic variation in plumage or morphology has been detected (Petersen et al. 2000 p. 3).

2.2. Distribution and Life History

2.2.1. Study Areas

Spectacled eiders have been studied most extensively at five study areas: two on the Y-K Delta (Kigigak Island and Kashunuk River), one on the ACP (Utqiagvik), and two in Russia (Chaun Delta and Indigirka Delta). Here we provide brief descriptions of study sites for reference throughout the document.

Yukon-Kuskokwim Delta breeding population

The Kashunuk River study site (Kashunuk), described in detail by Grand et al. (1997 pp. 1199–2002), is bisected by a 200 – 400 m (656 – 1312 foot) wide river. Kashunuk is tidally-influenced and can be almost completely flooded during periodic storm tides. With an elevation between 0 – 3 m (0 – 9.8 feet), habitat types include high-intertidal mudflats, sedge meadow, high-graminoid meadow, dwarf-shrub upland, and intermediate-sedge meadow (described in detail in Grand et al. 1997:1200 – 1201). Kashunuk is approximately 40 kilometers (km) by boat from the village of Chevak (population 1,014 (State of Alaska 2020 p. 110); this area is traditionally used for subsistence hunting or fishing. Spectacled eider nesting ecology research was conducted from 1993 – 2002 at Kashunuk (Flint et al. 2016 p. 6).

Kigigak Island (Kigigak) is a 32.5 km² (12.6 mi²) nearshore island located along the central coast of the Y-K Delta near the mouth of Baird Inlet, bordered by the Ninglick River and the Bering Sea (Figure 2). Kigigak has one of the highest overall nesting densities of waterfowl on the Y-K Delta (Flint et al. 2016 p. 6). The island is tidally-influenced, contains many small ponds and lakes, and exhibits diverse habitats similar to those described above at Kashunuk (Daniels and Friendly 2018 p. 2). Spring and fall storm tides regularly inundate the island, except for upland areas which are flooded only during severe tidal storms (Gabrielson and Spragens 2013 p. 3). Kigigak is about 30 km (18.6 mi) by boat from the village of Newtok (population 339; State of Alaska 2020 p. 111) and access is difficult at times due to extensive mudflats. Spectacled eider nesting biology, female survival, and habitat use was studied at Kigigak from 1992 – 2015 (Gabrielson and Spragens 2013, Flint et al. 2016, Moore and Sowl 2017), and in 2019 (Rizzolo 2019).



Figure 2. Locations of long-term study areas for spectacled eiders in Alaska and eastern Russia (in green).

Both Y-K Delta study areas have high densities of waterbirds, including: cackling goose (*Branta hutchinsii*), greater white-fronted goose (*Anser albifrons*), emperor goose (*Anser canagicus*), brant (*Branta bernicla*), common eider, tundra swan (*Cygnus columbianus*), and sandhill crane (*Antigone canadensis*), as well as several species of shorebirds and songbirds (Flint et al. 2016 p. 6). Nest predators include Arctic fox (*Vulpes lagopus*), red fox (*V. vulpes*), parasitic jaeger (*Stercorarius parasiticus*), and glaucous gull (*Larus hyperboreus*) (USFWS 1996 p. 6). Other Larids present on the Y-K Delta include mew gull (*Larus canus*), Sabine's gull (*Xema sabini*), and Arctic tern (*Sterna paradisaea*). Microtine species serve as a significant source of food for avian and mammalian predators when their numbers are high on the Y-K Delta. Tundra voles (*Microtus oeconomus*) and red-backed voles (*Myodes rutilus*) are the dominant microtine species, but collared (*Dicrostonyx groenlandicus*) and brown lemmings (*Lemmus trimucronatus*) are present in lower numbers.

Arctic Coastal Plain breeding population

The Utqiagvik study area (Utqiagvik) is located at the northern-most point of the ACP, adjacent to the community of Utqiagvik (population 4,536, (State of Alaska 2020 p. 114). The area consists of low-elevation tundra and poorly-drained wetlands underlain with permafrost, with emergent vegetation dominated by water sedge (*Carex aquatilis*) and pendant grass (*Arctophila fulva*). Low and high-centered ice-wedge polygons form small ponds, and larger shallow lake basins are scattered throughout the area (Safine et al. 2020 p. 350). While the weather is marine-influenced, wetlands are freshwater and not tidally-influenced. The faunal community differs from the Y-K Delta; brown lemming populations fluctuate annually and provide the primary source of food for avian and mammalian predators when their numbers are high. Collared lemmings are present but in lower numbers. Common mammalian predators are Arctic fox and least weasel (*Mustela nivalis*), and avian predators include snowy owls (*Bubo scandiacus*), pomarine jaegers (*Stercorarius pomarinus*), parasitic jaegers, long-tailed jaegers (*Stercorarius longicaudus*), glaucous gulls, and common ravens (*Corvus corax*; Safine et al. 2020:350). Other common waterbirds include greater white-fronted goose, long-tailed duck (*Clangula hyemalis*), king eider (*Somateria spectabilis*), and Steller's eider (*Polysticta stelleri*). The study area also encompasses a road network used by local residents and visitors for many purposes, including infrastructure maintenance (e.g., natural gas pipeline, landfill, and gravel extraction), subsistence activities, research, and tourism. Data collection on spectacled eiders at Utqiagvik occurred from 2009 – 2019 (Safine 2011, 2012, 2013, 2015, Graff 2016, 2018, 2020a).

Arctic Russia breeding population

The Chaun Delta study site (Chaun) is located on Ayopechan Island, the largest island in the delta where the Chaun and Palyavaam rivers meet the East Siberian Sea (Figure 2). Maximum elevation on the island is 5 – 6 m (16.4 – 19.7 feet), and peat bogs with lakes formed by thermokarst and alas depressions, and a network of tidal sloughs, dominate the topography (Solovyeva and Kokhanova 2017 p. 4). Similar to study sites on the Y-K Delta, occasional tidal floods keep most wetlands brackish, except for a few upland ponds and lakes. Dominant avian species include: tundra swan, long-tailed duck, king eider, glaucous gull, herring gull (*Larus argentatus vegae*), Sabine's gull, Arctic tern, and parasitic jaeger. Both Arctic and red fox also occur in the study area (Solovyeva et al. 2018 p. 745). Spectacled eider nesting biology was studied at Chaun from 2003 – 2016, except for 2006 (Solovyeva and Kokhanova 2017), and minimal monitoring activities continue on an annual basis.

Indigirka River Delta study area (Indigirka) is located on the northern coast of Yakutia, Russia, west of Chaun (Figure 2). Indigirka includes two separate study sites: an interior site 25 km inland from coast near the village of Tabor, consisting of polygonal tundra and drained-basin wetland complexes; and a coastal site consisting of ten small islands with deep lakes with emergent pendant grass (Pearce et al. 1998a:362 – 363). Biological work that included spectacled eider observations occurred in 1971 (Kitschinski and Flint 1974) and 1993 – 1995 (Pearce et al. 1998b, 1998a).

2.2.2. Distribution and Lifecycle

Spectacled eiders inhabit marine waters for the majority of their annual cycle, and migrate to coastal tundra habitat in Alaska and Russia to nest (Figure 1).

Pair Formation

Pair formation likely occurs on the wintering area, as individuals arrive paired on breeding areas (Dau and Kistchinski 1977 p. 67, McCaffery et al. 1999 p. 168, Eadie and Savard 2015 p. 368), and have been observed engaging in courtship behavior in the wintering area in late March (Lovvorn et al. 2012 p. 1088).

Pre-nesting Distribution

Observations of spectacled eiders arriving from the north or northwest to the Y-K Delta breeding area resulted in the hypothesis that spectacled eiders used pre-breeding staging areas north of the northern Bering Sea wintering area (Dau and Kistchinski 1977 pp. 65 – 66, McCaffery et al. 1999 p. 169). Data collected through satellite telemetry support this hypothesis. Spectacled eiders instrumented with satellite transmitters (platform transmitting terminals; PTTs) in the 1990s and again in 2008 – 2011 dispersed from the northern Bering Sea wintering area as sea ice retreated northward, in March or April (Petersen et al. 1999 fig. 1017, Sexson et al. 2014 pp. 52–53). Most eiders used nearshore areas of the western Bering Strait near Mechigmentskiy Bay, Russia, prior to moving to staging areas in ice leads immediately offshore of breeding areas in April (coastal areas off the Y-K Delta), and May (eastern Chukchi Sea or East Siberian Sea). Some marked spectacled eiders also staged in southern Norton Sound prior to moving to the Y-K Delta breeding area (Sexson et al. 2014 pp. 52 – 53). Timing of arrival to pre-breeding and breeding areas is dictated by the availability of open water in marine areas en route and adjacent to breeding habitat (Petersen et al. 2000 p. 3).

Timing of Arrival on Breeding Areas

As open water in terrestrial habitat becomes available, breeding pairs move to nesting areas on wet coastal tundra (Petersen et al. 2000 p. 3, Bart and Earnst 2005 p. 88) typically by the second week of May in the subarctic Y-K Delta and the first week of June in the ACP and Arctic Russia (Dau 1974 p. 10, Petersen et al. 2000 p. 3).

Nesting Distribution

In Alaska, spectacled eiders historically nested discontinuously from the Nushagak Peninsula on Bristol Bay north to Utqiagvik, and east nearly to Canada's Yukon Territory, as well as on St. Lawrence Island (58 FR 27474). Two breeding populations representing the highest nesting densities of spectacled eiders in Alaska are recognized: the Y-K Delta and ACP breeding populations. On the Y-K Delta, spectacled eiders now occur from Kwigillingok to Uwik Slough, mostly concentrated within 15 km of the coast from Kigigak Island north to Kokechik Bay (66 FR 9146). The coastal fringe of the Y-K Delta is the only subarctic breeding habitat where spectacled eiders occur at high densities. Spectacled eiders occur at lower densities on the ACP, which lies above the Arctic Circle, and their current distribution stretches from approximately Pt. Lay to the Canning River (Amundson et al. 2019 fig. A5.9). Within this area, most

spectacled eiders occur between Wainwright and the Colville River, within about 80 km of the coast (Amundson et al. 2019 fig. A5.9).

In Arctic Russia, spectacled eiders nest discontinuously from the Kolyma River to the Yana River Delta in Siberia, with highest nesting densities in the Chaun, Kolyma, Yana, and Indigirka River Deltas (58 FR 27474). Hodges and Eldridge (2001 pp. 127 – 142) described the distribution of spectacled eiders from an aerial survey conducted from 1993 – 1995 in Arctic Russia. The survey included the northern coastal plain between Kolyuchin Bay in the east and the Lena River Delta in the west, a distance of 2,340 km. While the coastal wetland plain extends up to 350 km (217.5 mi) inland, the survey did not go inland beyond 200 km of the coast (Hodges and Eldridge 2001 p. 128). Within that survey area, spectacled eiders were the most abundant waterfowl species, observed in the highest densities in the Indigirka River Delta followed by the Kolyma-Indigirka coast. Few were observed west of the Yana River (Hodges and Eldridge 2001 p. 132).

Nest Initiation

On the Y-K Delta, the mean nest initiation date of spectacled eiders varies among years and is correlated with the timing of snow and ice melt in tundra areas (Grand and Flint 1997 p. 928, Petersen et al. 2000 p. 8). Average annual mean nest initiation date from 1985 – 2014 was May 28 (range May 18–June 7 (Fischer et al. 2018 p. 153). While mean nest initiation dates in recent years have been earlier than average (Fischer et al. 2018 p. 153), a significant trend over time is not evident (Fischer et al. 2018 p. 153). On the ACP, annual mean initiation dates of spectacled eiders at Utqiagvik ranged between June 6 and June 25 from 2010 – 2019 (Safine 2011 p. 22, 2012 p. 22, 2013 p. 22, 2015 p. 18, Graff 2016 p. 19, 2018 p. 20, 2020a p. 19). In Arctic Russia, average nest initiation date at Indigirka was June 22 in 1994 (plus or minus 0.32 days, n = 122; Pearce et al. 1998:113). In 1995, nest initiation date varied by distance from the coast; initiation date was June 22 in coastal sites and June 15 in interior study areas (Pearce et al. 1998:113). At Chaun, median nest initiation dates from 2010 – 2016 ranged from June 10 (2016) to June 17 (2015) (Solovyeva and Solovyev 2010 p. 6, Solovyeva and Solovyev 2011 p. 6, 2013a p. 7, Kokhanova and Solovyeva 2015 p. 7, Solovyeva and Kokhanova 2017 p. 7).

Incubation and Hatch

While some incubation occurs before the clutch is complete, full incubation generally begins after the last egg is laid, and lasts approximately 23 days (Dau 1974 p. 24, Flint and Grand 1999 p. 414, Petersen et al. 2000 p. 11). Males do not incubate, and leave the breeding area by early incubation. One to three days after the last egg hatches, females move their broods from nests to freshwater ponds, usually traveling less than 3 km (1.9 mi), but occasionally up to 13 km (8.1 mi; Petersen et al. 2000 p. 12). Young fledge approximately 50 days after hatch (Petersen et al. 2000 p. 11), and then broods move from freshwater to marine habitats (Petersen et al. 1999 p. 1011).

Molt Migration

Males return to the marine environment after incubation begins, and begin molting primary feathers, rendering them flightless for 20 – 40 days. Migrating spectacled eiders remain primarily offshore (Petersen et al. 1999 p. 1012), and molt migration may be either prolonged or relatively rapid, with

individuals sometimes spending several weeks staging in the Beaufort, Chukchi or Bering seas prior to molt (Petersen et al. 1999 p. 1017). Petersen et al. (1999:1017) suggested that molt is synchronous among males based on arrival dates of marked birds at the molting area. Females that are unsuccessful at nesting move to molting areas in July, and successful females move to molting areas in August and September (Petersen et al. 1999 p. 1013). When moving between nesting and molting areas, spectacled eiders travel along the coast up to 60 km (37.3 mi) offshore (Petersen et al. 1999 p. 1013). Molting flocks gather in relatively shallow coastal water, usually less than 36 m (118.1 feet) deep.

Molting Distribution

After breeding, spectacled eiders molt in several discrete areas, with birds from the different populations and genders favoring different molting areas (Petersen et al. 1999 p. 1009, Sexson et al. 2016 pp. 466–467). Four late summer and fall molting areas have been identified: (1) eastern Norton Sound in the northern Bering Sea; (2) Ledyard Bay in the eastern Chukchi Sea; (3) Mechigmenskiy Bay off the coast of Russia in the western Bering Strait; and, (4) the Indigirka-Kolyma River Deltas in the East Siberian Sea (Petersen et al. 1999 p. 1012; Figure 1). Adult females marked on Y-K Delta breeding areas molt primarily in Norton Sound (Petersen et al. 1999 p. 1013, Sexson et al. 2014 p. 39), and adult females marked on ACP breeding areas molt in Ledyard Bay (Petersen et al. 1999 p. 1013, Sexson et al. 2016 p. 39). Males using the same breeding areas, however, did not always use the same molting area as females from that breeding area. Males from the Y-K Delta tended to use Mechigmenskiy Bay or Indigirka-Kolyma for molt; those using ACP breeding areas could be found in one of three molting areas: Norton Sound, Indigirka/Kolyma, or Mechigmenskiy Bay (Sexson et al. 2014 p. 62).

A comparison of satellite transmitter data collected from the 1990s and 2008 – 2011 showed that marked spectacled eiders during the latter period used the same general molting areas identified in Petersen et al. (1999 p. 1012), but the core distribution within each area had shifted (Sexson et al. 2016 pp. 468 – 470).

Winter Distribution

Based on data collected in the 1990s, after molting, spectacled eiders migrated in October and November to the central Bering Sea south-southwest of St. Lawrence Island (Petersen et al. 1999 pp. 1014 – 1015, Petersen and Douglas 2004 p. 84). This area is characterized by high microbenthic community biomass (Grebmeier et al. 2015 p. 102) and persistent openings in the sea ice (polynyas)(Grebmeier and Cooper 1995 p. 1440) that provide foraging habitat for spectacled eiders. In the mid to late 1990s, spectacled eiders were observed in large flocks in sea ice leads and polynyas through March or April. Locations of marked birds from Arctic Russia, as well as the number of spectacled eiders detected in the wintering area (at least one order of magnitude higher than the Alaska breeding populations combined) indicate that spectacled eiders from all three breeding populations intermixed and wintered in the same area (Petersen and Douglas 2004 p. 83). Spectacled eiders marked with satellite transmitters from 2008 – 2010 also moved to the 1990s wintering area about 70 km southwest of St. Lawrence Island (the area used by marked birds in previous years of satellite telemetry studies) in December, even when sea ice was not fully developed, and remained there through April (Cooper et al. 2013 pp. 39 – 40). In years with high sea ice concentration and southward extent in the

Bering Sea, spectacled eider distribution is thought to be limited to small areas of open water (Petersen et al. 2000 p. 3); sea ice concentration and winter weather severity may also affect demographic rates (Section 2.3.2).

More recent data suggest that the winter distribution of spectacled eiders, and the timing of northward pre-breeding movement, is changing, likely as a result of decreased extent of sea ice in the Bering Sea in late winter. Thirty-nine spectacled eiders were instrumented with satellite transmitters in May 2018 on the Y-K Delta, providing location data during winters 2018 – 2019 and 2019 – 2020. In winter 2018 – 2019, some marked eiders moved north of St. Lawrence Island in February (Figures 3 and 4). Three apparent patterns in late winter (February to April) distribution were observed. Some marked individuals remained in the 1990s wintering area south of St. Lawrence Island (the maximum distance moved was 75 km [46.6 mi] from early winter until spring migration; $n = 5$), some moved closer to the south coast of St. Lawrence Island (maximum distance moved = 87 km [54.1 mi; $n = 12$), and some ($n = 7$) moved north of St. Lawrence Island as far northwest as the coast of Chukotka (max. distance moved was 1085 km [674.2 mi]; USFWS, unpublished data; Figure 3). Northward movements coincided with periods of persistent southerly wind and the resulting northward retreat of sea ice (Figure 4), which occurred earlier compared to winters 1996 – 1999 and 2008 – 2011 when other cohorts of spectacled eiders were tagged with satellite transmitters (USFWS, unpublished data). Sea ice conditions in the Bering Sea in winter 2019 – 2020 differed from the previous year; ice was more consistent in the 1990s wintering area south of St. Lawrence Island. A smaller number of spectacled eiders had functioning PTTs that year relative to 2018 – 2019 ($n = 16$ in November to $n = 2$ in March), but their locations suggested that spectacled eiders remained south of St. Lawrence Island until March, when sea ice began to retreat (Figure 5). Overall, there was little variation in latitude of spectacled eider locations in 2019 – 2020 (Figure 5). Additionally, an aerial survey conducted in March 2020 documented winter flocks in closer proximity to St. Lawrence Island's south and east coasts than in previous years when the aerial survey was conducted (1995 – 1998, 2009 and 2010; Section 3.3.3; Figures 3 and 6), likely due to lower sea ice concentration in these areas. In summary, data collected from 2018 – 2020 suggest that some portion of the spectacled eider population moves north of the 1990s wintering area as early as February following retreat of sea ice to the north, as the lack of sea ice results in the availability of more open-water habitat.

Most of the distribution and movement information on spectacled eiders comes from adult females and males marked on ACP and Y-K Delta breeding areas. We know little about the non-breeding distribution and movements of adult spectacled eiders from the Arctic Russia population, and of spectacled eiders that have not yet reached breeding age. Presumably, juvenile spectacled eiders remain at sea year-round until they attempt to breed at age two or three, as do adult non-breeding birds. Winter distribution during the juvenile period is not well described. The only marked hatch-year spectacled eiders that were still alive and providing locations in their first winter ($n = 3$) wintered with adults in the northern Bering Sea (Sexson et al. 2014 p. 30). However, it has been hypothesized that a significant number of juveniles winter separately from adults. Photographic aerial surveys of the wintering population allowed observers to determine the ratio of white plumaged birds (adult males in third calendar year or older) and brown plumaged birds (males in second calendar year and all females) in

flocks. In 2020, 52.2 percent of birds counted were in white plumage. This is consistent with results of previous surveys that documented a mean of 54 percent of individuals in white plumage (range 53 – 55 percent; Larned and Tiplady 1999 pp. 4 – 5, Petersen et al. 1999 p. 1016, Larned et al. 2012 pp. 7 – 8). The proportion of white plumaged birds is much higher than expected given the known vital rates of the species; a stable population would have a higher number of brown plumaged birds (Flint et al. 2016 p. 23), suggesting that juveniles winter elsewhere. In addition, population modeling indicated that adult female survival was negatively affected by high sea ice concentration in the 1990s wintering area, but first-year survival increased with more severe sea ice concentration (Flint et al. 2016 p. 23). This suggests that sub-adult birds wintered in an area where they are not subject to the same extreme sea ice conditions as adults, and thus may not be present in the 1990s wintering area. Therefore, the most important areas for sub-adults remain largely unknown.

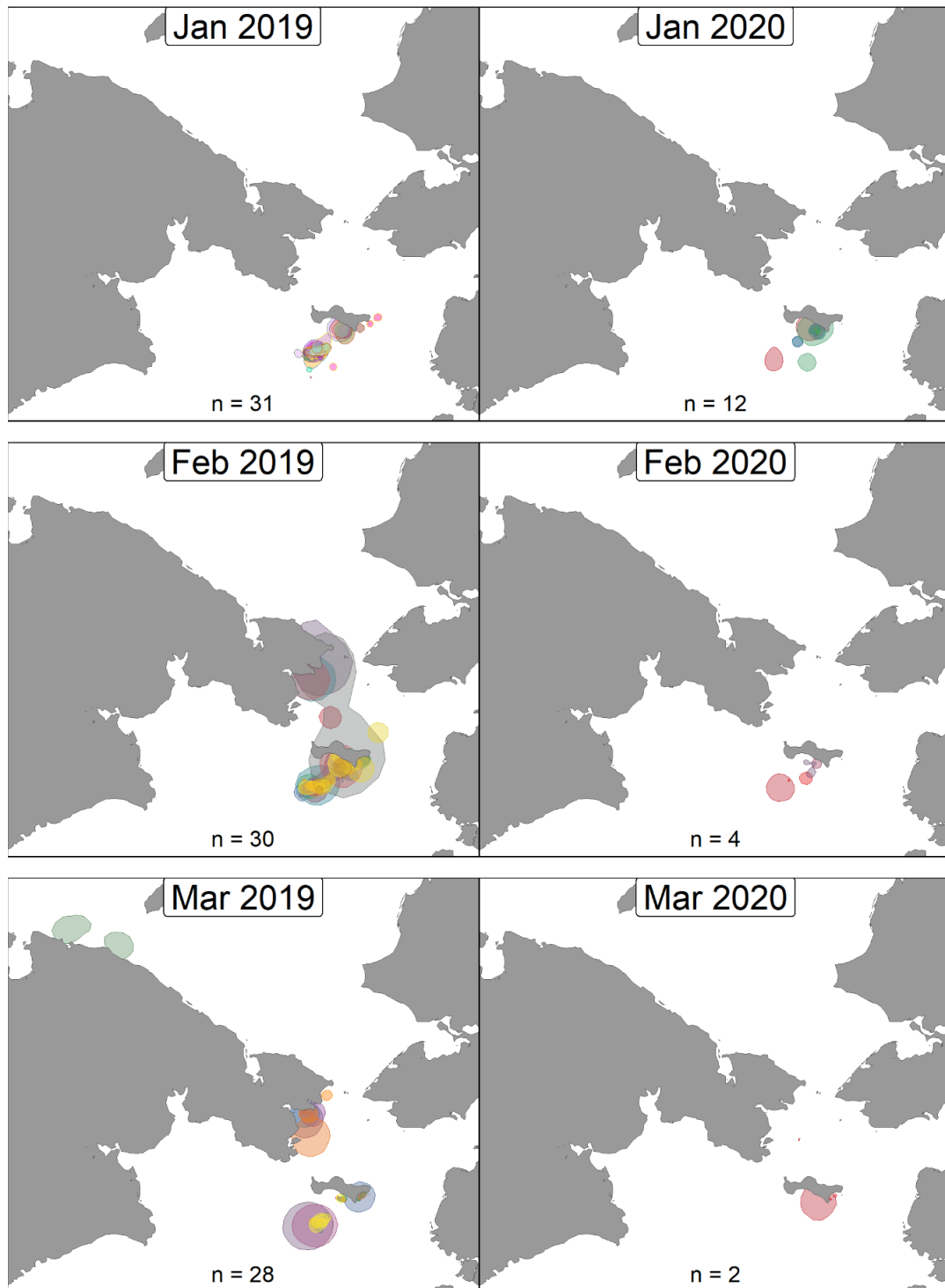
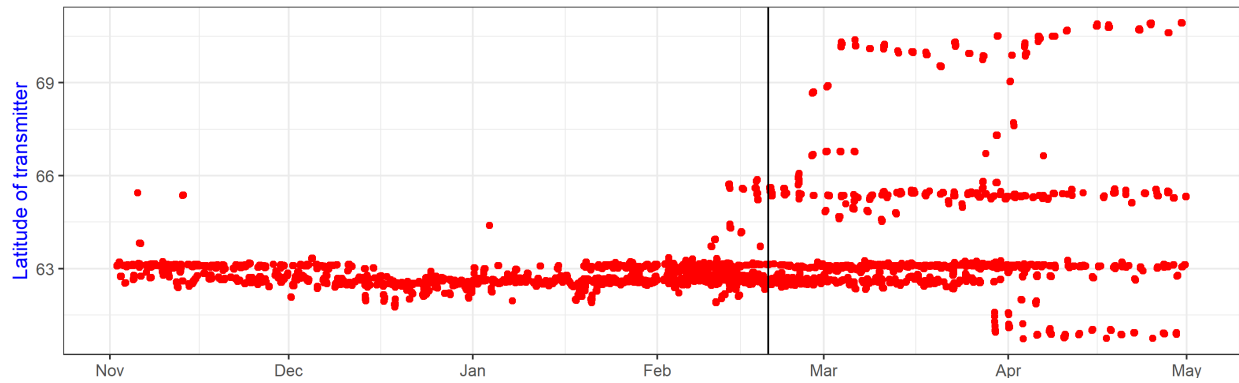


Figure 3. Distribution of marked spectacled eiders in the Bering and Chukchi Seas, January – March, 2019 and 2020 (USFWS, unpublished data). Colors depict individual home ranges (50% contour of kernel density distribution) for each month. Different colors represent different birds.

(a)



(b)

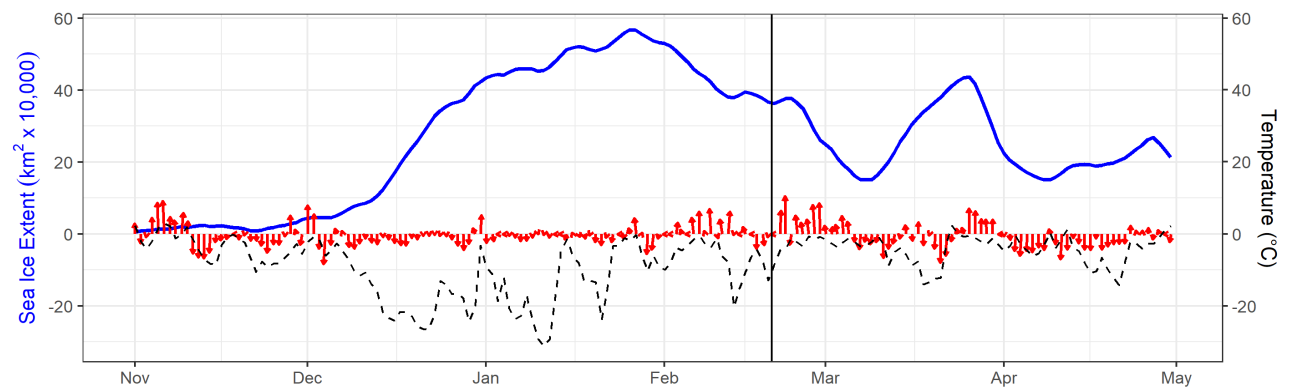
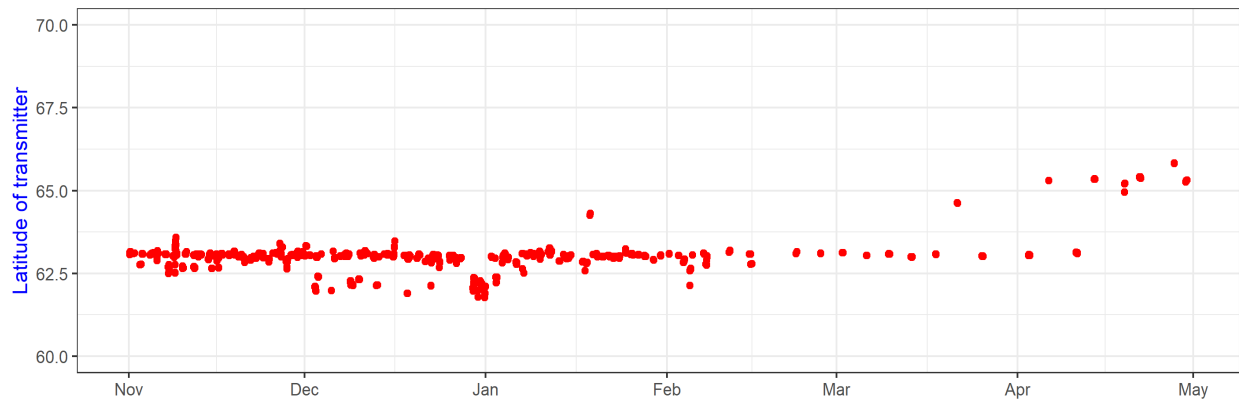


Figure 4. Relationship between latitude of spectacled eiders marked with satellite transmitters and sea ice extent, wind direction, wind speed, and air temperature in winter 2018 – 2019. (a) The latitude of transmitter locations of marked spectacled eiders (red dots) relative to date. (b) Sea ice extent (blue line), wind direction and speed (red arrows; arrow direction is wind direction from, length of arrow is wind speed), and air temperature (dotted line). Vertical line in both (a) and (b) indicates date in which sea ice extent moved north of St. Lawrence Island.

(a)



(b)

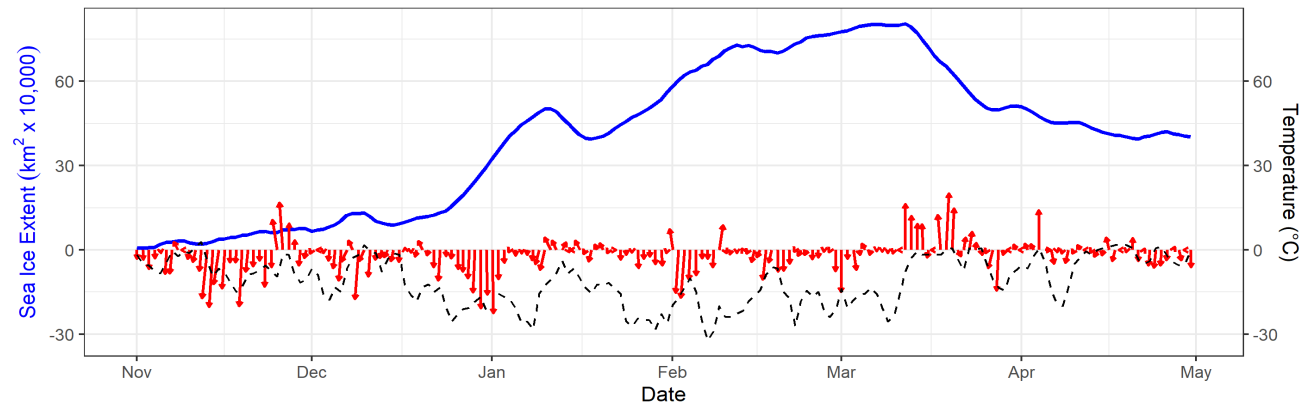


Figure 5. Relationship between latitude of spectacled eiders marked with satellite transmitters and sea ice extent, wind direction, wind speed, and air temperature in winter 2019 - 2020. (a) The latitude of transmitter locations of marked spectacled eiders (red dots) relative to date. (b) Sea ice extent (blue line), wind direction and speed (red arrows; arrow direction is wind direction from, length of arrow is wind speed), and air temperature (dotted line).

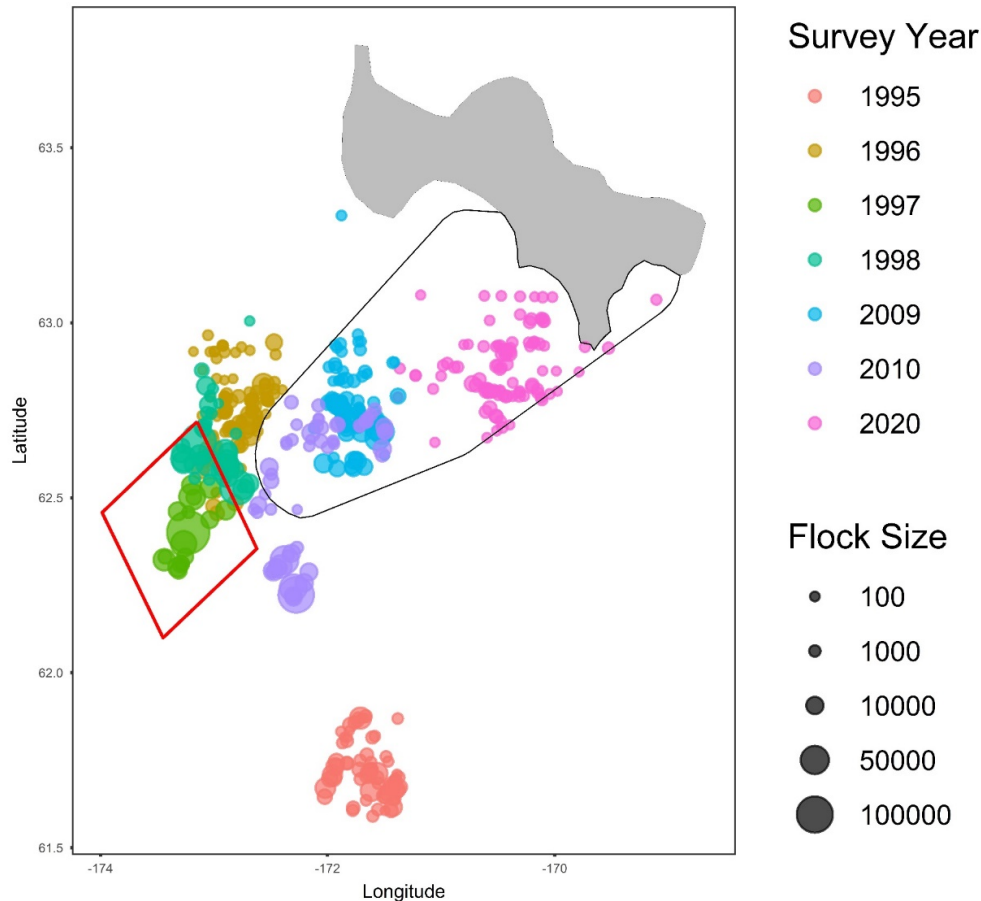


Figure 6. Location and size of spectacled eider flocks observed near St. Lawrence Island in the Bering Sea during winter aerial surveys. Black polygon is the 2020 survey area. Red polygon is the 1990s wintering area described in (Petersen and Douglas 2004 p. 86).

2.2.3. Demographic Rates

Age at First Breeding and Breeding Propensity

Age at first breeding has not been determined for males, but probably occurs at age three or four, coinciding with the acquisition of adult plumage (Petersen et al. 2000 p. 11). Females have been observed breeding first at age two and three. Of spectacled eiders banded as ducklings at Kigigak from 1999 – 2005, Lake (2007:13) observed no one-year old females nesting, but 24.1 percent of banded ducklings were observed nesting at age two. Similarly, no one-year old female spectacled eiders have been observed to nest during long-term field studies at Kashunuk (Flint et al. 2016 p. 24). Flint et al. (2016 pp. 22 – 24) estimated mean breeding propensity of two year old females as 0.29 (95 percent CI = 0.14 – 0.55). Many studies assume breeding propensity of spectacled eiders three years old or older is 1.0 (Flint et al. 2016 p. 24, Dunham 2019 p. 110); however, it is likely that breeding propensity of adult females aged three or older may vary annually due to environmental conditions affecting body condition or other factors, as has been observed with common eiders (Coulson 1984 pp. 531 – 532).

Natal Philopatry and Breeding Site Fidelity

Female-biased natal philopatry (returning to the site of hatch for reproduction) and breeding site fidelity (returning to the same breeding area, even if it is not an individual's birth place) is pronounced in species of sea ducks for which data are available (Eadie and Savard 2015 p. 375). Between 1999 and 2008 at Kigigak, 482 female spectacled eider ducklings were banded, and 64 were observed returning to the study site as adults (Moore and Sowl 2017 pp. 9 – 10). Given rates of duckling and juvenile survival, incomplete detection, and age at first breeding, this is a relatively high return rate and suggests that spectacled eider females have high natal philopatry.

Recaptures of females at the same study site in multiple years suggest spectacled eiders also demonstrate a high degree of breeding site fidelity. Grand and Flint (1998 p. 1107) estimated that at least 83 percent of females returned to nesting areas at Kashunuk each year. At Kigigak, Moran (2000 p. 56) found that females used nest sites located from 0 – 2.3 km (1.4 mi) from nest sites used in previous years (annual medians ranged from 140 to 314 m [459 – 1030 feet]).

Genetic analysis from samples collected from 1994 – 1996 also suggests female-biased natal philopatry. Haplotype frequencies of mitochondrial DNA (mtDNA) from different nesting regions (Y-K Delta, ACP and Arctic Russia) differed significantly, and even differed among females nesting in close proximity within each region, suggesting a high degree of philopatry over relatively short distances (Scribner et al. 2001 p. 2110). These results are consistent with satellite telemetry studies that indicate females return to Alaska breeding areas on which they were marked (Sexson et al. 2014:37; USFWS, unpublished data). Analyses of nuclear DNA from the three breeding populations indicate genetic differences were considerably lower for bi-parentally inherited markers, suggesting that males have lower fidelity to breeding areas (Scribner et al. 2001 pp. 2110 – 2111). This is consistent with the hypothesis that pair bonds are formed in the wintering area, when eiders from all three breeding areas are intermixed, and paired males follow their mate to her natal and/or breeding area.

Analysis of more recent genetic samples collected from 2008 – 2012 also showed population differentiation based on mtDNA, but genetic structure was reduced, particularly in comparisons among Y-K Delta and ACP breeding populations (Sonsthagen et al. 2019 p. 3). These results suggest that female dispersal may have increased with the increase in population size on the Y-K Delta, and lends support to the hypothesis that spectacled eider populations interact as a global metapopulation (Sonsthagen et al. 2019 p. 5). Dispersal between local populations within one breeding area is also supported by population modeling conducted by Flint et al. (2016 pp. 34–36) for the Y-K Delta. Thus, dispersal could occur within or between breeding areas.

Clutch Size

Average spectacled eider clutch size, or the number of eggs laid per nest, varies among years and study sites (Petersen et al. 2000 pp. 10 – 11). Average clutch size at Kigigak ranged from 4.0 – 5.5 from 1992 – 2015 (Gabrielson and Spragens 2013 p. 21, Moore and Sowl 2017 p. 14). Average clutch size at Utqiagvik ranged from 3.6 – 5.1 from 2009 – 2019 (Safine 2011 p. 23, 2012 p. 22, 2013 p. 23, 2015 p. 20,

Graff 2016 p. 20, 2018 p. 21, 2020a p. 20). Average clutch size at Chaun ranged from 4.08 – 5.0 from 2003 – 2016 (Solovyeva and Solovyev 2013b p. 18, Solovyeva and Kokhanova 2017 p. 15).

Nest Success

Nest success, the proportion of nests that successfully hatch at least one egg, varies annually and spatially at both local and regional scales (Flint et al. 2016 p. 17). Nests initiated earlier in the season are more successful than those initiated later (Grand and Flint 1997 p. 929). This may be because older, more experienced females, and/or females with high body condition, which tend to have higher nest success, initiate nests earlier in the season (similar to common eiders; Baillie and Milne 1982 p. 62). Nest mortality is greatly influenced by predators, including gulls, jaegers, and red and Arctic foxes. Spectacled eiders rarely re-nest in the same season after a failed nesting attempt (Petersen et al. 2000 p. 8).

At Kashunuk, nest success varied from 0.04 to 0.92 between 1992 – 2002, and averaged 0.53 (SE = 0.07) (Flint et al. 2016 p. 18). Over the same time period, nest success at Kigigak also varied from 0.04 to 0.92, and averaged 0.73 (SE = 0.06) (Flint et al. 2016 p. 18). However, no general correlation in annual nest success among these two Y-K Delta sites was evident, with the exception that the lowest estimates at both sites occurred in the same year (Flint et al. 2016 p. 17). Nest success estimates from Kigigak in more recent years (2003 – 2015) also varied annually, ranging from 0.05 (in 2015) to 0.91 (in 2007; Gabrielson and Spragens 2013:23, Moore and Sowl 2017:16, Rizzolo et al. 2019:7).

Few studies have estimated nest success of spectacled eider populations on the ACP, and because of lower nest densities, those studies also have small sample sizes. However, the available data suggest that nest success is lower on the ACP than on the Y-K Delta. Nest survival probability at Utqiagvik ranged from 0.19 – 0.72 from 2009 – 2019 (Safine 2011 pp. 23 – 24, 2012 p. 23, 2013 p. 23, 2015 p. 20, Graff 2016 p. 21, 2018 p. 22, 2020a pp. 20 – 21). East of Utqiagvik, mean apparent nest success (number of nests survived/number of nests found) in the Kuparuk oil fields was 0.42 from 1993 to 2007 (range 0.13 to 0.92; Anderson et al. 2007 pp. 12–14) and 0.31 from 1994 – 1999 on the Colville Delta (range 0.11 to 1.0; Bart and Earnst 2005 p. 93).

Studies in Arctic Russia also suggest lower nest success there than on the Y-K Delta. At Indigirka, apparent nest success was less than 0.02 in 1994 and 0.27 in 1995; predation was believed to be the cause of high failure rates, with foxes, gulls and jaegers the suspected predators (Pearce et al. 1998:115). At Chaun, nest success ranged from 0.0 to 0.5 from 2003 – 2016, and declined over that period (Solovyeva and Solovyev 2013b p. 17, Solovyeva and Kokhanova 2017 p. 14, Solovyeva et al. 2018 p. 747). Solovyeva et al. (2018) attributed the decline to increases in the large gull (Vega and glaucous gull) and mammalian predator populations, which depredate spectacled eider eggs, and a concurrent decrease in Sabine's gull and Arctic tern populations that provide some predator protection (p. 748 – 749).

Egg Viability

In addition to egg loss due to predation or abandonment, eggs can be inviable due to infertility, or from embryonic death due to factors such as inadequate incubation conditions, disease, or contaminant exposure. Spectacled eider nesting studies used different metrics to describe egg viability, making it difficult to compare between study sites. From 1969 to 1973 at the Onumtuk study site, northeast of Kashunuk, only 0.7 percent of spectacled eider eggs were addled or infertile (Dau 1974 p. 36). In contrast, at Kashunuk (1992 – 2004) and Kigigak (1992 – 2015), the percentage of spectacled eider nests that contained at least one inviable egg varied from 11 percent to 30 percent among years (Grand and Flint 1997 p. 928, Moore and Sowl 2017 p. 15, Hollmen et al. 2020b p. 1). The average number of inviable eggs in nests that contained any inviable eggs was two. Overall, 7.7 percent of eggs discovered were inviable at these study sites. At Utqiagvik, no inviable eggs were found in half of the years from 2010 - 2019. In the years when inviable eggs were detected, between 6 percent and 20 percent of spectacled eider nests had at least one inviable egg (Safine 2012 p. 23, 2013 p. 23, 2015 p. 20, Graff 2016 p. 21, 2018 p. 22, 2020a p. 21). In Arctic Russia, Pearce et al. (1998b p. 115) found only one inviable egg in 1994 and 1995 at Indigirka. At Chaun, 1.5 – 4.1 percent of spectacled eider eggs per year were inviable from 2003 – 2015 (Kokhanova and Solovyeva 2015 p. 15). Typical nest monitoring methods include candling or floating only a subset of the clutch to estimate incubation stage, which results in some inviable eggs going undetected, particularly in those nests that are later depredated. Variation in field techniques and characteristics of study sites may have influenced detection of eggs. Therefore, the number of inviable eggs reported are likely underestimates of the true rate.

In summary, relatively high rates of egg inviability were observed on Y-K Delta study areas in some years; however, spectacled eider egg viability appears to vary over time and among nesting areas.

Duckling and Brood Survival

Survival of spectacled eider ducklings and broods has been estimated from Y-K Delta and ACP study sites. Duckling survival, defined as the proportion of hatched young that survive until 30 days after hatch, has been shown to vary annually (Flint et al. 2006 p. 906). Flint et al. (2006 p. 907) also found that duckling growth and site-specific survival rates were correlated, suggesting that habitat conditions influence both growth and survival. At the Y-K Delta study sites, Flint et al. (2016 p. 19) estimated duckling survival of 0.39 at Kashunuk River using data collected from 1993 – 2002, and 0.67 at Kigigak Island from two years of data (1999 – 2000).

Brood survival is the proportion of females hatching a clutch that fledged at least one duckling. Mean brood survival across years was estimated as 0.55 (process variation¹ [σ] = 0.001) at Kashunuk and 0.85 (process variation [σ] = 0.009) at Kigigak (Flint et al. 2016 p. 19). Spectacled eider brood survival at Utqiagvik over two years (2011 – 2012, n = 17 broods) was 0.76 (95 percent CRI = 0.63 – 0.82; Rizzolo 2020 pp. 3–5). Survival of juveniles at Kashunuk from 30 days after hatch to departure (1997 – 1999)

¹ Process variation is the component of total variation that explains environmental variation. The other component is sampling variation. Where not explicitly stated, estimates of variation represent total variation.

was estimated at 0.71 (1997 – 1999; Flint et al. 2000:292). Duckling, brood, and juvenile survival have not been estimated for spectacled eiders in Arctic Russia.

Adult Survival

Most models of sea duck population dynamics are female-based, as female vital rates typically determine population dynamics due to females' higher natal philopatry and breeding site fidelity, and lower survival rates, than males (Flint 2015 p. 65). Sensitivity analyses of demographic parameters using demonstrated that spectacled eider adult female survival rate has the greatest effect on population growth rate compared to parameters related to reproduction (i.e., nest survival, breeding propensity, etc.; Flint et al. (2016:28,29,31). Therefore, for the purposes of this SSA, population dynamics models used to estimate viability and discussions of adult survival are primarily focused on females.

Annual survival rates of adult females breeding on the Y-K Delta from 1993 – 1996 varied depending on lead exposure, with females exposed to lead surviving at a lower rate than females not exposed to lead (0.44, SE = 0.01 and 0.78, SE = 0.05, respectively; Grand et al. 1998 p. 1017). This “unexposed” survival rate estimate is biased low because the sample of unexposed females included some females that were exposed to lead after blood levels were sampled. Flint et al. (2016: 25) used a method to correct for this bias, and estimated female survival of unexposed females at 0.88 (process variation = 0.06), and exposed at 0.48 (process variation = 0.14).

Using mark-recapture data from 1992 – 2004 from Kigigak and Kashunuk, Flint et al. (2016 p. 22) also found evidence that winter sea ice severity in the Bering Sea wintering area negatively influences adult female survival by 10 – 22 percent compared to years with relatively mild sea ice conditions (See Section 2.3.2 for a discussion of the mechanism by which sea ice affects survival). In this case, winter sea ice severity refers to a combination of indices that describe the percentage sea ice cover, the total days when sea ice coverage is 95 percent or higher, and the number of consecutive days of 95 percent or higher ice coverage, within the 1990s wintering area (Flint et al. 2016, p. 20). Estimated annual female survival rates varied from 0.63 to 0.94 (mean = 0.78; process variation = 0.01) at Kashunuk, and from 0.55 – 0.95 (mean = 0.81; process variation = 0.01) at Kigigak (Flint et al. 2016 p. 22). The variation in survival rates across the two study sites was greatest during years with more severe sea ice conditions in the Bering Sea; the authors suspect that the increased stressor of severe winter conditions exacerbated the physiological stress of lead contaminated individuals, resulting in higher mortality rates (Flint et al. 2016 p. 22).

Using a longer-term dataset from Kigigak spanning 1992 – 2015, Christie et al. (2018 p. 5) also found high annual variation in adult survival rates ranging from 0.50 (SE = 0.05) to 1.0. While the two most parsimonious models in the explored model set did not include a sea ice severity covariate, the third most likely model included a quadratic relationship of sea ice severity and female survival. An analysis of variance test showed that 44 percent of total deviance was explained by this quadratic relationship, while only 24 percent was explained by a linear relationship (Christie et al. 2018 p. 6). Parameter estimates from this model suggested that survival increased with number of extreme sea ice days (days with greater than 95 percent sea ice coverage in the 1990s wintering area) up to 52 days, flattened out,

then began decreasing at about 80 days of extreme sea ice days. Survival increased with minimal sea ice days (days with < 15 percent of sea ice coverage in the 1990s wintering area) until approximately 90 days, and then declined when there were more than 100 minimal sea ice days per year (Christie et al. 2018 p. 6). However, there is considerable uncertainty in the hypothesized negative relationship between survival and low sea ice extent due to the small number of years with low sea ice extent in the data set, as indicated by wide confidence intervals (Figure 7). Assuming this relationship holds true with a longer-term data set, it suggests that adult female survival is highest when sea ice conditions are moderate.

A separate analysis using an integrated population modeling approach with the same Kigigak mark-recapture data set and additional population count and nest success data also found support for a quadratic relationship between adult female survival and extreme sea ice days (Dunham 2019 p. 73,89). This analysis resulted in a higher estimate of mean adult female survival of 0.87 (95% CI = 0.794 - 0.931), but also showed high annual variation consistent with previous analyses, particularly over the most recent 10 years in the dataset, with annual values ranging from 0.47 to 0.99 (Dunham 2019 p. 71).

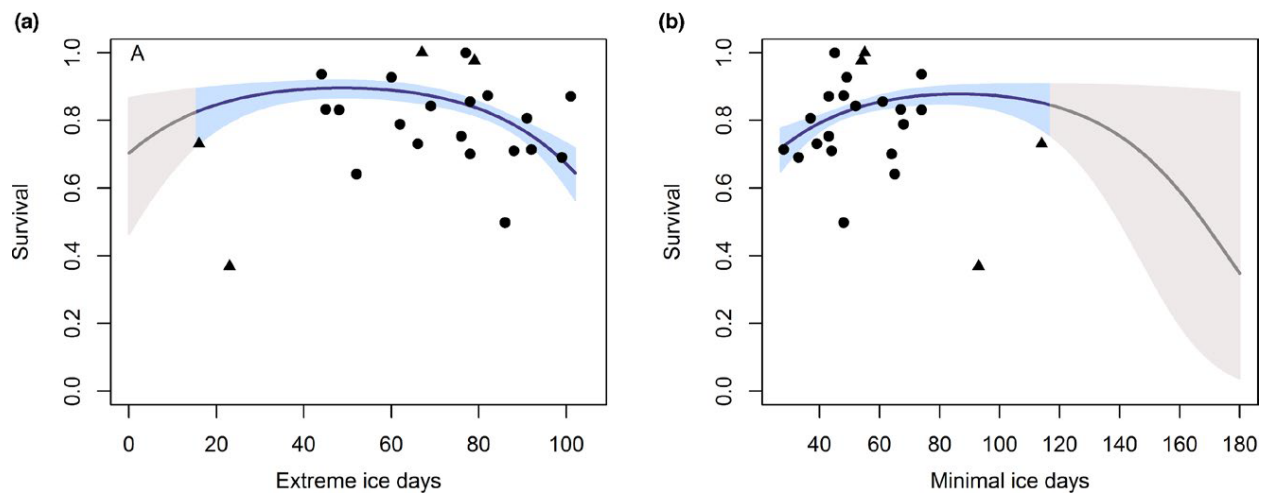


Figure 7. From Christie et al. (2018): Survival and 95% confidence intervals of adult female spectacled eiders as a function of (a) extreme sea ice conditions at the wintering area (number days per year with sea ice concentrations >95%) and (b) minimal ice conditions (number of days per year with sea ice concentrations <15%). The gray line and shading represents predicted survival under sea ice conditions lower than observed during our study period. Black circles indicate survival estimates from the model $p(\text{year} + \text{age})\phi(\text{year}, \text{adults only})$ and black triangles denote survival estimates from the simpler model $p(\text{age})\phi(\text{year}, \text{adults only})$. Both models were used because survival for certain years was not estimated in the fully time-parameterized model.

Solovyeva et al. (2017 p. 5) compared survival rates of spectacled eiders at Kigigak and Chaun from 2002 – 2015, and found no evidence that survival rates differed between sites, adding support to the hypothesis that survival is influenced more by winter conditions rather than conditions on the breeding grounds. However, the low sample size at Chaun may have limited the ability to detect differences between sites, particularly given the high annual variation in survival. From this analysis, estimated

annual female survival across all years (2003 – 2015) was 0.77 (SE = 0.04) at Chaun and 0.82 (SE = 0.01) at Kigigak (Solovyeva et al. 2017:5). We have no information about survival rates from the ACP breeding population of spectacled eiders.

In summary, survival of adult females is a major driver of spectacled eider population dynamics. Survival rate varies annually, and data suggest it may be primarily influenced by winter sea ice conditions, and in some breeding areas, exposure to lead.

First-year Survival

Using data from Kigigak and Kashunuk, Flint et al. (2016 p. 22) estimated first year female survival as 0.22 (95 percent CI = 0.15 – 0.31) and 0.27 (95 percent CI = 0.19 – 0.37), respectively, between 1992 – 2004. Similarly, using mark-recapture data from Kigigak from 1992 – 2015, Christie et al. (2018 p. 5) estimated first year female survival as 0.22 (SE = 0.02). Using an integrated population model based on mark-recapture data from Kigigak and count data from aerial surveys, Dunham (2019) estimated mean first-year survival of spectacled eider females as 0.31 (95 percent CI: 0.125 – 0.578).

Flint et al. (2016 p. 23) found support for a positive relationship between first-year survival and sea ice concentration, causing the authors to hypothesize that first-year birds winter outside of the 1990s wintering area (see also Section 2.2.2). This is in contrast with results from Dunham (2019), which did not find strong support for a positive relationship between sea ice concentration and first-year survival.

2.3. Habitat Requirements

In this section we summarize the available information on the habitat characteristics that individual spectacled eiders require to survive and reproduce. The availability of these requirements affects populations' demographic rates and probabilities of extinction.

2.3.1. Tundra Breeding Habitat

Diet and foraging behavior

No systematic sampling of spectacled eider diet has been conducted, and what is known about diet in the breeding areas is based on foraging observations, measurements of food availability, and limited collections of adult and juvenile eiders. During the breeding season, adults and ducklings feed in freshwater ponds and wetlands, eating aquatic insects, crustaceans, and vegetation, primarily from the surface (Dau 1974 p. 43, Petersen et al. 2000 p. 5). Stomach contents collected from a small sample of spectacled eiders upon arrival at Indigirka consisted primarily seeds of Pallas' buttercup (*Ranunculus pallasii*) (Kitschinski and Flint 1974 pp. 12 – 13). As ponds thawed, the bulk of stomach contents consisted of crane fly (*Tipulidae*) and caddisfly (*Trichoptera*) larvae, and the invertebrate species consumed diversified throughout the season to include several invertebrate species, including midge (*Chironomidae*) larvae (Kitschinski and Flint 1974 p. 13). Seventy percent of stomach contents from three adult spectacled eiders collected on the Y-K Delta prior to pond thawing contained crane fly larvae (Dau 1974 p. 43). Also on the Y-K Delta, nesting females were observed feeding on emerging midges

and females and juveniles were observed foraging on crowberries (*Empetrum nigrum*) from mid-August to departure from breeding areas (Dau 1974:43) .

Seeds, primarily from mare's tail (*Hippuris sp.*) and pondweed (*Potamogeton sp.*), comprise the largest proportion of total food items available at Kashunuk (O'Connell 2001 p. 19). These food items are likely important to spectacled eiders early in the season when invertebrate availability is still low due to snow and ice cover, and they are a more stable resource throughout the breeding season, whereas invertebrates change in abundance (O'Connell 2001 p. 26). Midge larvae are the most abundant invertebrate available for spectacled eiders over the entire breeding season at this site (O'Connell 2001 p. 22).

Duckling diet likely consists of the same foods consumed by adults in late summer, primarily aquatic invertebrates (Dau 1974 p. 41, Graff 2020b p. 1). Based on observations at Kigigak Island, ducklings forage on the aquatic invertebrates that are most available; in the benthos, the most common species include midges and worms (*Annelida*). Zooplankton are also consumed by ducklings, especially in very shallow pools where zooplankton are highly concentrated (Graff 2020b p. 1)

Nesting Habitat

Throughout their range, spectacled eiders nest in wet coastal tundra, typically less than 3 m from freshwater or brackish ponds or lakes. Favored nest sites in all three breeding areas include shorelines of lakes and ponds, small islands and peninsulas, polygon rims, and dry areas in wet meadows (Petersen et al. 2000 p. 9).

Saalfeld et al. (2017 fig. 7) modeled nest habitat associations for spectacled eiders on the Y-K Delta, and found that spectacled eider nest density is greatest within 10 km of the coast but can occur as far as 15 km inland. Other variables important to predicting spectacled eider nest density on the Y-K Delta included the percent coastal graminoid habitat, percent lower coastal salt marsh habitat, and shoreline complexity (all positive relationships; Saalfeld et al. 2017:10). At Kashunuk, spectacled eiders nest primarily in two community types: intermediate-sedge meadow and high-sedge meadow; both community types are generally flat, occasionally flooded by tides, have shallow ponds, with *Carex ramenskii*, *Calamagrostis deschampsoides*, *Elymus arenarius*, and *Poa eminens* as dominant terrestrial vegetation in intermediate-sedge meadow and *Carex canescens* and *Potentilla egedii* also present in the high-sedge meadow (Grand et al. 1997 p. 1201, O'Connell 2001 p. 11). *Carex spp.* were the most abundant plant species in and around spectacled eider nests at Onumtuk, another Y-K Delta study area (Dau 1974 p. 12).

On the Colville Delta, pre-breeding spectacled eiders disproportionately use small, deep, polygonal ponds with relatively narrow vegetated rims and sometimes with islets. This habitat type occurred in 2.7 percent of the study area but was used by 28 percent of spectacled eiders observed (Johnson et al. 2018 p. 11).

In Russia at Indigirka, spectacled eider nests were found in two types of habitat: 1) on small islands or shorelines in interior wetland complexes of large lake basins; and 2) in coastal wet tundra, where nests were associated with structural habitat features such as polygon rims, small islands, or wetland shorelines. Interior nests were almost exclusively found within glaucous and herring gull colonies, and vegetation height and density were more important predictors of nest sites than on the coast, where they were not associated with nesting gulls (Pearce et al. 1998b p. 115,120). Kitschinski and Flint (1974 pp. 7–9) also suspected an association with gull and tern colonies at interior nesting areas at Indigirka. At Chaun, most spectacled eiders nest on islands and shores of brackish tundra lakes, typically less than 1 m from water (Solovyeva and Solovyev 2013b p. 3,7). Solovyeva et al. (2018) suggested a nesting association between spectacled eiders and Sabine’s gulls and Arctic terns.

Brood-rearing Habitat

Limited data exist on the important characteristics of brood-rearing habitat. Brood observations at Kigigak in 2011 and 2012 suggest that broods primarily use wetlands containing many asymmetrical, interconnected ponds surrounded by mixed dwarf shrub/graminoid habitat, although habitat use characterized by the surrounding vegetation type varied by year. Although some broods used large, deep ponds at the highest elevation surrounded by dwarf shrub habitat, this habitat type was not used more often than other types despite it containing ponds with the lowest salinity levels (N. Graff, USFWS, pers. comm).

In addition, quality brood rearing habitat contains wetlands with emergent vegetation for both feeding and hiding from aerial predators. Pond depth is also important as ducklings can get stuck in ponds when water levels are too low, and invertebrate communities can be affected by water levels (D. Safine, USFWS, pers. comm.)

Fresh water

Adult spectacled eiders have supraorbital salt glands that excrete excess salt, allowing them to live in marine environments with limited freshwater. However, salt glands of spectacled eider ducklings are not fully functional at hatch. In common eider ducklings, a closely related species, salt glands begin to function at two days post-hatch at the earliest (DeVink et al. 2005 p. 527). Survival and growth rates of common eider ducklings are negatively correlated with salinity in available water sources (DeVink et al. 2005 p. 526), suggesting that sources of fresh water, at least for the first few days of life, are important for spectacled eider ducklings.

In a controlled laboratory experiment to determine consequences of saline water exposure, spectacled eider ducklings were exposed to saline concentrations of 3 and 6 parts per thousand (ppt; Hollmen et al. 2020a p. 1). No clinical abnormalities were observed in ducklings exposed to 3 ppt, but physiological and behavioral pathologies at exposure levels of 6 ppt were observed, suggesting decreased survival at exposure to 6 ppt salinity (Hollmen et al. 2020a p. 1). Salt glands of ducklings in the trial began functioning at one week of age (Hollmen et al. 2020a p. 1).

At Kashunuk, broods frequently travel up to 8 km (5 mi) in the first week after hatching to less saline habitats; in contrast, at Kigigak where salinity is lower, many broods remain in ponds next to nest sites (Flint et al. 2006:909). There are limited data on habitat use relative to salinity for very young spectacled eider broods (1 – 3 days old). However, in 2011 and 2012 at Kigigak, some young ducklings (< 1 week old) were observed using ponds with salinity levels associated with impaired survival and body condition of common eiders and spectacled eiders in the captive environment (Graff 2020b p. 1). Broods of all ages did not seem to select ponds based on salinity (Graff 2020b p. 1).

On the ACP, wetlands used by spectacled eider broods were fresh water, as the maximum salinity measured was < 0.5 ppt (Safine 2012 p. 31); thus, at this time salinity is unlikely a stressor for spectacled eider ducklings in the ACP breeding population.

2.3.2. Marine Habitat

Physical Characteristics

Spectacled eiders molt in shallow coastal water with sandy-gravelly substrates, typically less than 18 m (59 feet) deep (Petersen et al. 2000 p. 5, Savard and Petersen 2015 p. 323). Migration corridors between nesting, molting, and wintering areas occur over marine waters up to 60 km offshore of Alaska and northeastern Russia. Use of spring migration corridors and wintering areas may be limited by openings in sea ice (Cooper et al. 2013 p. 42, Lovvorn et al. 2015 p. 168).

In winter, spectacled eiders use marine habitats in the Bering, Beaufort and Chukchi seas with water depth less than 80 m (262 feet; Petersen et al. 2000 p. 5); mean water depth of winter locations of marked spectacled eiders from 2009 – 2011 was 42 m (138 feet; Cooper et al. 2013 p. 39). From November through April, spectacled eiders use areas in the open ocean, polynyas (open water at predictable, recurrent locations in sea ice covered regions), or open leads (ephemeral breaks in the sea ice, often along coastlines) in the sea ice of the northern Bering Sea and southern Chukchi Sea at water depths of less than 80 m (262 feet; Petersen et al. 2000 p. 5; USFWS, unpublished data).

Diet and foraging

Spectacled eiders feed primarily on benthic invertebrates when inhabiting marine areas. Diet composition in the marine environment has been directly measured only a handful of times by evaluating digestive contents of collected birds. Bivalves, particularly *Macoma spp.*, were found to be the primary prey of 36 spectacled eiders collected in May-June 1987, 1992, and 1995 at St. Lawrence Island; they also consumed gastropods, crustaceans and fish in smaller quantities (Petersen et al. 1998 pp. 125–126). Digestive tract contents of two spectacled eiders collected from the Pribilof Islands in 1939 consisted of amphipods and snails (Lovvorn et al. 2003:263).

Digestive tracts of 43 spectacled eiders collected in the Bering Sea wintering area in late March 2001 also primarily contained bivalves, but contents almost exclusively consisted of *Nuculana radiata* (Lovvorn et al. 2003 p. 261). Benthic samples showed that other species of invertebrates were available in the substrate, but *N. radiata* was the most abundant bivalve, and spectacled eiders consumed the

most abundant size range of *N. radiata* available (mostly 18 – 24 mm long) (Lovvorn et al. 2003 p. 261, Richman and Lovvorn 2003 p. 289). This suggests that spectacled eiders consume prey species and prey sizes based on availability. In March 2009, 42 spectacled eiders were also collected in the wintering area; however, none had food in their upper digestive tract (esophagus and gizzard). Diet was inferred by analyzing the contents of the lower intestinal tract, stable isotope ratios in muscle tissue, and fatty acids in subcutaneous adipose tissue. Similar to 2001, bivalves constituted the bulk of the collected eiders' diet, including both *N. radiata*, and to a lesser extent, *Ennucula tenuis* (Lovvorn et al. 2014 pp. 400 – 401).

Daylight and sea ice may constrain spectacled eiders' ability to forage effectively in winter. Given the high latitude of the wintering area, the winter months have few hours of daylight. Most sea ducks are thought to be diurnal foragers (Zydelis and Richman 2015:245 – 246), but it is unknown if spectacled eiders are capable of foraging in the dark. In addition to a possible reduction in time available for foraging during winter days, sea ice may constrain spectacled eiders' ability to access preferred food resources by covering high quality foraging areas (see Moderate Sea Ice section below).

Molt is an energetically-demanding process. Nutritional requirements of spectacled eiders during molt are unknown; feeding habits and diet composition during molt have not been studied. In addition, it is unknown if spectacled eiders rely on stored nutrient reserves during molt or if they require a certain amount or type of food to fuel feather growth. Studies of other species show that sea ducks typically forage on the most abundant, accessible food during molt (summarized in Savard and Petersen 2015:317 – 319). Most sea ducks spend more time resting during molt than during other times of the annual cycle and reduce their foraging time, perhaps to protect emerging feathers and reduce energetic demand by diving less frequently (Savard and Petersen 2015 p. 319).

Similar to molt, food requirements during migration and at staging/stopover areas between breeding, molting, and wintering areas have not been studied.

Moderate Sea Ice Concentration in Winter Areas

Sea ice is a defining characteristic of spectacled eider wintering habitat and may be important for energy conservation. Spectacled eiders forage by diving in sea ice openings, and use the sea ice as a roosting platform when resting. Energetic models estimate that spectacled eiders use 1.3 and 1.9 times more energy when resting and swimming in water at ambient air and water temperatures, respectively, than when roosting on sea ice (Lovvorn et al. 2009 p. 1607). Sea ice also dampens wave action, reducing energy requirements. A recent analysis of mark-recapture data in relation to sea ice concentration suggest that extremely low sea ice extent in the wintering area, which has occurred more frequently in recent years, may result in lower adult female survival rates (Christie et al. 2018 pp. 6 – 7). However, the mark-recapture time series used only included two years of low sea ice concentration (defined as greater than 100 days of sea ice coverage of less than 15 percent in a year), resulting in a high uncertainty in this relationship (Christie et al. 2018 p. 8).

Conversely, a strong negative effect of continuous periods of heavy sea ice concentration in the wintering area on spectacled eider survival has been demonstrated using a longer time series with years of extreme and moderate sea ice conditions (Flint et al. 2016 pp. 21 – 22, Christie et al. 2018 p. 7). Reduced survival in years of extreme sea ice conditions is hypothesized to be a result of poorer body condition caused by inadequate forage availability due to sea ice coverage over preferred foraging areas. While not thoroughly tested, this hypothesis is supported by measures of body condition of spectacled eiders collected in March 2001 and 2009. Body fat of eiders collected in 2009 (a year with high sea ice coverage) was 35 percent lower in adult males and 33 percent lower in adult females than birds collected on the same date in 2001 (a year with relatively low sea ice coverage; Lovvorn et al. 2014:402 – 403). Higher water content in muscle and organ tissues in birds collected in 2009 also indicated that eiders started catabolizing body protein to meet energy needs.

Researchers have also hypothesized that extreme weather and sea ice conditions affect breeding propensity (the proportion of the overall breeding population that attempts to nest in a given year) of spectacled eiders. Using data from 1988 – 2002, Petersen and Douglas (2004 pp. 87 – 88) suggested that the number of days with extreme sea ice ($\geq 95\%$ ice concentration), average minimum daily temperature during April, and average daily wind speed in April in the Bering Sea 1990s wintering area were negatively associated with annual breeding population estimates of spectacled eiders on the Y-K Delta the following year. However, in a similar analysis that includes a longer time series (1988 – 2010, 2012 – 2018), the associations between Y-K Delta breeding population abundance and weather and sea ice variables were not apparent (Rizzolo 2020b pp. 6–9). This analysis did find strong evidence that abundance of the breeding population on the ACP (1992 – 2016) was lower following winters with high average wind speed, but no relationship between sea ice concentration and abundance of the ACP population was evident.

In summary, ideal habitat conditions in the wintering area may include moderate sea ice concentration and mild winds, but additional study is needed to understand this relationship fully.

2.4. Population Needs

For this SSA, we define populations of spectacled eiders by breeding area (Y-K Delta, ACP and Arctic Russia), as female spectacled eiders have high natal philopatry and breeding site fidelity to these areas (supported by analysis of mtDNA; Section 2.2.2), and the breeding populations are geographically separate (Figure 1). Highly resilient populations are able to recover from environmental stochasticity (e.g., normal variation in weather) and demographic stochasticity (e.g., normal variation in reproductive rates). A highly resilient population has a high population abundance and a stable or increasing growth rate.

2.5. Species Needs

In order to adapt to changing environmental conditions, species need to maintain adequate representation and redundancy.

Representation refers to the ability of a species to maintain ecological and genetic diversity, which allows the species to adapt to changing environmental conditions. It can be measured by the distribution of the species across ecotypes, and through analyses of gene flow and genetic diversity. Historical distribution of spectacled eider breeding populations spanned three ecotypes: subarctic, tidally-influenced tundra in western Alaska, Arctic tundra in northern Alaska, and Arctic tundra in northeastern Russia. Arctic tundra habitat used by spectacled eiders includes both large river deltas (e.g., Lena River Delta in Russia and Colville River Delta in Alaska) and the coastal tundra that occurs between such deltas. Russia and Alaska Arctic tundra breeding areas have a slightly different assemblage of avian and mammalian species and vegetative communities (Section 2.2.1.). To ensure spectacled eiders have the ability to adapt to environmental change, the species must be comprised of widely distributed, resilient breeding populations in all three large ecotypes (subarctic Alaska, Arctic Alaska and Arctic Russia), and have adequate genetic diversity and behavioral plasticity (e.g., the ability to shift to different prey species or habitats) (Table 1).

Redundancy refers to the ability of a species to withstand catastrophic events which can be accomplished by having multiple, resilient populations with wide distributions. This allows the species to persist if one or more breeding populations is extirpated due to a catastrophic event such as a disease outbreak or oil spill. Similarly, spectacled eiders must use multiple molting and wintering areas or have a wide distribution during fall molt and winter to withstand catastrophic events. Historically, spectacled eiders occurred in three breeding areas (described above in Section 2.2.2.). Similarly, spectacled eiders have historically used four molting areas; these must each continue to be available, and with adequate habitat, and used, to maintain high redundancy. In the wintering area, particularly in years with high concentrations of sea ice, spectacled eiders can be restricted to limited openings in the sea ice south of St. Lawrence Island, making them vulnerable to catastrophic events. In years with low sea ice concentration, however, observations of marked spectacled eiders suggest they have a much larger distribution (USFWS, unpublished data; Section 2.2.2.), thereby reducing their vulnerability to catastrophic events.

Table 1. Requirements of individual spectacled eiders, spectacled eider populations, and the species as a whole, that affect viability.

Level	Requirement	Life stage affected/Why it is needed	Association with 3 Rs
Individual	Food in tundra breeding habitat: freshwater aquatic invertebrates and vegetation	duckling survival adult survival breeding propensity	Resilience: availability of these

	Nest habitat: wet coastal tundra within 3m of shallow ponds or lakes	breeding propensity egg survival	requirements affects demographic rates and probability of extinction
	Brood-rearing habitat: vegetated areas close to ponds, emergent vegetation, adequate pond depth	duckling survival	
	Fresh water in brood-rearing areas	duckling survival	
	Molting habitat: shallow coastal water, typically less than 36m deep	juvenile survival adult survival	
	Migration corridors to/from molting, wintering and breeding areas: coastal habitat up to 60km offshore with open water leads	juvenile survival adult survival	
	Winter habitat: marine waters with depth less than 80 m	juvenile survival adult survival breeding propensity	
	Food in molt, staging, and wintering areas: marine benthic invertebrates	juvenile survival adult survival breeding propensity	
	Moderate sea ice concentration in wintering area	juvenile survival adult survival breeding propensity	
Population	Stable or increasing population growth rate	Enables the population to respond and recover from stochastic disturbance	Resilience
	High population abundance	Enables the population to respond and recover from stochastic disturbance	Resilience
Species	Multiple, resilient breeding populations with wide distribution	Ability to persist if one or more breeding populations are extirpated	Redundancy
	Use of multiple molting and wintering areas or wide distribution in each	Ability to persist if one or more molting or wintering areas are no longer available, or if a catastrophic event occurs	Redundancy
	Breeding populations distributed across the species range and over multiple ecotypes	Maintains breadth of ecological adaptations to breeding environments	Representation
	Genetic diversity	Maintains ability to adapt to environmental changes	Representation
	Demonstrates plasticity in feeding behaviors and habitat use	Maintains ability to adapt to environmental changes	Representation

We considered what is known about historical distribution and habitat use to establish a baseline for evaluations of current and future representation and redundancy in following sections of the SSA.

3. Current Condition

3.1 Factors Affecting Viability

The following discussion summarizes the factors hypothesized to affect the current and/or future condition of spectacled eiders throughout some or all of the species' range. We began our analysis by reviewing threats described in the listing document, the 1993 Recovery Plan, the 2010 5-year review, Biological Opinions developed under Section 7(a)(2) of the ESA, and other available scientific information, to determine if they still remain important factors affecting viability. For each factor we describe what is known about how it currently affects spectacled eiders, and whether it may affect the species in the future.

3.1.1. Ingestion of Lead Ammunition

Historically, hunters have used lead ammunition (shotgun pellets) for hunting and shooting within spectacled eider nesting, brood-rearing, pre-migration staging, and migration areas. Presently, the illegal use of lead shotgun pellets for hunting continues to an unknown degree. Spectacled eiders ingest lead pellets when feeding and collecting grit in freshwater and brackish wetlands (Franson et al. 1995 p. 268, Flint et al. 1997 p. 440, 2016 p. 5). Ingestion of lead pellets has been identified as the source of high and harmful lead levels in spectacled eiders through blood samples, radiographs, necropsy, and lead isotope analysis (A. Matz, USFWS Biologist, pers. comm., Franson et al. 1995:268, Flint et al. 1997:440).

The effects of lead exposure on individuals and breeding populations are well-documented in waterfowl. Toxic effects of ingesting lead vary among individuals, but include lethal and sub-lethal effects (Jordan and Bellrose 1951 pp. 4 – 5, 15 – 17, Franson et al. 1995 p. 175, Baldassarre and Bolen 2006 p. 295). Sub-lethal effects include lower fecundity (Elder 1954:321), increased risk of power line collisions at intermediate lead exposure levels (Kelly and Kelly 2005 p. 333), and reduced immunosuppression (Franson 1986 p. 107). Lead exposure has been shown to kill spectacled eiders and to have measurable negative effects on annual survival rates of spectacled eider populations (Grand et al. 1998 p. 1106, Flint et al. 2016 pp. 24 – 25). The probability of exposure to lead is higher for females than males or juveniles, as they remain on the breeding grounds for a longer period of time (Flint et al. 1997 p. 440). Exposure probability is greatest closer to populated areas (e.g., villages) and areas used for travel and subsistence activities (i.e., coastlines, rivers, fishing and hunting camps; Flint et al. 2016:14).

Waterfowl hunting with lead shot has been prohibited in Alaska since 1991. The USFWS intensified efforts in 1998 to enforce prohibitions against the possession and use of lead shot for migratory bird hunting. Later, the State of Alaska, at the request of regional advisory boards, passed more restrictive

regulations that prohibit the use of lead shot for upland game bird hunting on the ACP and all bird and small game hunting on the Y-K Delta. There are indications that compliance with these regulations improved as a result of significant outreach and education efforts conducted in the late 1990s and early 2000s. However, compliance varies spatially and temporally and lead shot is still occasionally available for sale in villages within spectacled eider habitat. Hunters have been found in possession of lead shot, indicating that this factor has not been eliminated (USFWS, unpublished observations). In addition, embedded lead shot was detected in one of 41 spectacled eiders caught during research activities in May 2018 on the Y-K Delta (USFWS, unpublished observations) indicating that it is still being used for hunting. In response to observations in 2017 that lead ammunition used for waterfowl hunting was still being sold in villages near spectacled eider habitat, USFWS and partners increased outreach efforts to reduce the sale of such ammunition on both the ACP and Y-K Delta.

The risk of exposure likely differs for each of the three spectacled eider breeding populations. Lead exposure of spectacled eiders has been well documented on the Y-K Delta (Flint et al. 2016 pp. 4 – 5). Lead shot can be available for ingestion in ponds on the Y-K Delta for many years after deposition (more than 25 years, Flint and Schamber 2010:149); thus, shot distributed in the past can still affect birds. In addition, as explained above, there are indications that lead shot is still being used to some degree on the Y-K Delta. Therefore, lead exposure remains a possible stressor to the Y-K Delta breeding population of spectacled eiders.

On the ACP, breeding spectacled eiders are distributed more widely and occur at lower densities than on the Y-K Delta, such that fewer breeding spectacled eiders use areas with human activity, where lead could be deposited. While stores on the ACP have also sold lead shot in recent years, the USFWS has observed this less frequently than in the past, and continues to check stores annually and successfully communicates with store managers about alternatives (USFWS, unpublished observations). There is little hunting activity within areas developed for oil and gas extraction on the ACP, such as Prudhoe Bay and areas east of NPR-A; waterfowl sampled in these areas have lower exposure rates than those with more hunting activity (Wilson et al. 2004 p. 408). Additionally, spectacled eiders sampled in Utqiagvik from 2000-2014 had low levels of lead in their tissues (Miller et al. 2016 pp. 305, 308). Overall, lead exposure is likely lower for spectacled eiders on the ACP than on the Y-K Delta.

In Arctic Russia, the probability of lead exposure for spectacled eiders is unknown. The only ammunition available to Russian hunters is made with lead and there are no prohibitions on its use (D. Solovyeva and S. Rozenfeld, pers. comm.); therefore, it is likely that lead exposure occurs wherever spectacled eider distribution overlaps with areas used for hunting. Russia-breeding spectacled eiders are distributed throughout a much larger region than the Y-K Delta and ACP populations (Figure 1; Section 2.2.2). While it is difficult to find demographic information for villages in northeast Russia, the area is sparsely populated and human distribution is likely similar in scale to the ACP, with a few small villages located in the coastal tundra habitat used by spectacled eiders. Spectacled eiders are distributed in the Chukotka and Yakutia (Sakha) Districts, which have had a declining human population resulting in the abandonment of several villages and settlements in the past couple of decades (Kumo 2017 p. 7,14); therefore, lead deposition may be declining over time due to decreased human activity in the region. In

summary, while the likelihood of a spectacled eider encountering lead shot in wetlands in Russia may be lower than on the Y-K Delta because of the small overlap between human and eider distribution, lead exposure is still of concern for the Arctic Russia breeding population because lead is the only type of ammunition known to be used in the region.

Overall, lead exposure rates for each breeding population, and how rates have changed over time, are uncertain, but lead exposure is and will continue to be a factor affecting spectacled eiders until its use is eliminated in all three breeding areas and lead currently in the environment settles into sediment such that it is no longer available to eiders.

3.1.2. Harvest and Shooting

All harvest of spectacled eiders in Alaska was closed in 1991 by both the State of Alaska and USFWS. The USFWS promulgates spring-summer subsistence harvest and fall-winter general hunting regulations that allow for some migratory bird species to be harvested annually (USFWS 2020a p. 5). Included in the subsistence harvest regulations are closures on shooting and collecting eggs of spectacled eiders, and a closure on take of all migratory birds during the 30-day peak nesting period (USFWS 2020a p. 5). An intra-agency consultation under Section 7(a)(2) of the ESA on these regulations includes conservation measures such as law enforcement, outreach, education, and communication programs to minimize shooting of listed eiders and improve compliance with regulations (USFWS 2020a p. 7,33). However, it is unknown how effective the regulations and conservation measures are at minimizing the amount of shooting that occurs, as it is difficult to monitor take (USFWS 2020a pp. 37 – 39).

Recently, in response to an up-listing in the IUCN Red List to “Near Threatened” (BirdLife International 2018), the spectacled eider was included in the Red Data Book of the Russian Federation in March 2020 (D. Solovyeva, 2020, pers. comm.), and in the 2008 Red Book of Chukotka (Russian Academy of Sciences 2008). Therefore the species is legally prohibited from harvest, as is any activity that may result in habitat degradation or a reduction in numbers. The species is also nominally protected under the 1978 U.S. Migratory Bird Treaty Act with the former Soviet Union (Convention between the United States of America and the Union of Soviet Socialist Republics Concerning the Conservation of Migratory Birds and their Environment; Public Law 95–616), which specifies that each party shall prohibit the taking of migratory birds, the collection of their nests and eggs, and the disturbance of nesting colonies. Exceptions include subsistence purposes for indigenous people. We do not have reliable information on the enforcement of such regulations in Russia.

Spectacled eiders may be harvested during migration, during the breeding period on the tundra, or in marine staging/molting areas. During migration, shooting may occur as spectacled eiders fly past coastlines or along open leads in the sea ice where subsistence activities occur. Because they often fly in mixed-species flocks, and are approximately the same size as king eiders, migrating spectacled eiders can be difficult to distinguish from other eiders that are open to harvest. Spectacled eiders may also be harvested at nesting, brood rearing and pre-migration staging areas near villages, hunting camps, and

other areas used by humans. Breeding adult females are at most risk of being shot because they spend more time in these areas than males and juveniles. While spectacled eiders do not nest in high numbers immediately adjacent to human development, at least some portion of the breeding population could be subject to shooting due to their proximity to human infrastructure and hunting areas. Molting and wintering spectacled eiders may also be at risk of inadvertent harvest. Spectacled eiders molting in Ledyard Bay and Norton Sound may be taken during the course of other subsistence activities (e.g., marine mammal hunting) in July and August. However, during winter, most spectacled eiders occur in ice leads south of St. Lawrence Island, where risk of harvest would be extremely unlikely. In recent years, particularly in early winter, spectacled eiders have occurred much closer to St Lawrence Island (USFWS, unpublished data), increasing potential exposure to harvest, but no information on actual harvest during these time periods exists (USFWS 2020a p. 37).

Discussions with hunters on the ACP and observations of USFWS employees confirm that some spectacled eiders are taken through subsistence activities. Recent interviews with community members on St. Lawrence Island conducted by Alaska Department of Fish and Game indicated that sea ducks in general are not an important subsistence food for these communities and their meat is not as desirable as marine mammals, crab, or fish (ADFG, 2021, pers. comm.). Subsistence migratory bird harvest surveys are conducted annually through the Alaska Migratory Bird Co-Management Council and the State of Alaska. While the accuracy of harvest estimates is unknown and is likely affected by several biases, reports of spectacled eider harvest are generally consistent with the species' distribution and anecdotal observations. Using the available harvest data, combined with information on spectacled eider availability, direct observations, and information from local residents, USFWS estimates that hundreds of adult and/or juvenile spectacled eiders are harvested in Alaska each year (USFWS 2020a pp. 38–39).

The number of spectacled eider eggs harvested annually in Alaska is likely small because the closed season period may serve to reduce eggging and egg collectors tend to target other species (USFWS 2020a p. 43). Based on harvest reports, the USFWS estimates that tens of spectacled eider eggs may be collected annually in Alaska (USFWS 2020a p. 44).

Little is known about harvest levels in Russia. Of the few spectacled eiders banded at Chaun, one banded bird was shot near Uelen village in East Chukotka (D. Solovyeva, 2020, pers. comm.). Estimates from limited harvest surveys conducted in a subset of villages in Yakutia and Chukotka, Russia from 1999 – 2006 suggest that thousands of spectacled eiders may be harvested in the region annually (Syroechkovskiy and Klovov 2009 p. 27,32). Hunter surveys suggested that the number of spectacled eiders harvested is dependent on the village's location relative to eider migration paths (Syroechkovskiy and Klovov 2009 p. 29), and that a higher number of spectacled eiders are harvested in the lower Indigirka River Delta villages and in Lorino, near Mechiginskaya Bay in eastern Chukotka, than other areas surveyed (Syroechkovskiy and Klovov 2009 p. 24). Precise estimates and trends in harvest cannot be calculated from the available information. These surveys are subject to the same biases as, and methods are less consistent than, the Alaska harvest surveys; therefore, inference from these data is limited, but provides evidence that spectacled eiders are harvested in Russia. Moreover, local

knowledge of hunting regulations is poor, and enforcement is rare in the region (Syroechkovskiy and Klovov 2009 pp. 21, 31).

In summary, we roughly estimate that hundreds of spectacled eiders to be harvested or shot in Alaska annually, and perhaps thousands in Arctic Russia, but note empirical information upon which to base this estimate is lacking. Some regulatory pressure and outreach/education may lower harvest of spectacled eiders in the future, but to our knowledge, such efforts are confined to Alaska and have not occurred in Russia. Therefore, both harvest rates and the effectiveness of preventative measures to reduce harvest are uncertain.

3.1.3. Exposure to Contaminants (other than lead)

Contaminants such as trace elements (e.g., selenium [Se], mercury [Hg], cadmium [Cd], copper [Cu], arsenic [As]), hydrocarbons, and persistent organic pollutants are produced by industrial activities around the globe and distributed into the environment through atmospheric, marine and freshwater pathways (Lovvorn et al. 2013 pp. 256 – 258). Sources can also be local, such as small oil spills at harbors or release of wastewater effluent into marine areas, and natural, such as oil seeps (Franson 2015:169 and references therein). Essential trace elements also occur naturally in the environment.

Spectacled eiders can be exposed to contaminants through ingesting prey items that have contaminants in their tissues (Franson 2015 p. 170) or by coming in contact with the substance externally. Bivalves, the primary prey of spectacled eiders, can have high concentrations of both heavy metals and organic contaminants (Franson 2015 p. 171).

Trace Elements

Sea ducks seem to accumulate higher concentrations of trace elements than freshwater birds (Franson 2015 pp. 197 – 198). While spectacled eiders could be exposed to local sources of trace elements during the breeding season, particularly near villages, or near industrial development on the ACP, most exposure to trace elements probably occurs in marine areas used during the non-breeding season (Grand et al. 2002 pp. 1674 – 1675, Lovvorn et al. 2013 p. 250, Miller et al. 2016 p. 304).

Ca, Cu, and Se were elevated in spectacled eiders collected near St. Lawrence Island, Alaska (Trust et al. 2000 p. 112). Stout et al. (2002 p. 215) examined spectacled eiders collected at St. Lawrence Island, Utqiagvik, Russia and the Y-K Delta and also found that Ca, Cu, Pb and Se were higher than levels found in other waterfowl. Se was lower in spectacled eiders on the ACP than for eiders on the Y-K Delta, and also declined through the season (Wilson et al. 2004 p. 408). Consistent with previous studies, Miller et al. (2016 p. 302) found higher Se, Cd, and Cu in adults than juvenile spectacled eiders, and adult male spectacled eiders had exceptionally high Se, Cd, and Cu compared to adult females. Adult female spectacled eiders had higher levels of Se and Cu than females of other waterfowl species.

Toxicity data for trace elements are lacking for most sea duck species and the effects of specific concentrations/exposures have not been identified (Franson 2015 p. 170,205). Sea ducks may have higher tolerance to trace elements than freshwater species because of the ionic rich marine environment in which they spend the majority of their life cycle. When examined, no evidence has been found that trace elements occur at toxic levels for individual spectacled eiders. For spectacled eiders on the Y-K Delta breeding area, trace element (Se, Cd, As) concentrations were not related to egg viability or nest success (Grand et al. 2002 p. 1675). Trust et al. (2000 p. 110) found that spectacled eiders with high concentrations of trace elements had subtle biochemical changes, but were in apparently good health. It is possible that the biochemical changes could lead to poor body condition, possibly leading to lower reproductive capacity (Trust et al. 2000 p. 112), but that hypothesis has not been tested.

Persistent organic pollutants

Most persistent organic pollutants in Arctic environments are from atmospheric transport, and are typically biomagnified through the food chain (Franson 2015 pp. 209 – 210); therefore, it is possible that spectacled eiders could be exposed to persistent organic pollutants during any part of their annual cycle. Trust et al. (2000 p. 109) detected hexachlorobenzene (HCB) and dichlorodiphenyldichloroethylene (DDE) in one and two birds, respectively, of 20 spectacled eiders sampled from Alaska. Wang et al. (2005 pp. 762 – 764) detected polychlorinated biphenyls (PCBs) in spectacled eider eggs collected on the Y-K Delta in 1995, but these levels were thought to be low and sub-toxic. Stout et al. (2002 p. 215) found that organochlorines in sampled spectacled eiders were below levels believed to be toxic.

Hydrocarbons

No information is available about hydrocarbon exposure of spectacled eiders, but oil spills from shipping, well blowouts, pipeline leaks, etc., are a potential source of spectacled eider habitat contamination, both in the marine and terrestrial environment.

Overall, spectacled eiders inhabit areas with low probabilities of hydrocarbon exposure. Areas with fishing, industrial activity, and populated areas (and thus a higher probability of contamination by hydrocarbons) comprise a small portion of the total habitat used by spectacled eiders. We are unaware of any significant oil spills that have occurred in the Arctic marine environment from shipping or oil wells, although the potential for accidents exists. An accident with an oil tanker such as the Exxon Valdez is currently unlikely in spectacled eider habitat in Alaska, as oil is piped overland rather than shipped out of ACP production areas. But, nearshore pipelines in the Beaufort Sea, in spectacled eider habitat, could potentially be damaged by sea ice.

The risk of hydrocarbon spills is likely to increase in the future as vessel traffic through the Bering Strait and the Russian Northern Sea Routes increases (Azzara et al. 2015 p. 57). Historically, vessel traffic in the range of spectacled eiders is largely limited by sea ice during the winter and during spring migration (November through April). Future vessel traffic from shipping, tourism, and development is likely to increase in the Bering Strait (Azzara et al. 2015 p. 57) and along the Northern Sea Route along the Siberian coast (Yumashev et al. 2017 p. 149) particularly as the navigability window widens as Arctic sea ice decreases (Wang and Overland 2015 p. 58) and ships with ice breaking capacity are constructed. In fact, large LNG tankers are now traveling through the region during all months of the year, including

January (Smith 2021). The consequences of a major oil spill would be serious and potentially long-lasting given the remoteness of the area and harsh conditions resulting in difficulties conducting oil spill cleanup operations (Arctic Council 2009 p. 119,136,138).

In terrestrial habitat, oil spills have occurred from pipelines. For example, over 200,000 gallons of oil were released from a 0.25-inch hole in a 34-inch pipeline at Prudhoe Bay in 2006. Smaller spills from pipelines have occurred on the ACP since that time (e.g., in 2009, 13,500 gallons of crude spilled from a frozen pipe in the greater Prudhoe Bay area). This risk is likely to increase with future onshore development.

Nuclear Radiation

There is an increasing interest in nuclear power generation, particularly in northern Russia. Russia recently deployed a floating nuclear power plant at the port of Pevek, in the Chaun Delta in Chukotka, Russia (Boyd 2019 pp. 1–2). Release of radioactive material could occur, particularly when spent nuclear fuel and radioactive waste is shipped to storage facilities in mainland Russia. Rosatom, the Russian state nuclear corporation, has plans to construct more floating and land-based nuclear power plants for deployment in the Russian Arctic in the future (Sevunts 2020 pp. 1–4). We have no evidence that spectacled eiders have been impacted by nuclear radiation, but increasing development of nuclear power plants in spectacled eider habitat poses a future risk.

Conservation Measures

Some conservation measures are in place to monitor contaminants exposure and reduce risk. Opportunistic monitoring of contaminants in spectacled eiders occurs, but not in a systematic way (USFWS, unpublished data). Proposed new industrial development in spectacled eider habitat in Alaska is likely to undergo Section 7 consultation under the ESA, which should include measures to minimize contamination of habitat by hydrocarbons. Federal and state agencies coordinate to prepare for and respond to hydrocarbon spills. In addition, many POPs have been banned for use in developed countries (Franson 2015 p. 209).

Summary

While evidence that trace elements such as Cu and Se are elevated in spectacled eiders, there is little evidence of negative effect of this exposure. Spectacled eiders could be exposed to hydrocarbons in breeding areas in industrial developments and near communities on the ACP and Arctic Russia. Little industrial development occurs near spectacled eider breeding habitat on the Y-K Delta. Cumulative or synergistic effects of repeated exposure of individuals to hydrocarbons or heavy metals may be of concern, but at this time possible population-level effects of contaminants on spectacled eiders are unknown. With increased vessel traffic predicted for the Bering Strait and Chukchi Sea in the future (See Section 3.1.5.) as well as continued and possibly increased industrial activity around the globe, we expect contaminants to remain a possible factor affecting spectacled eiders in the future. Therefore, contaminants remain a possible stressor, with some increasing risk of negative effects from contaminants in the future.

3.1.4. Collisions with Onshore and Offshore Structures

Human-made structures in coastal areas, such as light poles, buildings, drill rigs, wind turbines, offshore oil facilities, guyed towers and poles, and overhead power lines pose collision risks for spectacled eiders. Collisions cause immediate mortality, injury leading to death, or temporary injury. Spectacled eiders on the breeding grounds and during migration are at most risk of collisions with these structures.

Annual variation in eider density and distribution, weather, lighting conditions, and structure configurations such as the presence or absence of guy wires, affect the level of collision risk (Longcore et al. 2008 pp. 486 – 489). Anderson and Murphy (1988 p. 37) monitored bird behavior and collisions with a 12.5 km (7.8 mi) power line in the Lisburne area (Prudhoe Bay oil fields) during 1986 and 1987 and found that strike rate was related to the height of flight (Anderson and Murphy 1988 p. 22). Day et al. (2003 pp. 14 – 15) estimated the mean flight altitude of 1.8 m (5.9 feet), and maximum flight altitude of 15 m (49.2), for eider species flying past St. Lawrence Island, Alaska, in the fall. Day et al. (2005 p. 81) estimated the mean altitude of eiders during the day at 5.9 m (19.4 feet), and night at 16.7 m (54.8 feet), near Northstar Island in the Beaufort Sea during fall migration. The tendency to fly low puts eiders at risk of striking even relatively low objects in their path. However, Day et al. (2003 p. 9) observed all eider movements over the ocean during fall migration in the Beaufort Sea, rather than land, suggesting that eider strikes on structures on land are less likely during migration and winter.

Relative to the Y-K Delta and Arctic Russia breeding populations, ACP-breeding spectacled eiders likely have a higher collision risk due to more extensive human development in the Prudhoe Bay oil fields, near Utqiagvik, and along the Beaufort Sea Coast, where several offshore oil facilities are operating or in construction (e.g., Nanashuk, Oooguruk, and North Star). While systematic surveys have not been conducted, from 1991 – 2019 spectacled eider collisions with power lines or structures near Utqiagvik and at oil and gas facilities at Prudhoe Bay have been documented although in low numbers (USFWS, unpublished data). USFWS consults with Federal agencies on most industrial and community development on the ACP and seeks to minimize collision risk through various measures such as limiting overhead power lines in new construction and marking existing power lines with diverters to improve their visibility.

It is difficult to measure the population-level effect of collisions, given the potential biases in observational studies estimating collision rates, including crippling and nocturnal biases (Murphy et al. 2016 pp. 314 – 315), and scavenger and observer biases (Ponce et al. 2010 pp. 606 – 607), and interacting factors such as weather and eider distribution. Little quantitative information is available for collisions in Alaska, and information on mortality rates associated with collisions throughout North America remains uncertain (review by Ronconi et al. 2015:36).

In summary, evidence suggests that some individual spectacled eiders are killed annually by collisions with structures. Development is projected to increase on the ACP and along the coast of Arctic Russia in the future (BOEM 2018, Rosneft 2020a), which will likely result in an increased risk of collisions for

spectacled eiders, but continued work to minimize the risk through Section 7 consultation and other means may mitigate some of that risk in Alaska.

3.1.5. Collisions with Vessels

Vessels used for shipping, fishing, research, and tourism transit through spectacled eider migration corridors, molting areas, and wintering areas in the Bering, Chukchi and Beaufort seas, posing a collision risk. Bright lights on fishing boats, particularly during storm or foggy conditions, can cause birds to become disoriented and land on or collide with vessels (Reed et al. 1985:380). Most documented collision incidents occurred at night with large, lighted fishing vessels. Published examples include a single event with 899 seabirds striking a vessel in the Southern Ocean (Black 2005 p. 67) and a similar event with crested auklets occurred in the North Pacific Ocean (Dick and Donaldson 1978 p. 235). Two spectacled eider collision events have been reported recently: (1) a flock of spectacled eiders struck a fishing vessel near St. Lawrence Island in October 2019 and 22 spectacled eiders were found dead on the deck (Anchorage Fish and Wildlife Conservation Office, pers. comm.); and, (2) 11 dead spectacled eiders were found after a collision with a research vessel in the Bering Strait area in October 2020.

Reported collisions inevitably, and perhaps significantly, underestimate the actual mortality due to ship collisions because: 1) collisions are often episodic, such that observations collected on a few vessels in a single year may not be representative of collisions in general; 2) monitoring for collisions is difficult and an unknown number of collisions may go undetected, even by trained bird observers; and, 3) low visibility often coincides with increased collisions, which may increase the number of undetected collisions.

ESA Section 7(a)(2) consultations related to fisheries and research may result in measures that minimize collision risk, such as reducing the use of bright lights on fishing vessels when unnecessary and reducing speeds in certain areas used by spectacled eiders; however, consultations on other types of vessel traffic such as tourism and shipping do not occur because there is no clear Federal nexus.

As noted above, vessel traffic in the range of spectacled eiders is currently limited by sea ice extent and is not generally a stressor between November and April. However, future shipping and vessel traffic is likely to increase in the Bering Strait (Azzara et al. 2015 p. 57) and along the Northern Sea Route along the Siberian coast (Yumashev et al. 2017 p. 149) particularly as the navigability window widens as Arctic sea ice decreases (Wang and Overland 2015 p. 58).

In summary, although spectacled eider collisions with vessels at sea are known to occur, the collective impacts of collisions on spectacled eiders are difficult to quantify. As shipping and vessel traffic in the region increase, so will the risk of collisions to spectacled eiders, unless preventative measures are implemented widely.

3.1.6. Habitat Loss and Disturbance

Human activity can affect spectacled eider habitat and disturb eiders of all age and sex classes throughout the annual cycle.

Long-term Terrestrial Habitat Loss

Construction of infrastructure related to community and oil and gas development in coastal tundra areas, including activities such as gravel extraction and building roads, homes, pipelines, and other infrastructure, destroys nesting and brood-rearing habitat of spectacled eiders. The amount of potential habitat loss within each breeding area differs.

On the Y-K Delta, no oil and gas extraction or mining activities have occurred within the central coast zone, considered the primary breeding area of spectacled eiders. Only a handful of villages are located within this area, and little expansion of village footprints is occurring. Therefore, long-term habitat loss from these sources has been minimal.

On the ACP, extensive oil and gas development has occurred and extends from the western border of Arctic National Wildlife Refuge to inside the eastern border of the National Petroleum Reserve – Alaska (NPR-A). Oil development has been gradually spreading outward across the ACP from the original hub at Prudhoe Bay, which began producing oil in the 1970s. These developed areas are within spectacled eider breeding habitat on the ACP but most spectacled eider breeding pairs occur to the west of currently developed areas (Figure 10), and the actual footprint of buildings, permanent roads, and drilling pads is a small proportion of the overall spectacled eider distribution on the ACP. However, given industry's interest in the NPR-A as expressed by lease sales, seismic surveys, drilling of exploratory wells, and the development of the Alpine and Greater Moose's Tooth fields at the eastern end of NPR-A, expansion of industrial development is likely to continue along the ACP further into spectacled eider breeding habitat. Some community expansion at Utqiagvik has destroyed small amounts of spectacled eider nest and brood rearing habitat. Overall, direct, long-term habitat loss on the ACP is minimal, but is likely to increase in the future.

In Arctic Russia, information on industrial activity within the breeding range of Arctic Russia population is sparse; it appears that no onshore oil and gas development has yet to occur within the range of spectacled eiders. Onshore oil and gas development seems largely focused on areas west of spectacled eider distribution, but has been expanding eastward to the Taimyr Peninsula (Rosneft 2020a, 2020b). Future activities include developing the Northern Sea Route for shipping of oil and gas resources to Asia by possibly developing on-shore support facilities, as well as expanding exploration to offshore areas in the East Siberian/Chukchi Sea (Rosneft 2020a). The degree of community development in the Arctic Russia breeding area is unknown, but village populations have been decreasing in recent years (Kumo 2017 p. 7).

In summary, we have no evidence to suggest that foraging, nesting, or brood-rearing habitat loss is currently limiting spectacled eiders on the ACP, Arctic Russia, or Y-K Delta. Community and oil and gas

development have impacted a small proportion of spectacled eider terrestrial habitat to date, but they are expanding and may be of concern in the future.

Disturbance from Terrestrial Development

Human activities that may disturb breeding spectacled eiders include the use of existing oil and gas facilities and roads near nesting areas, and on-tundra activities such as scientific research, subsistence activities, tourism, recreation, reindeer herding, and activities related to oil and gas development that include aircraft landings and walking on the tundra.

Human disturbance may affect individuals in several ways. First, spectacled eiders may be forced to move away from preferred foraging or nesting habitats, reducing their ability to meet nutritional requirements or successfully reproduce. Second, spectacled eiders may be flushed off nests during incubation, exposing eggs or themselves to higher predation risk or nest abandonment (Afton and Paulus 1992 pp. 75 – 76, Swennen et al. 1993 p. 51, Mallory 2015 p. 348). Grand and Flint (1997 p. 931) reported 14 percent lower nest success for spectacled eider nests visited by researchers than those unvisited. Bowman and Stehn (2003 p. 13) estimated that the likelihood of mortality for spectacled eider nests increased by 4 percent after a researcher visit. Similarly, Meixell and Flint (2017 p. 9) estimated that observer visits to greater white-fronted goose nests were responsible for a 7-35 percent reduction in nest survival. Likelihood of nest abandonment or depredation presumably varies with the number and frequency of disturbance events. Third, when disturbed during the brood-rearing period, females and broods may flush and scatter, reducing the ability of the female to provide warmth for young ducklings that do not yet have the ability to thermoregulate (Afton and Paulus 1992 pp. 83, 88 – 89) and protect ducklings from predators. Fourth, it is possible that disturbance could reduce an adult's probability of survival, particularly if disturbance is frequent and in conjunction with other physiological stressors such as severe weather, low food abundance, high numbers of predators, disease, or contaminant exposure. However, we have no data indicating a significant effect of disturbance on adult survival.

The degree to which spectacled eiders successfully nest in disturbed areas or move to other less disturbed areas is unknown. Disturbance from these types of human activities is usually a one-time, isolated, and short-lived event, and not every disturbance event will result in nest or duckling depredation or nest abandonment. Repeated disturbance is of greater concern, and the frequency of disturbance is likely higher closer to communities, research camps, and industrial development; however, an unknown level of habituation to human activity may occur in some individuals.

Some conservation measures may lessen disturbance. Many development infrastructure and research projects in Alaska have a Federal nexus and therefore require Section 7 consultation under the ESA, which considers the environmental baseline and effects to listed species, and minimizes impacts through conservation measures. Subsistence harvest regulations prohibit egg gathering and hunting during the 30-day peak nesting period (USFWS 2020a p. 5), which may reduce the level of disturbance during that time period. Activities of residents and tourists are not regulated, but outreach to improve awareness of disturbance on nesting eiders is conducted annually by the USFWS in ACP villages.

The amount of human disturbance to spectacled eiders differs among breeding areas. The ACP likely has the most human activity; in particular, there is a significant effort in the summer to conduct surveys related to oil and gas extraction that includes thousands of aircraft landings and on-the-ground surveys in NPR-A (USFWS 2018 pp. 2 – 3) and in the Prudhoe Bay area, as well as recreation, tourism, research and subsistence activities near villages, especially Utqiagvik. On the Y-K Delta, research activity has declined, but subsistence activities such as hunting, egging, berry picking, and establishing fish camps occur. Little is known about human disturbance in the Arctic Russia breeding area, but there are indications that reindeer herding, and other subsistence activities, can cause disturbance in some areas.

In summary, while individual spectacled eiders are affected by human activity every year, given their wide breeding distribution and the relatively limited industrial and community footprints within breeding habitat, the likelihood that this factor is currently affecting the population measurably is low. However, as infrastructure expands the effect of disturbance may increase.

Disturbance in Marine Habitat

Oil wells have been drilled and are producing from four offshore gravel islands that have been constructed in the Beaufort Sea. Human activity on and surrounding these areas may disturb migrating or staging spectacled eiders, but the area affected is small relative to spectacled eider distribution in the area, and eiders can move away from such disturbance.

Vessels used for shipping, fishing, research, and tourism transit through spectacled eider migration corridors, molting areas, and wintering areas in the Bering, Chukchi and Beaufort seas, may cause disturbance. Vessel traffic is of most concern when vessels transit through spectacled eider molting areas between August and November when molting birds are present, as they have limited ability to move away and may be under higher energetic demands. A small amount of vessel traffic is subject to ESA Section 7(a)(2) consultation with USFWS, resulting in minimization measures such as reduced vessel speed, reduce lighting, or avoiding molting areas such Ledyard Bay or Norton Sound during the molting period.

Overall, the majority of spectacled eider marine habitat has no or little human activity, but as vessel traffic and industrial development increases in the Bering, Chukchi, Beaufort and East Siberian seas, disturbance may become more of a concern.

3.1.7. Disease, Parasites, and Biotoxins

Spectacled eiders may be affected by naturally-occurring diseases, parasites, and toxins through a direct effect on individuals or through impacts to food quality (e.g., harmful algal blooms).

Spectacled eiders could be exposed to a variety of viruses; for example, reoviruses (T. Hollmén, unpublished data, cited in Hollmen and Franson 2015:101) and avian influenza (AI; Hollmen and Franson 2015:103). Antibodies to AI viruses were detected in serum samples of 80 – 95 percent of eiders sampled (common, spectacled and Steller's eiders combined; Wilson et al. 2013:3). Low pathogenic AI

viruses likely circulate naturally in sea duck populations, and no highly pathogenic viruses have been isolated from spectacled eider samples (Hollmen and Franson 2015 pp. 102 – 104). Hall et al. (2020:4,6) detected low AI virus prevalence in sea ducks sampled in the Atlantic Ocean, and their data suggested that ocean water temperature is important factor affecting virus prevalence.

In addition to viruses, spectacled eiders could be exposed to parasites, bacteria, fungi, and algal toxins, but likelihood of exposure varies by cohort, distribution, and other factors. Occasional deaths and health effects from algal toxins present in bivalves have been documented in sea ducks, although no known cases have been found in spectacled eiders despite bivalves being their primary prey (Hollmen and Franson 2015 pp. 113 – 114). Blood collected from spectacled eiders on the ACP and Y-K Delta breeding areas was tested for presence of hematozoa, and *Leucocytozoon* sp. were detected in 6.5 percent of adults and 41.9 percent of juveniles (Reed et al. 2018 p. 2). No *Haemoproteus* or *Plasmodium* species were detected. This may be a typical infection rate, but little is known about its ecological implications.

Effects of various disease and parasites to individual spectacled eiders has not been evaluated. Exposure may result in a one-time, temporary effect to individuals, or the effect may be chronic (e.g., parasites), affecting future reproductive potential and survival. Studies on population-level effects of disease and parasites on sea ducks are lacking (Hollmen and Franson 2015 p. 98). The entire population could be exposed to various diseases, parasites and toxins; however, we don't know how many individuals are actually affected annually. Contagious disease is particularly of concern during years of high sea ice concentration when wintering spectacled eiders occur in large, tight flocks of 10's or 100's of thousands of birds. Given their remote and marine distribution, however, it is unlikely that we will detect population-level mortality events.

In the future, warming temperatures may also increase exposure to novel diseases, parasites or biotoxins through range shifts in disease organisms or their vectors (Van Hemert et al. 2014 p. 550). Avian influenza and other viral diseases may spread through climate change-induced alterations in migratory routes, stopovers, or ecological communities. Some shifts may already be underway; avian malaria and avian cholera were recently documented for the first time in Alaska (Loiseau et al. 2012 pp. 1 – 6, Bodenstein et al. 2015 p. 936). However, if spectacled eider winter distribution becomes wider and sparser with decreasing sea ice extent (as has been observed in 2019 and 2020; Section 3.1.9), then the risk of transmission of contagious disease may lessen relative to years when spectacled eiders are highly concentrated in small sea ice leads.

Harmful algal blooms have been detected in the Bering Sea, Chukchi Sea, and other Alaskan waters (Lefebvre et al. 2016 pp. 16 – 17, Natsuike et al. 2017a p. 8). The frequency of harmful algal blooms is predicted to increase in the future due to warming ocean temperatures (Natsuike et al. 2017b p. 7), high densities of *Alexandrium* spp. cysts reported in sediment samples from the Chukchi Sea (Natsuike et al. 2013 p. 56), and the potential for bivalves to bioaccumulate the biotoxin saxitoxin during blooms.

In summary, we have no evidence that disease, parasites, or toxins are currently affecting spectacled eider populations. However, we have very little data on this factor; therefore, we can make no

definitive conclusions. We expect exposure to diseases, bio-toxins, and parasites may increase in the future with a changing climate.

3.1.8. Predation

Predation is a major driver of spectacled eider nest failure and duckling mortality, and annual variation in nest success is likely determined by fluctuations in local predator populations or broad-scale predator dynamics (e.g., Arctic and red fox and microtine cycles; Grand and Flint 1997:931, Flint et al. 2016:17). Natural egg and duckling predators include pomarine, parasitic, and long-tailed jaegers; snowy owls; common ravens; glaucous, Vega, and mew gulls; and Arctic and red fox. Nesting females may be killed by snowy owls, Arctic fox, red fox, and mink (Petersen et al. 2000 p. 12). Wintering birds may occasionally be taken by walrus (*Odobendus rosmarus*; Lovvorn et al. 2010:55). Given that predation is a natural part of the system in which spectacled eiders have evolved, our primary concerns are whether nest predation has increased over time due to anthropogenic factors or to changes to the ecosystem (e.g., changes in sympatrically-nesting goose populations), and whether nest predation is expected to c in the future due to these factors.

Anthropogenic influences on nest predator populations

Human development within the range of spectacled eiders may artificially increase the availability of food and nest/denning sites for gulls, ravens, and foxes, thereby driving population increases and indirectly increasing nest predation. Buildings have provided nest sites for ravens, allowing them to expand their range to parts of the ACP and Y-K Delta that were not inhabited prior to human development (Day 1998:18 – 20, Powell and Backensto 2009:12; USFWS, unpublished observations). Increased food sources from landfills and marine mammal carcasses may support higher gull, raven and fox populations than were historically present (Day 1998 pp. 6 – 11, Powell and Backensto 2009 pp. 16 – 17). Reduced fox harvest, anthropogenic food sources in rural communities, and an increase in availability of denning sites at human-built structures may have resulted in increased numbers of Arctic foxes in developed areas of the ACP (Day 1998 pp. 26 – 28).

Aerial surveys primarily designed to enumerate waterfowl populations in Alaska also collect data on raven, gull and jaeger populations. On the ACP, the mean annual growth rate of glaucous gulls increased from 1992 – 2017 (1.034, 95 percent CI = 1.021 – 1.048; Wilson et al. 2018:8). The observed raven and jaeger populations remained stable over the same time period (mean annual growth rates of 1.011, 95 percent CI: 0.969 – 1.055 and 1.000, 95 percent CI = 0.990 – 1.011, respectively; Wilson et al. 2018:8). While this large scale aerial survey does not suggest changes in the raven population on the ACP as a whole, local observations indicate that the number of nesting ravens has increased at Utqiagvik and Prudhoe Bay (USFWS, unpublished observations).

On the Y-K Delta, aerial survey observations from 1992 to 2016 suggest that the glaucous gull population was stable, the mew gull population increased, and the jaeger spp. population declined (mean annual growth rate of the population index was 0.992, 90 percent CI = 0.976 – 1.008 for glaucous gulls; 1.025,

90 percent CI = 1.010 – 1.040 for mew gulls; and 0.979, 90 percent CI = 0.967 – 0.991 for combined jaeger observations; Swaim 2017:8). Ravens are not documented on the annual aerial breeding pair survey for water birds on the Y-K Delta, but in recent years biologists have observed ravens in parts of the spectacled eider breeding area where they had not been observed in the past (USFWS, unpublished observations).

No systematic surveys of nest predators are conducted in Arctic Russia so it is unknown if predator populations are changing over time. However, observations at Chaun suggest that the composition of avian predator populations may be shifting, as they observed increases in nests of large gull species and decrease in small gull and tern species over the past three decades (Solovyeva and Zelenskaya 2016 p. 848). A concurrent decline in nest success and nest density of spectacled eiders at their study site led the authors to hypothesize that these changes have resulted in higher spectacled eider nest predation rates (Solovyeva et al. 2018 pp. 748–749); however, this hypothesis has not been tested.

Some local efforts have been taken to reduce predator populations that have increased due to human subsidies. For example, Utqiagvik uses a trash incinerator to reduce trash on the landscape, although the incinerator is not always operational and thus not all trash is incinerated (USFWS, unpublished observations). Similarly, the oil and gas industry generally follows guidelines for proper waste management and disposal and facility design that minimize nesting and denning of nest predators (e.g., (USFWS 2020b p. 24,26). However, there is no information regarding the effectiveness of these conservation measures.

In summary, an increase in gull populations on the ACP and possibly in Arctic Russia, and local increases of ravens near communities, may impact spectacled eider productivity. Changes to fox populations due to anthropogenic factors are suspected but have not been quantified.

Increases in nesting goose populations

The decline of spectacled eiders between the 1960s and 1990s coincided with dramatic declines in cackling, greater white-fronted, and emperor goose populations on the Y-K Delta (Fischer et al. 2018 p. 149). Since the 1980's, nesting populations of geese have been increasing on both the Y-K Delta and ACP spectacled eider breeding areas. Aerial surveys on the central coast of the Y-K Delta indicate significant increases in breeding populations of cackling geese, greater white-fronted geese, and emperor geese from 1985-2017 (Swaim and Wilson 2017 pp. 3 – 4).

On the ACP, indices derived from aerial surveys from 1986-2017 show significant positive trends for greater white-fronted geese, brant, and snow geese (Wilson et al. 2018 p. 14). Targeted surveys in other parts of the ACP also show increases in goose abundance. For example, Burgess et al. (2017 p. 14,16) documented increasing nest counts of snow geese near Ikpikpuk River Delta colony along the Beaufort Sea coast, from approximately 50 nests in the 1990's to over 12,000 in 2015, and the colony has expanded in area. Also, the greater white-fronted goose molting population in the Teshekpuk Lake Special Area increased geometrically from 1976 – 2005 (Flint et al. 2008 p. 551). However, colonial-nesting geese in numbers like those seen at Ikpikpuk River Delta or areas of Canada such as Karrak Lake do not currently exist across large areas of the ACP spectacled eider breeding area. Goose nesting

densities are lower on the ACP than they are on the Y-K Delta breeding area. We have no data on nesting goose densities in spectacled eider breeding areas in Arctic Russia.

Goose population declines on the Y-K Delta were attributed to over-harvest (King and Derksen 1986 p. 472), but causes of the spectacled eider decline are unknown. While spectacled eiders and geese could have been affected by the same environmental or anthropogenic factors leading to their decline and subsequent increase, it is also possible that goose densities affect spectacled eider reproductive success and survival rates. Hypotheses related to the influence of sympatrically-nesting geese on spectacled eider populations include: 1) spectacled eider nest success may be higher in areas with high goose nesting densities due to predator swamping or protection by geese; 2) higher goose densities results in high nutrient deposition into ponds from feces, resulting in higher productivity, invertebrate abundance and thus food availability for adults and ducklings; 3) high-density colonial geese can destroy habitat; and/or, 3) inadvertent or intentional harvest of eiders may be less likely when geese are available for harvest.

Increased numbers of nesting geese may influence nest and brood survival probability through altered predator-prey dynamics. In a review, Flemming et al. (2016 pp. 397–398) suggest that species can benefit from predator satiation near goose colonies; however, these positive effects may be negated by aggressive interactions with geese, and the greater abundance of generalist predators such as gulls and Arctic foxes could elevate the risk of nest predation (Iles et al. 2013 p. 689, Flemming et al. 2019 pp. 10 – 11). Baldwin et al. (2011 pp. 409 – 410) found that cackling geese had higher nest survival probability inside a Ross's goose (*Chen rossii*) colony than outside of it, which supported the predator swamping hypothesis, and Robertson (1995 p. 112) also found that common eider nests on islands with snow goose nests had higher hatching success than those without nearby snow goose nests (p. 112). However, Samelius et al. (2011 pp. 1480 – 1481) demonstrated that geese and their eggs subsidize fox diets such that regional fox populations are larger than could be supported by small mammals alone, possibly increasing nest predation risk. These complex predator-prey community dynamics have not been specifically investigated in regards to spectacled eiders.

Beyond possible impacts to nest predation, increasing goose populations may affect surrounding wetlands, nest habitat, and harvest rates of spectacled eiders. Alisauskas and Kellett (2014 p. 137) hypothesized that nutrient inputs from large goose colonies increase eutrophication of water bodies, resulting in a hyper-abundance of invertebrate food for nesting king eiders and their ducklings; however, this hypothesis has also not been tested.

It is also possible that high densities of nesting geese may alter nesting habitat or compete with spectacled eiders for nest sites, at least in localized areas. In the future, if colonial-nesting snow geese increase dramatically on the ACP, preferred spectacled eider nest habitat may be altered. In the extreme case of colonial-nesting snow and Ross' geese, grazing altered plant communities significantly (Ganter et al. 1996 p. 967, Abraham et al. 2006 pp. 272 – 273, Alisauskas et al. 2006 pp. 203 – 204). Heavy grazing can remove significant amounts of vegetation, and waterfowl generally prefer nest sites with vegetative concealment. These drastic differences in above ground plant biomass from the

presence of colonial-nesting geese have also been linked to a negative relationship between small mammal abundance and goose numbers (Samelius and Alisauskas 2009 p. 97).

At the time of listing, harvest was hypothesized as a potential cause of the spectacled eider decline on the Y-K Delta, along with other threats (USFWS 1996 p. 7). Spectacled eiders have not been an important subsistence food resource, as harvest has been primarily focused on geese. It is possible, however, that when goose populations declined, hunters may have switched to eiders opportunistically, and then switched back to geese when their populations increased to the point that they became more available. However, we have no information to support or refute this hypothesis.

In summary, nesting goose populations have been increasing on both the Y-K Delta and ACP breeding areas. The density of sympatrically-nesting goose populations may affect spectacled eider populations, but research is needed to understand this relationship prior to making conclusions.

Predation summary

In the future, as the number of structures and anthropogenic attractants associated with industrial development and human habitation on the ACP increases, reproductive success of spectacled eiders may decrease. In contrast, increasing nesting goose populations within spectacled eider breeding areas may result in predator swamping and some nest protection. Additionally, predator populations may change due to changes in habitat and ecological communities due to climate change (Section 3.1.10.).

In summary, spectacled eiders in the vicinity of communities and industrial areas have likely been impacted by increased predator populations, although the severity of impacts are difficult to quantify. Future changes to predator populations may impact spectacled eider reproductive rates, but the cumulative direction and magnitude of such effects are unknown.

3.1.9. Climate Change

Greenhouse gas emissions increased at an unprecedented rate during the 20th century, resulting in global climate change characterized by warming atmospheric and ocean temperatures, diminishing coverage of snow and ice, and rising sea levels (Intergovernmental Panel on Climate Change (IPCC) 2014: 44). Global climate change is widely accepted as one of the most significant risks to global biodiversity. Here we describe the available data regarding abiotic and biotic changes that have occurred within spectacled eider habitat and are predicted to occur in the future, and then discuss possible implications for spectacled eiders.

Climate change projections used in this SSA report are based on Representative Concentration Pathway (RCP) scenarios (IPCC 2014 p. 57), a set of scenarios used for generating projections of climate change. There are four RCPs, selected to be representative of the range of theoretically possible atmospheric conditions which could exist at 2100, consistent with a large body of scientific literature at the time of their development. We focus here on RCP4.5 and RCP8.5 projections, the “medium-low” and “highest”

scenarios, respectively, in the RCP set. The RCP4.5 and RCP8.5 scenarios are widely used together in the scientific community, and were selected as the basis of projections for assessing climate change impacts in the Fourth National Climate Assessment (USGCRP 2018 p. 6). At 2100, the atmospheric conditions under RCP4.5 are associated with a projected global average temperature that is 2.5 °C (likely range 1.7 to 3.3 °C) higher than the average temperature between 1850 – 1900, and the conditions under RCP8.5 are associated with a projected global average temperature 4.3 °C (likely range 3.2 to 5.4 °C) higher than 1850 – 1900 (IPCC 2019 p. SPM-8). Using a range of climate change projections is one way to acknowledge uncertainty inherent in modeling and uncertainty about future human actions which influence changes in climate. RCP4.5 and RCP8.5 projections are best viewed as providing a technically plausible range, and that reality is somewhere within this range, rather than at either end.

Variable Conditions and Regime Shifts in Bering Sea

Spectacled eiders spend a large amount of their annual cycle in the northern Bering Sea where the ecosystem and climactic conditions vary greatly among years. At the start of the 21st century, the climate of the Bering Sea shifted from a historical pattern of alternating warm and cold years to a pattern dominated by an extended cold period (2006 – 2013) with briefer warm periods before and after (2002 – 2005, 2014 – 2017; Stabeno et al. 2012:35, 2017:5).

At a longer time scale, conditions in the North Pacific Ocean over the past century have been characterized by regime shifts influenced by long-term cycles in oceanic conditions, such as the Pacific Decadal Oscillation. These regime shifts are defined by rapid changes in ecosystem structure, detected through changes in biological variables such as altered primary productivity, invertebrate populations, and fisheries that then persist at a decadal time scale (Overland et al. 2008 pp. 92 – 93). While long-term climate change makes it likely that ocean temperature will continue to increase over time, water temperature may still fluctuate in the Northern Pacific and Bering Sea rather than reflect a linear relationship with time (Overland et al. 2008 pp. 98 – 99).

Population abundance indices of eiders (king, common, spectacled and Steller's eiders pooled) were correlated with documented regime shifts in the North Pacific in 1977 and 1989, but not the regime shift that occurred in 1998 (Flint 2013 p. 61). Decreases in survival and recruitment have been reported for other avian species in the Bering Sea related to the 1977 and 1989 regime shifts (Irons et al. 2008 p. 1460). However, it is unclear how global climate change may affect long-term climactic cycles in the North Pacific and Bering Sea, and we do not have adequate information to characterize the effects of shifts in marine regimes on spectacled eiders.

Physical changes to marine environment

Here we describe the physical changes predicted to occur in the marine environment, including increasing air temperature, increasing sea surface temperature, decreasing sea ice extent and thickness, and ocean acidification. The potential effect of these changes to spectacled eiders is described in the “Biotic changes” section below.

The global average surface temperature increased 0.85 °C from 1880 – 2012 (IPCC 2014 p. 2). The period from 1983 – 2012 was likely the warmest 30 year period of the last 1400 years in the Northern

Hemisphere (IPCC 2014 p. 2). Temperature changes in the Arctic have been more dramatic. Arctic surface air temperature has likely increased by more than double the global average over the last two decades (IPCC 2019 pp. 3–3). Arctic temperatures from 2014 – 2019 have exceeded all records since 1900 (Overland et al. 2019 p. 2). These increasing temperatures drive changes in Arctic Ocean characteristics, such as increases in sea surface temperature and decreases in sea ice.

In the future, the global mean surface temperature change predicted for 2016 – 2035 relative to 1986 – 2005 is similar for all RCPs, and likely to increase by 0.3°C to 0.7°C (IPCC 2014 pp. 58–60). By the mid-21st century, the magnitude of projected warming is affected by choice of emissions scenarios; for example for the period 2046-2065, the likely range of surface temperature increase is 0.9 – 2.0°C for RCP4.5, and 1.4 – 2.6°C for RCP8.5. The Arctic region will continue to warm more rapidly than the global mean (IPCC 2014 pp. 60–61).

Increasing sea surface temperatures in the Arctic are primarily driven by solar radiation absorbed by the sea surface when not covered in ice. It is also influenced by advection of warm water from more southerly locations; for example, the Chukchi Sea receives warm water from the North Pacific Ocean (Timmermans and Ladd 2019 p. 1). Over large sectors of the seasonally ice-free Arctic, summer sea surface temperatures increased at around 0.5°C per decade from 1982 – 2017 (IPCC 2019 pp. 3–3). The sea surface temperature in the Chukchi and Beaufort seas in August 2019 were approximately 1-7°C warmer than the 1982 – 2010 August mean (Timmermans and Ladd 2019 p. 2). Trend analyses of the Chukchi Sea has shown warming over the past 96 years, and the rate of warming has increased in recent decades (Danielson et al. 2020 p. 11); the August 2019 mean sea surface temperatures in the Chukchi Sea region were the second highest on record (Timmermans and Ladd 2019 p. 3).

Based on multiple data sets and modeling efforts, Arctic sea ice, including that in the Chukchi, Beaufort, and Bering seas, has been declining in extent and thickness in recent decades. It is very likely that Arctic sea ice extent (the total area of the Arctic with at least 15 percent sea ice coverage) has declined since 1979 in each month of the year through 2018 (IPCC 2019 pp. 3–12). It is virtually certain that Arctic sea ice has thinned (IPCC 2019 pp. 3–14). There is also high confidence that the Arctic sea ice melt season has extended by 3 days per decade since 1979 due to earlier melt onset, and 7 days per decade due to later freeze-up. This longer melt season is consistent with the observed loss of sea ice extent and thickness (IPCC 2019 pp. 3–15).

Changes in sea ice are particularly apparent in the Bering Sea. Four of the five seasons between 2014 – 2015 and 2018 – 2019 had a mean sea ice extent lower than any season prior to 2014 – 2015 (Siddon and Zador 2019 p. 44, Stabenro and Bell 2019 p. 8954). The Bering Sea ice extent for the winter of 2020-2021 was below the 1981 – 2010 median (Figure 8).

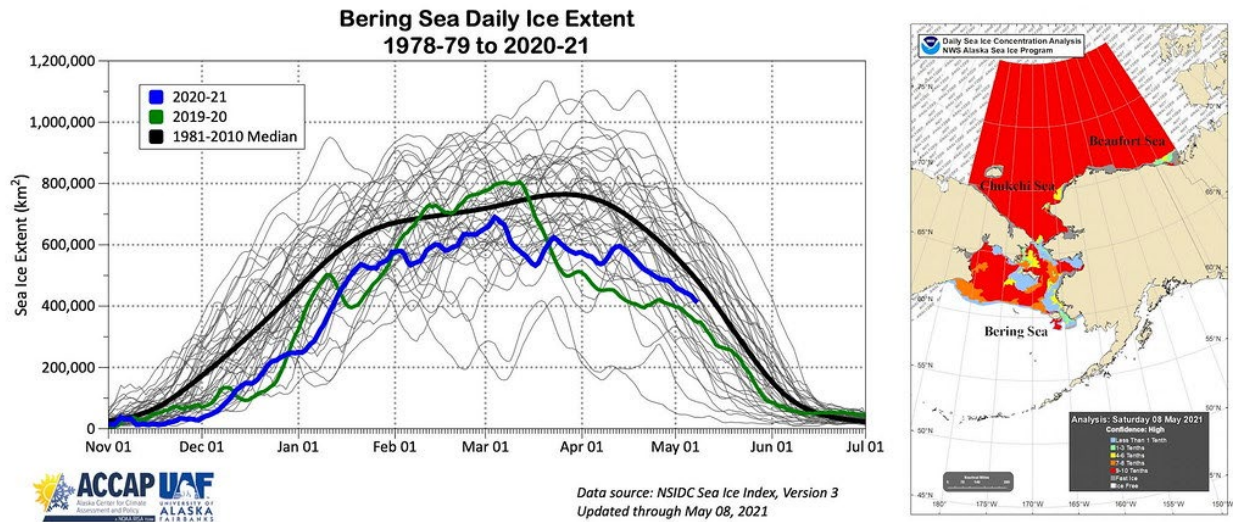


Figure 8. Daily sea ice extent in the Bering Sea. Graph from Alaska Center for Climate Assessment and Policy. Accessed May 10, 2021: <https://uaf-accap.org/sea-ice/>. Data from NSIDC Sea Ice Index, Version 3.

Until recently, a persistent feature of the Bering Sea has been a bottom pool of cold water at less than 2°C that extended south of the Bering Strait after sea ice retreats north; in some years it extends as far as the Alaska Peninsula, and in warmer years to St. Matthew Island. This feature affects the distribution of pelagic fish, as some avoid water temperatures less than 1°C (Stevenson and Lauth 2019 p. 408). Due to low sea ice extent and thickness over the Bering Sea shelf in winters 2017/2018 and 2018/2019, the cold pool was non-existent in 2018 (Stabeno and Bell 2019 p. 8953), and still reduced from its normal distribution in 2019 such that it only occurred over the northwest portion of the Bering Sea shelf (Siddon and Zador 2019 p. 50).

It is very likely that projected Arctic warming will result in a continued loss of sea ice in the future. Predictions span a large range in the timing of when the Arctic may become ice-free in the summer, and for how long, due to climate variability, scenario uncertainty, and model uncertainties (IPCC 2019 pp. 3–25). For stabilized global warming of 1.5°C, there is a projected 1 percent chance of a given September being sea ice free in the Arctic at the end of the 21st century; for stabilized global warming of 2°C, the chance rises to 10 – 35 percent (IPCC 2019 pp. 3–5).

About a quarter of carbon dioxide (CO₂) released by human activities is taken up by the oceans (IPCC 2019 pp. 3–20). Carbon dioxide dissolves in surface water to form carbonic acid, which, upon dissociation, causes a decrease in pH (acidification) and carbonate ion concentration. This can affect organisms that form shells and skeletons using calcium carbonate, such as benthic invertebrates preyed upon by spectacled eiders. It is virtually certain that global ocean pH is declining, and it is lowering the chemical stability of mineral forms of calcium carbonate (IPCC 2019 pp. 5–29). Models project global surface-ocean declines in pH under both RCP4.6 and RCP8.5 scenarios (IPCC 2019 pp. 5–29, 5–30). The changes in pH will be greatest in the Arctic Ocean and high latitudes of the Pacific Ocean because of

their lower buffer capacity and upwelling systems than more southerly oceans (IPCC 2019 pp. 5–30). Ocean acidification has been documented in the North Pacific (Byrne et al. 2010 p. 3), and is predicted to continue to increase in the future (Feely et al. 2009 p. 45, IPCC 2014 p. 62).

Ocean acidification is predicted to reduce calcification which has deleterious effects on growth rates and development in bivalves (Gazeau et al. 2007 p. 3, Fabry et al. 2008 pp. 419 – 420, IPCC 2014 p. 67). Although the degree to which ocean acidification affects marine invertebrates varies by species and other habitat factors (Kroeker et al. 2013 p. 1888, Goethel et al. 2017 p. 113,118), abundance of bivalves in the Bering, Chukchi, and Beaufort seas may decrease in the future as a result of decreasing pH levels.

Biotic Changes and Effects to Spectacled Eiders

Increasing sea surface temperature and reduction in sea ice extent affects timing and development of primary and secondary production cycles, which provide energy to the entire ocean food web. Ocean primary productivity in Arctic regions showed positive trends over the 2003 – 2019 period, but patterns varied spatially and temporally (Frey et al. 2019 p. 2). It is unclear how primary productivity in the Bering Sea is affected by sea ice loss – some studies suggest years with early sea ice retreat are more productive, whereas others provide evidence that primary productivity is reduced in years with less sea ice (Brown and Arrigo 2013 p. 54).

The distribution and composition of the benthic invertebrate community has shifted over the past 30 years in the northern Bering Sea and southern Chukchi Sea, possibly associated with changing physical conditions such as sea ice extent and variations in ocean currents, and the associated deposition of usable organic carbon to the benthos (Grebmeier et al. 2018 p. 14, Goethel et al. 2019 p. 135, Lovvorn et al. 2020). The benthic community in a polynya south of St. Lawrence Island, near the 1990s wintering area of spectacled eiders, has been periodically sampled since the mid 1980's. The bivalve *Macoma calcaria* dominated the area in the 1950's and 1960's and was then replaced by *Nuculana radiata* by late 1980's (NRC 1996 cited in Lovvorn et al. 2003:260). More recently, the dominant benthic taxa for the three most southerly stations have changed from mainly bivalves (*N. radiata* and *Ennucula tenuis*) to mainly polychaetes (*Axiiothella catenata* and *Maldane sarsi*), but the northern two sites remained dominated by *E. tenuis* bivalves, although in some years the bivalve *M. calcaria* has been dominant (Grebmeier et al. 2018 p. 8). Goethel et al. (2019 p. 132) demonstrated a northward shift of *M. calcaria* from 1998-2014, and a concurrent southward shift in the area north of St. Lawrence Island, with highest abundances of bivalves in areas with higher primary production. Overall, benthic biomass in the southern part of the 1990s wintering area has declined since 2005 relative to the stations further north. The core of the high benthic biomass hotspot was previously centered at 62°N, but it is now located further north around 63°N (Grebmeier et al. 2018 p. 7).

The significant reduction in benthic biomass in this area may represent a decline in prey availability for spectacled eiders (Grebmeier et al. 2018 p. 11). However, the spatial scale of benthic sampling conducted in this area may not detect small shifts in species composition or prey availability for spectacled eiders. In addition, if spectacled eiders forage on the most available bivalve prey and size of that prey, as has been suggested (Lovvorn et al. 2003, Richman and Lovvorn 2003), then species shifts in their wintering range may not significantly affect bioenergetics. However, if overall biomass is

decreasing, or the invertebrate community shifts from a bivalve-dominant to polychaetes or sand dollars, effective spectacled eider foraging may be compromised. It is important to note that many factors, including sea ice extent and weather, also affect prey availability, energetics, and resulting body condition.

Pelagic fish distribution in the Bering Sea is shifting. The disappearance of the cold pool has opened more northern habitats to pelagic fish species, such as walleye Pollock (*Gadus chalcogrammus*) and Pacific cod (*Gadus microcephalus*), which were distributed significantly further north on the Bering Sea Shelf in 2017 (relative to 2010; Stevenson and Lauth 2019:413), and in 2018 and 2019 (Richter-Menge et al. 2019 p. 85). More recently, die-offs of piscivorous/plankivorous seabird species in the Bering Sea and Gulf of Alaska have been attributed to decreased prey abundance and/or prey quality due to warming ocean temperature and direct competition with increasing numbers of warm-water fish species expanding northward into the Bering Strait region (USGS 2016; R. Kaler, USFWS pers. comm.) As benthivores, spectacled eiders have not been associated with these die-offs, and from 2017 to 2020, only three eider carcasses (one eider sp., and two king eiders) were reported.

A reduction or disappearance of sea ice during portions of the winter may affect spectacled eiders directly by requiring them to remain in open water rather than use a roosting platform of sea ice. In addition, the presence of sea ice dampens wave action, and without it, eiders may have to contend with extreme wave conditions. Both of these factors have been hypothesized to affect energetic requirements of spectacled eiders, possibly affecting body condition and resulting in cross-seasonal effects of lower breeding propensity or even survival (See Section 2.3.2; Lovvorn et al. 2009:1607, 2014:402 – 403). However, patterns of distribution of spectacled eiders marked with satellite transmitters in winters 2018 – 2019 and 2019 – 2020 indicate that some spectacled eiders may move in response to sea ice concentration and extent, rather than remain in the 1990s wintering area and contend with open water conditions (Figures 4 and 5; Section 2.2.2).

In summary, the Bering and Chukchi seas are undergoing physical changes due to climate change, including increasing water temperatures, ocean acidification, and loss of sea ice. Loss of sea ice could directly affect spectacled eider energetic requirements or result in earlier northward movements in the winter. Ecosystem changes, such as shifts in benthic invertebrate communities, have been observed, but the effect of such changes on spectacled eiders has not been measured.

Changes to terrestrial habitat

Terrestrial breeding habitat, tundra areas where spectacled eiders nest and raise broods, has likely been impacted by climate change, and is predicted to continue to change in the future. Some of the impacts from increasing air temperatures in the subarctic and Arctic include: erratic weather patterns, warmer springs with earlier break-up, permafrost degradation and erosion resulting in declines in pond area and abundance, increased pond temperatures, changing snow conditions, and storm surge flooding on the Y-K Delta that increases salinity in freshwater ponds. We discuss these and the possible implications for spectacled eiders below.

Fewer cold temperature extremes on seasonal time scales are likely (IPCC 2014 p. 60); thus, years with harsh springs and late snowmelt would be predicted to decrease over time. This could lead to a longer breeding season, and increase spectacled eider productivity by reducing the energy required for thermoregulation of adults, eggs, and ducklings.

There are also potential negative effects of warming temperatures. Permafrost temperatures in most regions of the northern hemisphere have increased since the early 1980s and the area of permafrost near the surface is projected to decrease by between 37 percent and 81 percent by the end of the 21st century (IPCC 2019 pp. 3–59). Increased temperatures in northern Alaska have caused ice wedge degradation, polygon drainage, and changes in vegetation (Liljedahl et al. 2016 pp. 313 – 314). Near Utqiagvik, Loughheed et al. (2011 pp. 593 – 595) observed higher mean and maximum pond water temperatures, nutrients, and primary productivity between the 1970s and 2007-2010. Additionally, paired photographs suggest that *Carex aquatilis* has increased in biomass since the 1970s (Loughheed et al. 2011 p. 597). Andresen and Loughheed (2015 pp. 5,7-8) observed a 30.3 percent net decrease in pond area and a 17.1 percent decrease in pond abundance in the Utqiagvik Triangle area from 1948 – 2010, and suggest that increased thaw depth, temperature, and vegetation cover play an important role in this change.

These wetland changes, separately or combined, may positively or negatively affect the quality of spectacled eider nest and brood-rearing habitat. Changes in pond temperature could change the availability of aquatic invertebrates, such as chironomids, which tend to emerge according to pond temperature (Hansson et al. 2014 p. 4). Depending on spectacled eiders' ability to initiate nests earlier in the season, hatch of invertebrates may or may not occur when it is most advantageous to females or their ducklings (i.e., phenological mismatch; Visser et al. 1998:1868 – 1869, Hansson et al. 2014:5 – 6). This could lead to decreases in spectacled eider reproductive success. Reduction in pond depth may affect duckling movement and predator avoidance, and their ability to feed; shallower ponds are more difficult to exit, but benthic food resources may be more accessible. Increases in emergent vegetation such as *Carex aquatilis* may improve escape duckling cover and foraging habitat.

Permafrost degradation could lead to continued decrease in freshwater pond area and abundance on the ACP and Arctic Russia, as was observed near Utqiagvik (Andresen and Loughheed 2015 p. 5), and increased coastal erosion may degrade shorelines and drain large lakes (Mars et al. 2007 p. 586, Jones et al. 2009 p. 2). There is strong evidence that permafrost loss caused by climate change is decreasing large lake area and abundance in areas with discontinuous permafrost in Siberia (Smith et al. 2005 p. 1429), and parts of subarctic Alaska (Riordan et al. 2006 p. 5). However, the impacts of warming permafrost and erosion on surface water dynamics in areas with continuous permafrost (such as the ACP and Arctic Russia) are dependent on many spatial and temporal variables, and thus difficult to predict (Smith et al. 2005, p. 1429). This could reduce the availability of tundra pond habitat for nesting and brood-rearing spectacled eiders, but the specific changes to habitat, how that affects nest habitat availability, and the timescale at which this may happen is unknown.

Increases in air temperature and decreases in sea ice have influenced snow conditions such as the duration of snow cover, snow depth, and snow quality in the Arctic. Long-term trends for snow cover

extent are negative across the Arctic: 3.4 percent (plus or minus 2.1 percent) per decade and 15.2 percent (plus or minus 6.4 percent) per decade for May and June, respectively (1981 – 2019). There has been a decrease in snow-water equivalent of more than 10 percent over the entire Arctic since 1981 (Richter-Menge et al. 2019 pp. 15 – 16).

Snow cover duration in the Arctic is projected to decrease by 5 percent to 25 percent by the end of the century, depending on the RCP used, due to later snow onset in autumn and earlier snow melt in spring (IPCC 2019 pp. 3–63). Total precipitation is predicted to increase in the high latitudes (IPCC 2014 p. 60), with a concurrent increase in precipitation coming as rain in winter rather than snow, and rain-on-snow events are expected to become more frequent (ACIA 2004 p. 22). These conditions are predicted to negatively influence microtine populations in the Arctic (Aars and Ims 2002 pp. 3451 – 3452, Kausrud et al. 2008 pp. 93 – 95, Gilg et al. 2009 p. 2642) with possible cascading effects to predator-prey dynamics and the entire tundra ecosystem.

Climate change is likely to impact habitat in the subarctic Y-K Delta breeding area more rapidly and severely than the northern ACP and Arctic Russia breeding areas, and in more diverse ways. Such changes are likely to occur at a higher rate than in northern Alaska because of the much warmer permafrost temperatures in the subarctic.

The Y-K Delta is characterized by low relief and large tide cycles that makes it particularly susceptible to storm-driven flood tides (Jorgenson and Ely 2001 p. 132). Most documented storms from 1898 – 2012 that caused extensive coastal flooding occurred in October and November (Terenzi et al. 2014 p. 363), but occasionally storm surges have occurred in the spring (most recently in 2018, USFWS unpublished observations). These storm surges impact vegetation (Terenzi et al. 2014 p. 371), and result in increased salinity in freshwater tundra ponds, accelerated permafrost melt, and/or increased rate of coastal erosion, although the complex processes of the coastal ecosystem make exact changes difficult to predict (Jorgenson and Ely 2001 p. 135). In the future, the frequency and severity of coastal storm surge floods on the Y-K Delta are predicted to increase due to several interacting factors (IPCC 2014 p. 67). The presence of sea ice dampens the effects of winds on storm surges. Fewer months of shore fast and sea ice in the Bering Sea, coupled with a rise in sea levels, would result in more frequent overbank flooding on the Y-K Delta, perhaps even occurring monthly and during the spring/summer nesting season (Terenzi et al. 2014 p. 372). This could be detrimental to nest success of spectacled eiders, particularly given that the nest in highest densities in coastal zones impacted by storm surges, although the risk would decline the further inland nests are found.

Storm surges can affect water quality via salt water inundation of wetlands, making them brackish (Terenzi et al. 2014 p. 371). Researchers have hypothesized that salinity of freshwater ponds on the Y-K Delta breeding area has increased with the increasing storm surge frequency and magnitude related to climate change. This salinity may affect duckling and juvenile survival (Flint et al. 2006 p. 909) and/or aquatic invertebrate abundance or availability (Section 3.1.10.). As described in Section 2.3.1., spectacled eider ducklings require sources of fresh water, particularly in the first week after hatch. Hollmén et al. (2020a p. 1) combined wetland salinity data collected on the Y-K Delta breeding area and spectacled eider nest distribution at 1 km resolution. Approximately 10 – 60 percent of the Y-K Delta

population inhabited an area with greater than 6 ppt salinity. However, salinity can vary at a small scale from pond to pond (less than one km resolution); therefore, low salinity wetlands may be available even in areas that fall in the high salinity category at low resolution (Hollmen et al. 2020a p. 1). In addition, at Kashunuk broods were observed traveling up to 8 km (5 mi) to less saline habitats in the first week after hatching (Flint et al. 2006:909).

Direct observations of spectacled eider hens and broods in 2011 and 2012 at Kigigak suggest that there is no clear relationship between pond salinity and brood habitat use. Hens and broods were not observed to disproportionately use the lowest salinity ponds (i.e., those in higher elevation). Although observations of 1 – 3 day old broods were limited, some broods used ponds with high salinity levels, including salinity levels that have been shown to impair common eider ducklings in captive environments (Graff 2020b p. 1). Also, ducklings did not concentrate or prefer specific ponds at any of the observed locations, suggesting consistent aquatic invertebrate availability across the sampling area at Kigigak. While sampling of wetlands at Kashunuk in 1997 – 1998 suggested that food abundance had a negative relationship with salinity (O’Connell 2001 p. 37), data collected at Kigigak were not consistent with these findings (Graff 2020b p. 1). At Kigigak, some zooplankton were only found in low salinity water but the most common invertebrates (benthic and planktonic) were found in all habitat types and salinity levels measured (salinity ranged from ~0 – 16 ppt; Graff 2020b p. 1).

Summary – Terrestrial Habitat

While there is high certainty in predictions of how climate change will affect some physical characteristics of the northern environment, such as air and ocean temperatures, there is limited resolution and inherent uncertainty in the links between physical factors, biological factors, and their subsequent impacts to spectacled eiders.

The individual responses of spectacled eiders to the documented terrestrial habitat have largely gone unmeasured. If these changes decrease habitat quantity or quality, body condition of adults and/or ducklings may be impacted, and breeding probability and reproductive success may be negatively affected, particularly if other factors are also acting on individuals (such as disease, contaminants, etc.). Given that effects to individuals and their responses are unknown, we have even more uncertainty about how this could translate to population-level response. However, it is likely that some of these environmental changes are occurring at a faster rate on the subarctic Y-K Delta than on the ACP and Arctic Russia; in particular, increased storm surge flooding and salinization of wetlands.

3.1.11. Summary of Current Factors Affecting Viability

Several factors may affect individual spectacled eiders, and some may rise to the level that they affect population resilience. For many factors, we have limited information with which to quantify their effects to spectacled eiders, and therefore determine if they are significant population-level threats. However, two factors, exposure to lead and climate change (specifically the effect of both extreme and low sea ice conditions) are of the most concern for spectacled eiders and have been shown to affect demographic rates (Grand et al. 1998 p. 1106, Flint et al. 2016 pp. 21–22, 24–25, Christie et al. 2018 p. 7). All of the factors identified in this analysis are expected to continue, at some level, in the future. Some, such as collisions, exposure to contaminants, and salinization of freshwater wetlands, are

expected to increase in severity in the future. With adequate management actions, harvest and lead exposure may decrease over time. Climate change has altered, and is expected to continue to alter, both the tundra breeding habitat and the marine non-breeding habitats of spectacled eiders. The northern Bering Sea in particular is expected to undergo drastic changes in the next several decades due to a reduction in sea ice. This may have consequences for spectacled eider food availability and thermoregulation in winter, resulting in reductions in breeding propensity and survival; however, whether spectacled eiders can behaviorally adapt and find adequate foraging conditions is unknown.

Influence Diagram: *Hypothesized* relationships between climate change factors and spectacled eider population resilience

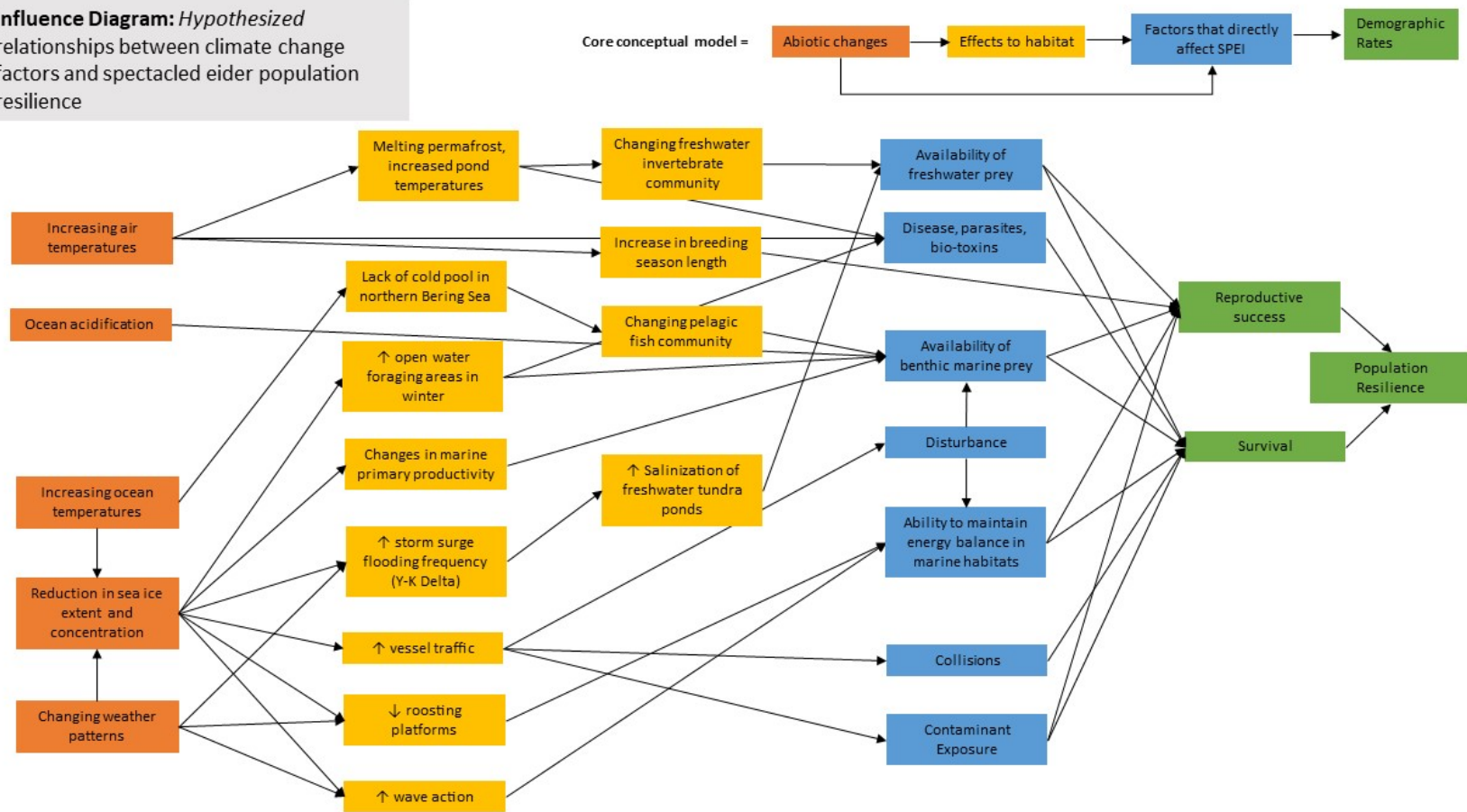


Table 2. Summary of factors that may affect spectacled eiders at the individual and/or population level.

Factor that may affect population resilience	Currently affecting spectacled eiders?	Predicted to affect spectacled eiders in the future?
Ingestion of lead ammunition	Yes	Yes, but could decrease with adequate outreach and communication, or change in hunting regulations/policies
Harvest and shooting	Yes	Yes, but could decrease with adequate outreach and communication, change in hunting regulations/policies, or increases in goose populations
Contaminants exposure (other than lead)	Unknown	Yes, possible increase in exposure probability from continuing anthropogenic input into the environment and increased oil spill risk from vessel traffic
Collisions with structures	Yes	Yes, increasing with more infrastructure/development
Collisions with marine vessels	Yes	Yes, increasing with more shipping, research and fisheries vessel traffic
Habitat loss and disturbance	Yes	Yes, increasing with more terrestrial development and vessel traffic
Disease, parasites, and bio-toxins	Unknown	Yes, possible increase with warming temperatures but decreased risk if wintering birds are less concentrated
Predation	Yes	Yes, possible increase with more infrastructure/development, and/or changes in ecosystem, or decrease with increasing goose populations
Climate change	Yes	Yes, extreme changes to spectacled eider habitat are highly likely; the effect on spectacled eiders, individually and at the population level, are unknown

3.2. Current Condition of Required Habitats

Here we summarize the current condition of habitat requirements of spectacled eiders identified in Section 2.3, given what is known about the factors influencing spectacled eiders and their habitat (Section 3.1).

Tundra habitat

Breeding habitat (i.e., wet coastal tundra within a network of lakes, ponds and sloughs, is not spatially limited in any of the three breeding areas. However, the quality of such habitat may be compromised, at least at local scales, from the following factors:

- Lead ammunition has contaminated breeding habitat, particularly on the Y-K Delta and Arctic Russia breeding areas;

- Industrial and community development in breeding areas has contributed to long-term habitat loss, on-going disturbance, and structures that pose collision risk in breeding areas, particularly in localized areas of the ACP;
- Nest predator populations may have increased from human subsidies, particularly near industrial sites and communities on the ACP; and,
- Wetland habitat on the Y-K Delta may be changing due to increasing number of storm surge floods and resulting salt water inundation of wetlands. However, data on salinity are limited, and the impacts to invertebrate availability for adult females and ducklings, and effects of salinity on wild ducklings, are unknown.

Habitat in molting areas and migration corridors

The physical characteristics of molting habitat and migratory routes (shallow coastal water, typically less than 36 m deep) are not limited. Prey availability may be changing over time in these areas; however, few data are available to evaluate the current condition of marine invertebrate communities. Benthic sampling of the Ledyard Bay molting area (also used during spring migration) suggests that bivalve abundance varies spatially and temporally and is affected by shifting sea ice in the spring (Lovvorn et al. 2015 p. 167,169). The core area used by molting spectacled eiders in Ledyard Bay largely consists of sandy substrate dominated by sand dollars (*Echinarcachinius parma*; Feder et al. 1994 pp. 178 – 179, Sexson et al. 2016 p. 472). Sand dollars are not thought to be eaten by spectacled eiders, leading to a hypothesis that spectacled eiders may not select molting areas for high prey availability (Sexson et al. 2016 p. 472). Bottom faunas in other molting areas, including Norton Sound, off Mechigmenkiy Bay and offshore between Indigirka and Kolyma deltas are not described. However, an empirical model based on benthic sampling from 2010-2014, when extrapolated to areas that lack benthic samples, indicates that benthic biomass in the northern Bering Sea and southern Chukchi Sea is highest in the area northwest of St. Lawrence Island along the eastern coast of Chukotka peninsula (Lin et al. 2019 p. 454), suggesting that food resources may be abundant in the Mechigmenkiy Bay area. Studies of the Chukchi and Bering sea ecosystems indicated that changes to the benthic community are occurring that may affect food availability and thus habitat suitability of existing migratory corridors and molting areas (Section 3.1.10).

Winter habitat

In years with extreme sea ice extent and concentration, openings in the sea ice from which spectacled eiders can forage are limited. Historically, the conditions in the wintering area in the Bering Sea varied annually, but the area is currently undergoing drastic changes related to climate change. Since listing, scientists have observed warming sea surface temperatures, changes in weather patterns, and more years with low sea ice extent in the Bering Sea. When sea ice is less extensive, spectacled eiders have access to more benthic foraging area, thereby opening more wintering habitat. However, there is some evidence that the existence of a moderate amount of sea ice is beneficial to eiders, as it lessens thermoregulation requirements (Section 2.3.2). In addition, observations include changes in species composition of benthic invertebrates used by spectacled eiders in the wintering area (Section 3.1.10), which could affect the availability of food for spectacled eiders during an energetically demanding part

of their life cycle; however, we do not know the population level effects of such changes, or whether spectacled eiders will respond by moving to equally or more productive areas. Vessel traffic is increasing in wintering areas as sea ice extent decreases, resulting in increasing risk of contaminant exposure, collisions, and disturbance.

In summary, spectacled eiders occupy remote habitats which are largely intact. However, some factors may be affecting spectacled eider habitat quality, and these effects may increase in the future (salinization of wetlands, changes to benthic prey communities), or may decrease (fewer years with extreme sea ice conditions). The magnitude of the effect of these factors on spectacled eider population resilience is uncertain; therefore, in this SSA we primarily focus on measures of population abundance and growth rate to evaluate resilience, as these should reflect the effects of influential factors and habitat changes on the population resilience.

3.3. Population Resilience

3.3.1. Methods

Here we describe our approach to analyzing resilience of each breeding population. We relied primarily on estimates of abundance and population growth rate rather than specific habitat characteristics or documentation of threats. The cause(s) of the population decline on the Y-K Delta, which was the primary factor influencing the listing of the species, are unknown. In addition, stressors influencing current population growth and potential habitat limitations are not well-understood. Therefore, we provide a qualitative assessment of stressors and habitat condition to support the evaluation of population resilience, but do not define quantitative condition categories for these factors. We have information on population abundance and growth rate for ACP and Y-K Delta breeding populations, and therefore based condition categories on these parameters. There is considerable disagreement about interpretation and applicability of minimum population size “rules of thumb,” (Flather et al. 2011 p. 314, Jamieson and Allendorf 2012 pp. 578 – 579, Shoemaker et al. 2013 pp. 548 – 549). However, a minimum population size in the thousands is a reasonable goal for maintaining long-term persistence and evolutionary potential (Brook et al. 2006 pp. 378–379, Traill et al. 2007 p. 164, 2010 p. 315, Flather et al. 2011 p. 314, Reed and McCoy 2014 p. 869); therefore, we chose to categorize a population with at least 10,000, between 5,000 and 10,000, and fewer than 5,000 individuals, as being in high, moderate, and low condition, respectively. We chose to use the lower 95 percent confidence bound of the estimated population size as the metric for meeting a threshold, rather than the mean, to reduce the risk of misclassifying a population as in a higher condition than it is in reality. For example, a population with a mean estimate of 6,000 individuals and a lower confidence bound of 4,000 would be characterized as low rather than moderate condition. We chose positive, stable, and negative population growth rates to represent high, moderate, and low conditions,

respectively (Table 3). Rather than using the mean estimate² of population trend, trends were categorized by the 95 percent confidence interval around the mean: A positive trend is indicated by the lower bound of the 95 percent confidence interval being greater than an increase of one percent per year (i.e., lower 95 percent CI ≥ 0.01); a negative trend is indicated by the upper bound of the 95 percent confidence interval being lower than a decrease of one percent per year (i.e., upper 95% CI ≤ -0.01); and, stability is indicated by a confidence interval that meets neither of these thresholds (Sauer and Link 2002 p. 1746).

Few studies have been conducted on the Arctic Russia breeding population, and it is not regularly monitored (Section 3.3.2). The only measure of resilience available for the Arctic Russia population is an estimate of the minimum global population size obtained from occasional surveys during winter, from which a minimum size of the Arctic Russia population can be inferred. Due to the lack of information on population parameters, we felt it was not appropriate to assign the Arctic Russia breeding population to condition categories. Instead we simply report the available information on abundance in the section below.

Table 3. Categories developed to determine the condition of each spectacled eider breeding population in terms of resilience.

	Condition Category (Population Resilience)		
	High	Moderate	Low
Population abundance (Y-K Delta, ACP)	Lower 95% CI > 10,000 individuals	5,000 < Lower 95% CI < 10,000 individuals	Lower 95% CI < 5,000 individuals
Population growth rate¹	Positive	Stable	Negative

¹A positive trend is indicated by the lower bound of the 95 percent confidence interval being greater than an increase of one percent per year; a negative trend is indicated by the upper bound of the 95 percent confidence interval being lower than a decrease of one percent per year; stability is indicated by a confidence interval that meets neither of these thresholds (Sauer and Link 2002 p. 1746). Growth rate was calculated as the geometric mean rate of change.

3.3.2. Analysis and results

In this section we describe the historical and current abundance, population growth rate, demographic rates, and changes in distribution for each breeding population when such information is available. For the Alaska-breeding populations, we also compare current resilience of each breeding population with the condition categories described in Section 3.3.1.

² Geometric mean.

To characterize population abundance and growth rate, we primarily considered results of two recently-conducted analyses that used the best available data but different methods. First, (Dunham et al. 2021) used a Bayesian state-space model to estimate the abundance and growth rate of the Y-K Delta and ACP populations using count data from 2007 – 2019. Note that the analysis only used data from 2007 forward, as survey methods from 2007-2019 were consistent over time. Second, we (USFWS) used the available long-term datasets (data from all years) describing abundance and demographic rates of the Alaska-breeding populations to develop an integrated population model (IPM) linked to a population viability analysis (PVA) for the Y-K Delta and ACP breeding populations (IPM-PVA; Appendix B).

Yukon-Kuskokwim Delta Breeding Population

Historically, spectacled eiders nested in higher densities on the Y-K Delta than they do at present. Based on nest plot surveys conducted from 1961 to 1976, the density of spectacled eider nests was 4.4 nests/km² in the coastal region, 1.0 nests/km² in central region, and 0.2 nests/km² in the interior region (Dau and Kistchinski 1977 p. 70, Stehn et al. 1993 p. 264). Nest densities were multiplied by the total area of each region to estimate a breeding population size of approximately 50,000 spectacled eider pairs (100,000 individuals) nesting on the Y-K Delta annually (Dau and Kistchinski 1977 p. 70).

However, by 1992, aerial and nest plot survey observations suggested that the Y-K Delta spectacled eider breeding population had declined by approximately 96 percent (Stehn et al. 1993 p. 271), and the population size declined from 9 – 14 percent per year from 1985 – 1992 (Stehn et al. 1993 pp. 267 – 268). Evidence of the dramatic decline in spectacled eider nesting on the Y-K Delta was corroborated by Ely et al. (1994 p. 83), who observed a 79 percent decline in spectacled eider nests at a study site near the Kashunuk River between 1969 and 1992. Caution should be taken in relying on specific population sizes and rates of decline derived from survey data without incorporating estimates of observation error (e.g., Stehn et al. 1993), using observations from one study site (e.g., Ely et al. 1993), or extrapolating limited sampling to much a larger area (e.g., Dau and Kistchinski 1977). However, based on the weight of evidence, the Y-K Delta breeding population undoubtedly declined between the 1960s and the early 1990s. This decline formed the basis for listing the species as threatened under the ESA, even though the cause(s) of decline were not known (58 FR 27474).

Since listing, the USFWS has relied on two coordinated surveys to provide indices of spectacled eider abundance on the Y-K Delta. A standardized aerial survey was conducted in the coastal zone of the Y-K Delta from 1985-2019, where the majority of spectacled eiders nest (breeding pair survey; methods described in Fischer et al. 2018 p. 150 and Lewis et al. 2019 p. 193). This survey provided indices of population abundance, uncorrected for detection. In addition, a ground-based nest survey was conducted in a smaller subset of the Y-K Delta breeding area from 1985 – 2019 (nest plot survey; methods described in (Fischer et al. 2018). In this survey, the number of nests per plot was adjusted for imperfect detection using estimates of nest detection rates in relation to nest status (active or destroyed when found), nest site location, and crew experience (Fischer et al. 2018 pp. 151 – 152). Corrected nest estimates were then extrapolated to the entire Y-K Delta coastal zone (12,832 km²) for an estimate of nest abundance (see Fischer et al. 2018:152). A visibility correction factor (VCF) was

calculated from the ratio of ground-based nest counts to aerial pair counts based on a small sample of nests collected in a single breeding season in 1968, and was used to adjust the aerial counts upward to account for individuals not observed by air (3.58; (Stehn et al. 1993 p. 265, Lewis et al. 2019 p. 192). The VCF was uniformly applied to all aerial data, and thus did not account for variation in detection by year, observer, or ecological factors.

More recently, Lewis et al. (2019) improved the VCF used to correct aerial indices for detection rate. They examined general linear models describing variation in detection that included several ecological covariates. The best approximating model included spectacled eider density and survey timing, with density influencing the VCF to the greatest extent (Lewis et al. 2019 p. 200). Nest plot survey data were then used to stratify the aerial survey area into high, medium and low densities of spectacled eiders, and VCFs were calculated based on the ratio of nest density to the aerial breeding pair density in each of the three strata. The resulting VCFs were 3.09 (SE = 0.19) for high, 2.46 (SE = 0.17) for medium, and 1.35 (SE = 0.15) for low density strata (Lewis et al. 2019 p. 199), indicating that aerial detection decreases as eider density increases. Importantly, estimates of abundance using density-adjusted VCFs were consistently lower than estimates produced when using the traditional VCF (Lewis et al. 2019 p. 202). While this method is a significant improvement and its application provides more reliable abundance estimates from aerial surveys compared to previous measures, sources of variation and uncertainty in detection have not been completely addressed and may lead to biased estimates (Dunham et al. 2021).

Dunham et al. (2021 p. 3) used a Bayesian state-space model to estimate the abundance and growth rate of the Y-K Delta breeding population of spectacled eiders, using annual estimates of breeding birds corrected for detection according to methods in Lewis et al. (2019) as the observation data and error. The use of data from 2007 – 2019, rather than the full data set, because surveys were conducted using consistent methods on the ACP since 2007, and therefore the same time frame was used for the analysis of Y-K Delta data for consistency.

To reflect uncertainties in detection and observation process, Dunham et al. (2021) fit the aerial survey data to four competing state-space models to estimate abundance and growth of the breeding population. Uncertainties included the effect of using informative versus uninformative priors; the effect of including data from 2015, when the aerial survey count was inconsistent with ground observations possibly due to a new aerial observer; and, the effect of the assumption that detection is constant across years, as was assumed in Lewis et al. (2019). Models incorporating these factors included: (1) YKD1: use data from all years and include informative priors; (2) YKD2: use data from all years and use uninformative priors; (3) YKD3: exclude data from 2015 and use informative priors; and, (4) YKD4: use data from all years, informative priors, and allow for a latent observation process of annual variation in detection. A considerable portion of variation in the Y-K Delta population growth rates and process variation was attributed to the negative bias introduced by the 2015 data and the assumption of constant VCFs across years; therefore, we report the results of the YKD4 model as the best available information, as it takes into account latent variation in observation processes. The estimated mean abundance of the Y-K Delta breeding population in 2019 from model YKD4 is 16,113

eiders (95 percent credible interval (CRI¹): 12,313 – 21,352). This estimate represents adult spectacled eiders present in 2019 within the surveyed nesting grounds, and does not include non-nesting individuals from this breeding population which may have remained in the marine areas during the survey period. The estimated mean population growth rate of the Y-K Delta breeding population is 0.016 (95 percent CRI: -0.065 – 0.091) from 2007 – 2019.

We conducted a Bayesian population viability analysis derived from an IPM-PVA to estimate abundance and population growth rate, and evaluate the probability of extinction of the Alaska breeding populations given current conditions and future scenarios (See Appendix B for full methods). The IPM-PVA used the available demographic and population abundance data for each population using the entire time series (1988-2019) and incorporated model parameter uncertainties. Some spectacled eider demographic parameters have not been determined; in those cases, a formal expert elicitation process was used to develop estimates (see methods in Appendix C). The IPM-PVA model structure is similar to the YKD4 model, except that 2015 survey data was excluded and informative priors were developed from expert elicitation (Appendix B). From the IPM-PVA, the estimated mean abundance of the Y-K Delta breeding population in 2019 is 14,027 spectacled eiders (95 percent CRI = 9,781 – 18,257) and the mean annual population growth rate from 1988 – 2019 is 0.051 (95 percent CRI = 0.036 – 0.067; Appendix B). To provide a direct comparison of the growth rate reported in Dunham et al. (2021), the mean growth rate estimated by the IPM-PVA from 2007 – 2019 is 0.006 (95 percent CRI: -0.025 – 0.030).

In addition, Lewis et al. (2019), in their calculation of density-adjusted VCFs, found that the number of spectacled eiders in high and medium-density strata increased significantly from 1988 to 2015, but the number of eiders in the low density stratum remained relatively stable (p. 201). The low density stratum is the higher-elevation areas surrounding the lower-elevation habitats where the higher concentrations of spectacled eiders are found. This suggests that distribution of spectacled eiders on the Y-K Delta did not expand to other habitats as the population increased.

In summary, based on the Bayesian state-space model, the Y-K Delta breeding population has increased since the species was listed in 1993 (when the population was estimated to number less than 3,000 spectacled eiders; Stehn et al. 1993 p. 271), and there are likely more than 12,000 spectacled eiders in the population. The estimated mean population growth rate from 2007 – 2019 is positive, but the lower 95 percent confidence interval bound is not greater than one percent per year, suggesting that the population is stable. However, when considering the entire time frame from 1988 to 2019, and using an IPM, the average annual growth rate is 0.051 (95 percent CI 0.036 – 0.067), indicating that the population increased over the long-term (Appendix B). The population appears to have leveled, or is growing at a slower rate, in recent years.

¹ In Bayesian statistics, a credible interval is an interval within which an unobserved parameter value falls within a particular probability; in this case the probability that the true number falls between 12,313 – 21,352 is 0.95.

Arctic Coastal Plain Breeding Population

At the time of listing there were few data on abundance of spectacled eiders on the ACP. However, biologists were concerned that the ACP breeding population was declining at a similar rate as observed for the Y-K Delta population (USFWS 1993 p. 27475). The number of spectacled eiders observed nesting at a Prudhoe Bay survey area appeared to have declined by 80 percent from 1981 – 1991, although data were not sufficient to estimate a trend (Warnock and Troy 1992 p. 22). The apparent proportional decline, however, was similar to the decline in the number of nests on the Y-K Delta estimated by Stehn et al. (1993 p. 268, USFWS 1993 p. 27475). Reasons for the local decline at the Prudhoe Bay site, and decline on the Y-K Delta, are unknown.

Although the USFWS has conducted aerial waterfowl breeding pair surveys on the ACP since 1986, the timing of the original survey was too late in the season to adequately detect nesting spectacled eiders (Larned et al. 2006 p. 1). In 1992, a second, earlier survey designed specifically for eiders was initiated. In 2007, these two surveys were merged and consistent methods have been used from 2007 through 2019 (Wilson et al. 2018). Until recently, the detection rate of spectacled eiders had not been estimated; therefore, the aerial survey provided indices of abundance, which had an unknown relationship to true population size. Recently, however, Wilson et al. (2017) estimated detection probability of spectacled eiders using a double-observer aerial survey approach. They estimated detection probabilities using a model describing variation in detection probability based on an effect between front- and rear-seat observers, species, and group size. The resulting detection probabilities were 0.7096 (SE: 0.0760) for single birds, 0.7761 (SE: 0.0642) for pairs, and 0.8120 (SE: 0.0579) for small flocks. The application of these detection probabilities allows for the conversion of indices to estimates of abundance, thus representing a significant improvement to the use of indices. But, as with the Y-K Delta data above, sources of variation such as annual variability in detection have not been accounted for, which could still lead to biased estimates. In addition, the ACP detection rate is based on air-to-air comparisons, which cannot account for birds that are not observable by either aerial observer, whereas the Y-K Delta detection rates are based on air-to-ground comparisons, in which more birds are, theoretically, more available for detection.

To reflect uncertainties in detection and observation process, Dunham et al. (2021) fit aerial survey data to two competing state-space models to estimate abundance and growth of the ACP breeding population. They constructed two models for the ACP population to evaluate the effect of the assumption that detection probability is constant across years (i.e., in Wilson et al. 2017): (1) ACP1: use informative priors; and, (2) ACP2: allow for a latent observation process of annual variation in detection. Both models used informative priors and included data from 2007-2019. Similar to model YKD4 in the the Y-K Delta analysis above, the ACP2 model likely represents the best available information. The resulting mean abundance estimate in 2019 is 6,401 eiders (95 percent CRI: 3,766 – 9,750). The posterior mean for the growth rate of the ACP population from 2007 – 2019 was -0.005 (95 percent CRI: -0.092 – 0.082). Similar to the Y-K Delta estimates, this number primarily represents the individuals on the nesting grounds, and does not include juvenile or non-breeding adult spectacled eiders that may remain in marine areas during the survey period.

From the IPM-PVA using the available abundance and demographic data from 1988 – 2019, the estimated abundance of the ACP breeding population in 2019 is 5,408 spectacled eiders (95 percent CRI = 3,696 – 7,364), and the estimated annual growth rate of the ACP population -0.004 (95 percent CRI = -0.018 – 0.008; Appendix B). Restricted to 2007 – 2019, the mean annual growth rate is estimated as -0.025 (95 percent CRI: -0.055 – 0.004).

In another recent analysis, Amundson et al. (2019) estimated distribution, abundance, and population trend of spectacled eiders on the ACP using aerial survey data from 1992 – 2016. They analyzed count data by assigning bird observations to 6 x 6 km² grid cells and used models that accounted for seasonal variation in detection probability (i.e., variation in the start of nesting season and the timing of the survey). They assumed that detection is consistent for observers across years and space. Analyses indicated that densities of spectacled eiders were greatest near the Chukchi coast of the ACP (Figure 10), and density declined along the coast south of Point Lay over the duration of the dataset (Amundson et al. 2019 p. 6). Spectacled eider abundance was stable throughout much of their core range on the ACP, but numbers increased between Admiralty Bay and Teshekpuk Lake over the time period (Amundson et al. 2019 p. 7). Mean density estimated from this analysis was 0.24 spectacled eiders per km² (95 percent CI: 0.221 – 0.263), estimated total population size for 2016 was 5244 (95 percent CI: 3090 – 8260), and population trend was 1.013 from 1992 – 2016 (95 percent CI: 0.989 – 1.037; Amundson et al. 2019 p. Table A4.13).

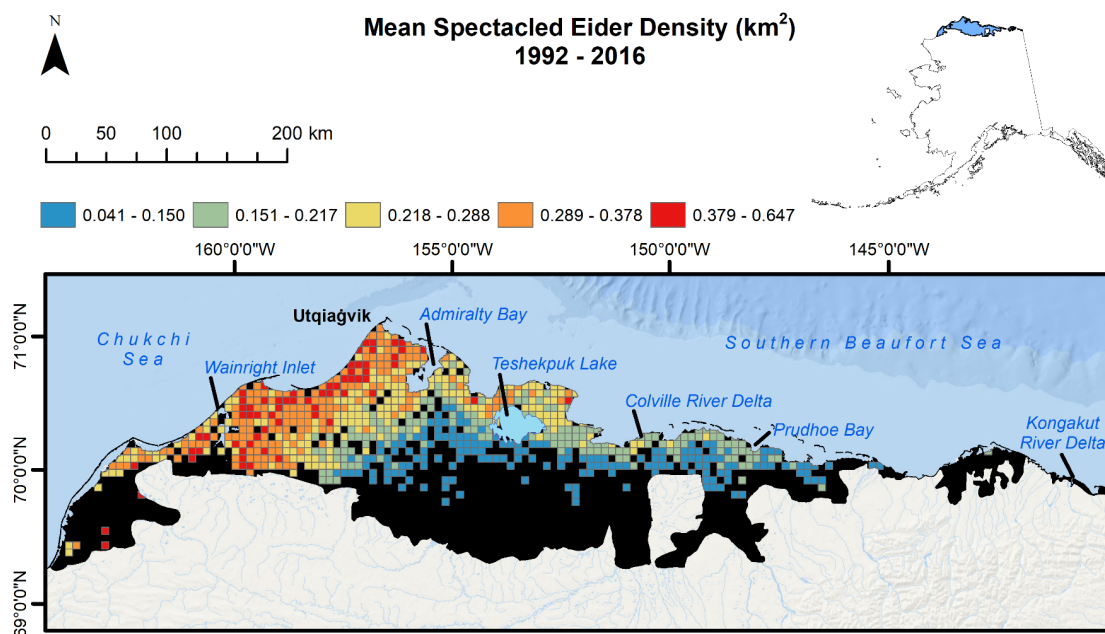


Figure 10. From Amundson et al. 2019, Appendix 5, Figure A5.9: Spectacled eider density (km²) across the Arctic Coastal Plain, Alaska, 1992-2016. Black represents areas with no observations.

In summary, the current abundance in the ACP breeding population is likely more than 3,500 individuals. We have limited information on the historical abundance of spectacled eiders in the ACP breeding population; however, the available information suggests that the breeding population has been stable since listing.

Arctic Russia Breeding Population

When the species was listed in 1993, data were insufficient to estimate population size or trend in Arctic Russia (58 FR 27475). Aerial surveys conducted from 1993 to 1995 produced an index of 146,245 spectacled eiders (coefficient of variation = 0.08, unadjusted for detection rate; Hodges and Eldridge 2001 p. 133) within 200 km of the Arctic coast between the Kolyuchin and Lena Rivers, where the majority of the population is thought to nest. The survey had to be conducted in three consecutive breeding seasons to cover the extensive distribution of spectacled eiders in Arctic Russia (over 157,000 km²; Hodges and Eldridge 2001 p. 133) and navigate political and logistical barriers. Approximately 20 percent of the spectacled eiders observed were in flocks, which is a large number compared to Y-K Delta and ACP surveys, in which flocks are rare (Stehn et al. 2006 p. 6). This suggests either there are a relatively large number of non-breeding spectacled eiders in Arctic Russia during the breeding season, or the survey timing was late relative to the nesting season (and failed breeders and males were flocking prior to departure to molting areas), potentially biasing the results.

In 2019, Russian biologists began conducting aerial waterfowl surveys in the range of spectacled eiders in Arctic Russia. Methods and survey area differed among years, but in 2020 protocol and transects were similar to those used by Hodges and Eldridge (2001). However, a record early spring in Russia and unprecedented tundra fires put into question whether the timing of the 2020 survey was appropriate. Final results from this survey are not yet available; however, initial comparison of densities observed in key breeding sites indicate substantially lower estimates in 2020 compared to the 1990s. For example at Indigirka Delta, the 1990s estimate was 46,000; the 2020 estimate was 3,000 (J. Fischer, USFWS, pers. comm.). While preliminary results suggest a possible decrease in population size, these data cannot be used to produce an estimate of population trend due to timing and survey design differences. Therefore, current Arctic Russia breeding population abundance and trend are unknown, other than what can be inferred from estimates of the wintering population, which comprises all three breeding populations of spectacled eiders.

Winter Surveys of the Global Population

Aerial surveys of the wintering area in the Bering Sea were conducted in 1995 – 1998, 2009, 2010, and 2020 to estimate the size of the global population (Larned et al. 2012, Safine and Bowman 2020). These surveys were designed as photo censuses, assuming the distribution of the global population was constrained to limited, observable areas of open water in the sea ice at the time the survey was conducted. Satellite telemetry and aerial surveys through 2010 suggested that individuals from all three breeding populations winter together in large, tight flocks in an area south of St. Lawrence Island in late February through March, and begin moving north to staging and breeding areas in late March/early April (Section 2.2.2). In most years, the survey area was delineated based on locations of spectacled

eiders that were marked with satellite transmitters in the previous summer. However, potential sources of error in these surveys have not been quantified. Errors that cause over-counting, such as double-counting and species misidentification, are thought to be small compared to sources of error that cause under-counting, such as the movement of birds prior to or during the survey, or the true winter distribution being larger than the survey area. Thus, we consider the results from these surveys as *minimum counts of the global population* of spectacled eiders rather than estimates of population abundance.

Prior to 2020, two surveys were conducted under optimal conditions and considered to be accurate characterizations of the minimum size of the global population. A photographic survey in 1997 provided the best count during surveys flown from 1995 – 1998, as it was conducted during nearly continuous ice coverage in the wintering area; it resulted in a minimum estimate of 363,030 spectacled eiders (95 percent CI = 333,526 – 392,532; Larned and Tiplady 1997:3 – 4, Larned et al. 2012:7; Table 4). The best survey flown from 2009-2010 was conducted in 2010 under similar conditions and resulted in a minimum estimate of 369,122 spectacled eiders (95 percent CI = 358,163 – 380,081; Larned et al. 2012:7; Table 4).

In 2020, another aerial survey was conducted using similar methods to those used in previous surveys (Safine and Bowman 2020). However, flock locations, flock size, and sea ice conditions were notably different. Flocks were located farther northeast and closer to the south coast of St. Lawrence Island than in previous surveys. No flocks were encountered in the southwest portion of the survey area, near the area identified as the “core wintering area” based on satellite transmitter locations from the 1990s (Petersen and Douglas 2004 p. 84). The 2020 survey also encountered more flocks with fewer individuals per flock. Of 105 flocks, 78 percent were comprised of less than 1,000 individuals, and the largest flock contained 5,227 individuals. Maximum flock size from previous surveys ranged from 21,000 to 151,000 individuals. Despite a high overall percentage of sea ice cover in the survey area, the northern portion had dynamic sea ice conditions with numerous sea ice openings, resulting in smaller, dispersed flocks. Additionally, during the survey on 3-4 March, flocks were seen along the eastern border of the survey area. To examine this, a higher altitude reconnaissance flight was conducted on 8 March, after three days of inclement weather, to determine if large flocks were present east of the survey area. Large (greater than 10,000 individuals) flocks were not detected, but flocks of hundreds to low thousands were present outside the survey area. These flocks were not enumerated or included in the total count because it was unknown if these flocks had been encountered during the 3 – 4 March survey, as sea ice had shifted substantially between 4 and 8 March. Another contributing factor to a lower count was a lower number of active PTT transmitters just before and during the survey period, making it more likely that a portion of the population may not have been included in the survey area that was defined based on a small sample of PTT-marked birds. Taken together, the differences in flock location, flock size, sea ice dynamics, and sample size of marked birds, indicate that the 2020 survey area did not capture the entire population (Safine and Bowman 2020 pp. 22–23).

Additionally, detection within the survey area was likely less than 100 percent. Larger flocks are more detectable, and spectacled eider flocks observed in this survey were consistently small. Transects were

placed 7.8 km apart, assuming a maximum detection distance of 3.9 km; however, most flocks were encountered within 2 km perpendicular distance from a transect (Safine and Bowman 2020 p. 23). Despite high overall percentage of sea ice cover in the survey area, the northern portion where spectacled eiders were present in 2020 (but not in previous years) had larger areas of open water. This resulted in a larger search area and reduced visibility of flocks, and further compromised detection at times.

In summary, results of all winter surveys of spectacled eiders, including the 2020 count (76,952 spectacled eiders in 105 flocks, Table 4; USFWS, unpublished data) should be considered minimum population counts, as they did not account for detection probability of flocks within or outside of the study area. While a decline in the Arctic Russia breeding population cannot be ruled out as a possible contributor to the lower count, due to the detection issues described above it is not appropriate to compare this count with counts in previous years to make inference about population growth rate.

Table 4. Results from late winter surveys of Spectacled Eiders in the Bering Sea. Data from Larned et al. 1995b, 2012; Larned and Tiplady 1997, 1999; USFWS, unpublished data. The 95% confidence interval reported for surveys from 1995 – 2010 is a calculation of the error associated with estimating flock size from photographs; it does not include error associated with availability of birds for counting (e.g., detection rate of flocks within or outside of the survey area). For 2020, all spectacled eiders were counted directly, and not estimated using a sampling approach; therefore, there is no estimate of precision/confidence interval associated with this count.

Survey Date	Number of Flocks	Minimum Flock Size	Maximum Flock Size	Minimum Global Population Size	95% Confidence Interval
13-16 March 1995	20	--	30,000	NA ^a	NA
05 April 1995	40	527	20,905	148,059	137,136 – 158,982
23 March 1996	115	--	--	226,810 ^b	NA
08 March 1997	19	800	150,966	363,930	334,426 – 393,432
11 March 1998	43	--	--	374,792	371,278 – 378,305
21 March 1998	50	--	--	282,311	279,300 – 285,322
01-02 April 2009	84	25	30,984	305,261	298,646 – 311,876
14-16 March 2010	50	24	94,633	369,122	358,163 – 380,081
03-04 March 2020	105	10	5,227	76,952 ^c	NA

^a Not a population survey. Aerial reconnaissance that first documented winter distribution.

^b Only visual counts made due to low sea ice cover resulting in a low confidence estimate.

^c Considered a population census with all individuals encountered during survey counted; therefore, there is no measure of precision associated with this number.

Minimum size of Arctic Russia breeding population

Because abundance information on the Arctic Russia breeding population is not available, we subtracted the combined estimates of the Alaska populations (Y-K Delta and ACP) from the count of the global population (Y-K Delta, ACP and Arctic Russia combined) to calculate the minimum size of the Arctic Russia population. Prior to this calculation we converted the Alaska breeding population estimates obtained in May/June 2019, which consisted only of breeding adults, to population estimates at the time of the winter survey (March 2020), which consisted of breeding birds, non-breeding birds, and hatch year birds. Adult birds, *AHY*, composed of breeding and non-breeding birds, was estimated as BB/BP , where BB is the breeding bird population size and BP is breeding propensity. For BB_{YKD} and BB_{ACP} , we used the upper 95 percent confidence interval bound of the 2019 breeding bird estimates on the Y-K Delta ($N = 18,257$) and ACP ($N = 7,364$), respectively. We assumed breeding propensity is 0.78 based on a long-term study of common eiders (Coulson 1984 p. 531). We assumed that the majority of annual adult mortality occurs between the breeding season and when the global population is counted (early March). Adult birds surviving to the count were represented by ϕAHY , where $\phi = 0.88$ is an estimate of adult apparent survival based on female birds not exposed to lead on Kigigak Island (Flint et al. 2016 p.25). Finally, we assumed hatch year birds in each breeding population that are counted in the winter survey are estimated as $\theta(\phi AHY)$ where θ is the age ratio between *HY* (hatch year) and *AHY* birds, estimated as 0.08 (estimated by dividing total *HY* males by *AHY* males in photographed flocks; Larned et al. 2012 p. 7). We also assumed that these parameters were the same for all three populations and were reasonable estimates for how the 2019 population size changes between the breeding season and late winter, when the global population is counted. Therefore, each population's contribution to the global population at the time of the winter survey was estimated as:

$$Nb_i = \phi \frac{BB_i}{BP} (1 + \theta)$$

where Nb is the estimated total birds in each population, i . The Arctic Russia population Nb_{AR} , where N_{global} is the minimum count from the 2020 winter population survey, was estimated as:

$$Nb_{AR} = N_{global} - (Nb_{YKD} + Nb_{ACP})$$

Following this methodology, the Arctic Russia population in March 2020 was comprised of *at least* 45,734 spectacled eiders, recognizing that this is an underestimate of the true population size.

Demographic changes in Arctic Russia

Solovyeva et al. (2018 p. 6) documented an annual decline in spectacled eider nest density of 8 percent at Chaun from 2009 – 2016. The decline in nest densities was attributed to a combination of: (1) increases in large gulls (Vega and glaucous gulls) that depredate spectacled eider eggs; and, (2) declining nesting populations of Sabine's gulls and Arctic terns, with which spectacled eiders have a nesting association. Spectacled eiders are hypothesized to gain protection from nest predators such as Arctic fox by nesting near these defensive species (Solovyeva and Zelenskaya 2016 p. 848, Solovyeva et al.

2018b pp. 7 – 8). This study is limited to one site within a widely distributed breeding population (Figures 1 and 2), but may suggest that the nesting effort and/or the spectacled eider population at this local site are decreasing in Russia due to a changing ecological community. How this potential decrease at one site relates to the larger Arctic Russia breeding population is not known and warrants further study. We have no information on whether other demographic parameters or nesting distribution in Arctic Russia have changed since listing.

In summary, information on abundance, distribution, and trend of the Arctic Russia breeding population is extremely limited. At the time of listing, the number of spectacled eiders in this population was unknown; winter surveys in the 1990's and 2010 indicated over 200,000 spectacled eiders in the population. The only information to evaluate current condition of this breeding population is the most recent estimate of the minimum global population size, less the Alaska breeding population estimates, which suggests the Arctic Russia population consists of at least 45,000 spectacled eiders. Beyond this crude estimate, we have no information from which to characterize its current level of resilience.

3.3.4. Summary of Resilience

In summary, by comparing the best available information on population abundance and growth rate with the condition categories described in Section 3.3.1, we consider the current resilience of the Y-K Delta breeding population to be moderate-high, and the current resilience of the ACP breeding population to be low-moderate (Table 5). The estimated minimum size of the Arctic Russia population suggests high resilience, but without concurrent estimates of population growth rate, we do not have sufficient information to adequately characterize its current level of resilience.

3.4. Representation

Representation is the ability of a species to adapt to changes in its environment. We evaluated the representation of spectacled eiders by determining the number of ecotypes used throughout the annual cycle. We also discuss the possible range of behavioral flexibility in response to environmental conditions and the genetic diversity of the species.

Table 5. The current condition of spectacled eider breeding populations based on an integrated population model (Appendix B) and Bayesian state-space model (Dunham et al. 2021).

Population	Analysis	Abundance (Y-K Delta and ACP) or minimum population count (Arctic Russia)/ Resulting condition category	Population Growth Rate 2007-2019/ Resulting condition category	Overall Level of Resilience
Y-K Delta	IPM-PVA (Appendix B)	14,027 (95% CRI = 9,781 – 18,257)/ MODERATE	0.006 (95% CRI = -0.025 – 0.030)/ MODERATE	MODERATE-HIGH
	Bayesian state-space model (2007-2019; Dunham et al. 2021)	16,113 (95% CRI = 12,313 – 21,352)/ HIGH	0.016 (95% CRI = -0.065 – 0.091)/ MODERATE	
ACP	IPM-PVA (Appendix B)	5,408 (95% CRI = 3,696 – 7,364)/ LOW	-0.025 (95% CRI -0.055 – 0.004)/ MODERATE	LOW-MODERATE
	Bayesian state-space model (2007-2019; Dunham et al. 2021)	6,401 (95% CRI = 3,766 – 9,750)/ LOW	-0.005 (95% CRI = - 0.092 – 0.082)/ MODERATE	
Arctic Russia	calculation in this section	minimum population size: 45,734/ HIGH	N/A	unknown

The current distribution of spectacled eiders has remained largely the same as the historical distribution described in USFWS (1993). During the breeding season, spectacled eiders continue to occupy both subarctic (Y-K Delta) and Arctic (Arctic Russia and ACP) habitats. Spectacled eiders also continue to use all four identified molting areas (Sexson et al. 2016 p. 466; USFWS unpublished data). Given that molting areas are hundreds to thousands of miles apart, the benthic prey communities and food availability of the four molting areas likely differ. Researchers thought that spectacled eiders, while moving locally in response to sea ice coverage, were primarily limited to the area south of St. Lawrence

Island (Petersen et al. 2000 p. 2). This likely remains true in years of extreme sea ice concentration and extent. However, recent data suggest they occupy a much larger area, from the historical wintering area to as far north as the East Siberian Sea, when sea ice retreats. This information suggests that some spectacled eiders are able to change their behavior by moving large distances when not constrained by sea ice. However, we do not yet know the consequences of this behavior; i.e., whether moving in response to sea ice retreat is adaptive in that it allows for the persistence of those exhibiting the behavior.

We have little information to evaluate the genetic adaptive capacity of spectacled eiders. However, evaluations of spectacled eider genetic population structure do not suggest limited genetic diversity in the species (Scribner et al. 2001 p. 2109, Sonsthagen et al. 2019 p. 3), and recent analyses suggest more gene flow between breeding populations than was previously thought (Sonsthagen et al. 2019). In addition, the limited information on spectacled eider diet suggests that while spectacled eiders are benthic prey specialists, they can rely on different species depending on availability (Section 2.3).

Overall, the current representation of spectacled eiders can be characterized by the following points: 1) spectacled eiders are widely distributed in at least three ecotypes during the breeding season (Y-K Delta, ACP, and Arctic Russia); 2) they use four molting areas with benthic communities that likely differ in composition; 3) they may be able to shift their distribution in response to changing sea ice conditions in the Bering Sea in winter; 4) evidence supports that they shift their diet depending on availability of benthic bivalve species; and, 5) there is no indication that genetic diversity is limiting their adaptive capacity.

3.5. Redundancy

Redundancy is the ability of a species to withstand catastrophic events, and is described by the number and distribution of resilient populations. The global population of spectacled eiders continues to be comprised of three breeding populations that use at least four different molting areas spanning a relatively large and geographically diverse area. However, the high degree of female breeding area fidelity and natal philopatry (Section 2.2.3) makes it unlikely for spectacled eiders to re-colonize an area should the population become extirpated. In addition, historically, the entire global population used one wintering area in the Bering Sea where birds were concentrated in openings in the sea ice, which may make them susceptible to catastrophic events such as a disease outbreak. In the last decade there have been fewer years with extreme sea ice conditions and wintering flocks appear to be more dispersed in milder years (USFWS, unpublished data). While extreme sea ice conditions are predicted to still occur in some years, a wider winter distribution increases the ability of the species to withstand a catastrophic event.

3.6 Summary of Current Condition

In summary, spectacled eiders occupy remote habitats that are largely intact. However, some factors may be affecting habitat quality or may directly affect individual spectacled eiders. For many of these factors, we have limited information with which to quantify their effects and determine if they are significant population-level threats. Two factors, exposure to lead and climate change, have been shown to affect demographic rates. Climate change has also altered both the tundra breeding habitat and the marine non-breeding habitats of spectacled eiders. There is evidence that the northern Bering Sea in particular is undergoing drastic changes due to a reduction in sea ice and warming temperatures. This may have consequences for spectacled eider food availability and thermoregulation in winter; however, whether spectacled eiders can behaviorally adapt and find adequate foraging conditions is unknown.

Because the effects on spectacled eiders from changes in habitat quality due to climate change are largely speculative at this time, in this SSA we used measures of population abundance and growth rate to evaluate resilience. These measures should reflect the effects of influential factors and habitat changes on the population resilience. Given the available information, we consider the current resilience of the Y-K Delta population moderate - high, and the current resilience of the ACP population low - moderate. The information about Arctic Russia breeding population abundance and growth rate is insufficient to determine the level of population resilience.

The global population of spectacled eiders is comprised of three geographically separate breeding populations, one with moderate-high resilience (Y-K Delta), one with low-moderate resilience (ACP) and one with unknown level of resilience (Arctic Russia). Spectacled eiders use at least four different molting areas. However, in winter the global population is occasionally restricted to a relatively small area with openings in the sea ice in the Bering Sea, reducing their level of redundancy during this period of their life cycle. Although we do not have the data to evaluate the genetic adaptive capacity of spectacled eiders, representation of the species can be described by the following factors: distribution in three ecotypes during the breeding season, indications that some individuals shift distribution in response to changing sea ice conditions, and their ability to use different bivalve species depending on availability.

Table 6. Summary of the state of knowledge of spectacled eiders at listing (1993), when the last 5-year review was conducted (2010), and currently (2020).

Measures of Abundance and Growth Rate	At listing (1993)	2010	2019-2020
Global	unknown	Aerial survey of wintering area in Bering Sea in 1996 and 1997 indicated a minimum population size of 363,000 spectacled eiders (Petersen et al. 1999).	Aerial survey of wintering area indicated minimum population size of 363,030 spectacled eiders in 2010 (Larned et al. 2012). An aerial survey in 2020 indicated minimum global population size of 76,952 spectacled eiders; however, sea ice conditions and other factors may have contributed to the difference between 2010 and 2020 counts.
Y-K Delta	Historically, extrapolated from nest plot surveys, 50,000 breeding pairs thought to be in the Y-K Delta population (Dau and Kistchinski 1977). Population had declined by approximately 96% to 2,000 pairs by 1992 (Stehn et al. 1993).	Indices of population abundance from aerial survey, not corrected for detection error, indicated approximately 6,000 spectacled eiders in 2009 (USFWS 2010). Approximately 7200 spectacled eider nests were estimated for that same year (equivalent of at least 14,400 individuals; USFWS 2010).	The mean estimated abundance (corrected for detection) in 2019 was 14,027 (95% CRI = 9,781 – 18,257; Appendix B), and the annual population growth rate from 1988 – 2019 was 0.051 (95% CRI = 0.036 – 0.067; Appendix B). However, the annual population growth rate from 2007 – 2019 was estimated as 0.016 (95% CRI = -0.065 – 0.091; Dunham et al. 2021), indicating that the population growth rate may have slowed in recent years.
Arctic Coastal Plain	Unknown, but one breeding site showed similar decline as YKD (Warnock and Troy 1992).	Indices of population abundance from aerial survey indicated approximately 5,000 spectacled eiders (USFWS 2010).	The mean estimated abundance (corrected for detection) in 2019 was 5,408 eiders (95% CRI = 3,696 – 7,364; Appendix B), and the annual population growth rate from 1992 – 2019 was -0.004 (95% CRI = -0.018 – 0.008; Appendix B).
Arctic Russia	Aerial survey conducted from 1993 – 1995 indicated approximately 146,000 breeding birds, unadjusted for detection (Hodges and Eldridge 2001).	Rough estimate deduced from winter survey of global population indicated greater than 300,000 spectacled eiders (Larned et al. 2012).	At least 45,734 spectacled eiders in the Arctic Russia population deduced from the winter survey of global population and estimated breeding population sizes of Y-K Delta and ACP.

Biology			
Population Dynamics	unknown	High female-biased philopatry and breeding site fidelity (Scribner et al. 2001). Males have lower fidelity to breeding areas.	Adult female survival most important driver of population dynamics (Flint et al. 2016).
Demographic Rates	Nest ecology study at upper Kashunuk (Dau 1974); nest plot surveys conducted on the Y-K Delta from 1967 – 1970 and 1988 – 1992. Nest ecology study at Prudhoe Bay 1981– 1991 (Warnock and Troy 1992).	Long-term nest ecology and mark-recapture projects at Kigigak, Kashunuk, and Chaun produced estimates of clutch size, nest survival, brood survival and adult female survival. Short-term nest ecology study at Indigirka.	Spectacled eider nest ecology data collected at Utqiagvik from 2009 - 2019, continued at Kigigak until 2015.
Distribution	Y-K Delta thought to be primary breeding area; Non-breeding distribution (molt, winter, migration) unknown.	Wintering and molt sites discovered and described (Petersen et al. 1999). Surveys of Arctic Russia conducted over 3 years documented wide distribution of that breeding population (Hodges and Eldridge 2001).	Shifts in molting area distribution (Sexson et al. 2016) and changes to winter distribution related to sea ice extent (USFWS, unpublished data) were documented. No additional information about Arctic Russia breeding distribution.
Threats/Stressors			
Causes of Y-K Delta decline	Unknown	Unknown	Unknown
Lead exposure	Unknown	Lead exposure shown to affect adult female survival rates at local sites on Y-K Delta (Franson et al. 1995; Flint et al. 1997, Grand et al. 1998).	Lead exposure shown to affect vital rates at the subpopulation level (Flint et al. 2016).
Effect of sea ice conditions in wintering area	n/a	Winter sea ice severity negatively associated with breeding population abundance (Petersen and Douglas 2004). Energetic modeling suggested that the presence of moderate coverage of sea ice reduces energetic costs (Lovvorn et al. 2009, 2014).	Winter sea ice severity is negatively associated with adult female survival (Flint et al. 2016; Christie et al. 2018). Low sea ice extent may also be negatively associated with female survival (Christie et al. 2018; Appendix B).

4. Future condition

4.1. Y-K Delta and ACP Breeding populations

4.1.1. Future Scenarios

To evaluate future condition of spectacled eiders, we developed four future scenarios that incorporated the plausible range of effects from two factors previously shown to impact spectacled eider demographic rates: exposure to lead on the breeding grounds (Y-K Delta population only) and sea ice conditions in the 1990s wintering area south of St. Lawrence Island (Y-K Delta and ACP populations). The scenarios were applied to projections of population size of the Y-K Delta and ACP breeding populations in an IPM-PVA. A detailed description of methods for this analysis can be found in Appendix B. There are not adequate data to conduct an IPM-PVA on the Arctic Russia breeding population; thus, we discuss future condition of that population using qualitative methods below.

Lead exposure has been shown to affect survival of adult females in the Y-K Delta breeding population, and is thought to be a continuing problem there, whereas lead exposure may occur on the ACP but at lower rates (Section 3.1.1). Therefore, we only applied predictions of lead exposure to the Y-K Delta population in future scenarios. Flint et al. (2016) estimated the probability of lead exposure, defined as blood lead level greater than 0.2 parts per million, in the breeding female Y-K Delta population at the time of hatch from 1994 – 1999 as 0.118. The model in Flint et al. (2016) linked observed lead exposure rates at certain locations to the distance to villages and major rivers, and extrapolated the observed relationship spatially across the Y-K Delta. Using this information as a baseline, we assumed that the spatial distribution of breeding birds and lead deposition is proportionately constant through time. We assume that by continuing and/or increasing outreach efforts the deposition of lead will decrease on the Y-K Delta, and lead currently in wetlands will settle into the sediment over time (Flint and Schamber 2010 p. 149). However, the rate of decline is unknown due to uncertainty around future deposition and settlement rates. Therefore, we used two alternative hypothesized lead exposure curves that vary by the year in which new lead deposition ceases:

- 1) Lead 2026: This scenario assumes that lead deposition will cease in 2026, after additional management actions are conducted following the 5-year review. Community-led bans on the use of lead waterfowl ammunition in western Alaska occurred in the mid-2000s and a comprehensive outreach program was conducted by Alaska Department of Fish and Game (ADFG) and USFWS for several years until about 2008. We expect that lead deposition has decreased due to these efforts, but observations of lead shot being sold in stores within and near spectacled eider habitat in Alaska indicates the use of lead ammunition has not ceased.
- 2) Lead constant: This scenario assumes that lead deposition does not stop and will continue to occur such that it is constant over time.

For each scenario, we assume a 10-year decay rate in the availability of lead in wetlands (Flint et al. 2016 p. 25). The effect of lead exposure on female survival in the model is represented by a prior based on effect estimates in Flint et al. (2016) that incorporates sampling variance.

Analysis of existing data sets suggest sea ice conditions in the 1990s wintering area may affect adult female survival rates (Flint et al. 2016, Christie et al. 2018), and have been hypothesized to affect breeding propensity (Petersen and Douglas 2004). This effect has been characterized as the number of observed extreme sea ice days (greater than 95 percent coverage of the 1990s wintering area; Petersen and Douglas 2004:82). Future estimates of extreme sea ice days relative to RCP4.5 and RCP8.5 climate scenarios are considered for both the Y-K Delta and ACP breeding populations by modeling linear and quadratic effects (similar to Christie et al. 2018). Priors for the effect sizes were centered on zero. The RCP4.5 and RCP8.5 climate scenarios assume radiative forcing (i.e., the net gain in warming of the Earth) stabilizes at 4.5 and greater than 8.5 Watts per meter squared by 2100, respectively (IPCC 2014 p. 124).

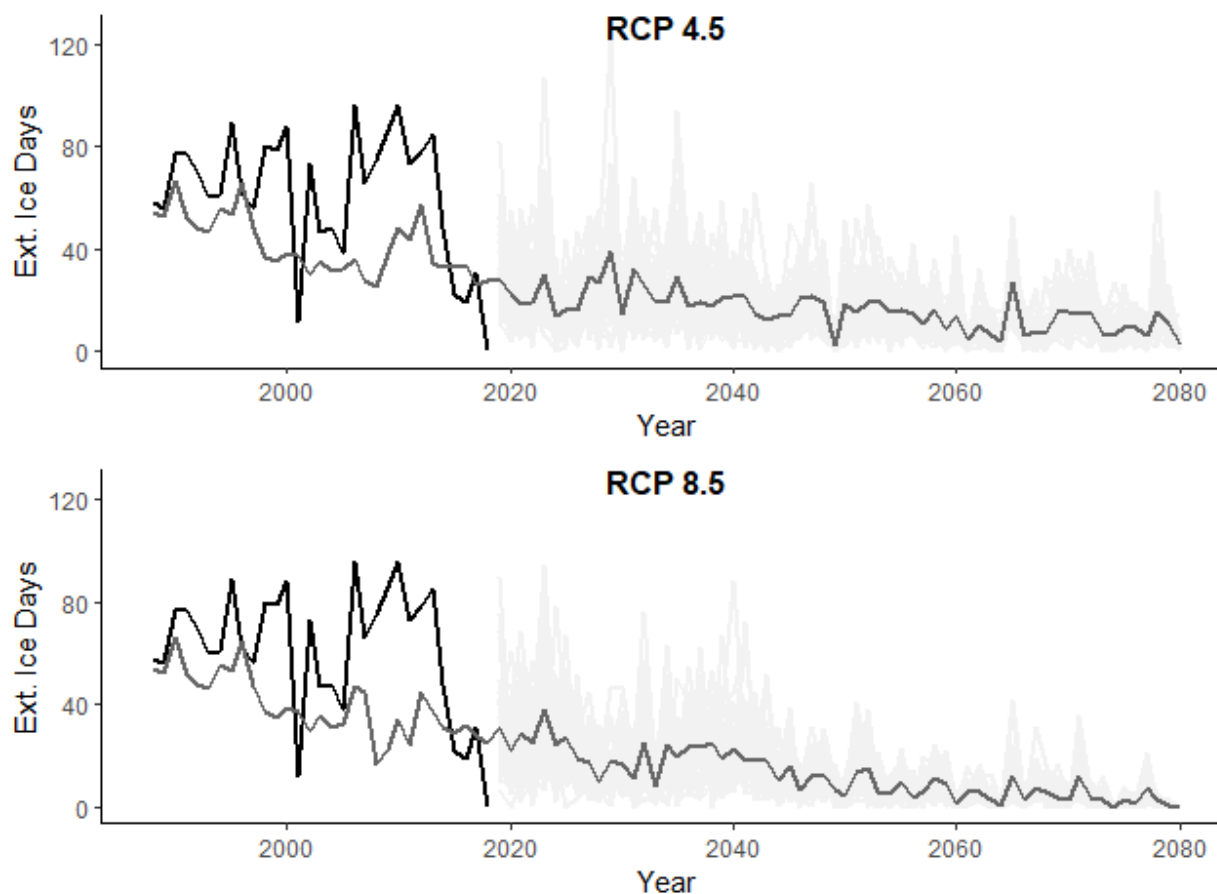


Figure 11. Number of extreme sea ice days, defined as greater than 95 percent coverage of the 1990s wintering area, generated using the RCP4.5 (A) and RCP8.5 (B) climate change scenarios. Observed (black) and RCP model-

averaged estimate (dark grey) of extreme sea ice days, and random draws from model averaged estimates (light grey) used in population projections.

4.1.2. Methods

The IPM-PVA model incorporates the available demographic and population abundance data for the Y-K Delta and ACP populations and model parameter uncertainties. For the Y-K Delta, count estimates, derived from aerial survey data that are corrected for detection using an average visibility correction factor across years and observers derived from ground surveys (Lewis et al. 2019), are available from 1988 – 2019. For the ACP, count estimates were based on aerial observations from 2007-2019 adjusted for detection probabilities from a double observer method (Wilson et al. 2018). Long-term nest survival, clutch size, and mark-recapture data collected at Kigigak from 1992 – 2015 (Flint et al. 2016 and USFWS, unpublished data) were used to estimate survival probabilities and develop a prior for average annual fecundity, less breeding propensity. Adjustments were made to these parameters to apply to the wider Y-K Delta and ACP populations through expert elicitation and inclusion of a local dispersal parameter. Some spectacled eider demographic parameters have not been estimated; in those cases, a formal expert elicitation process was used to develop priors for the IPM-PVA (see Appendix C).

The IPM-PVA resulted in a posterior distribution of breeding population size projections, from which 40,000 samples were taken. The percentage of samples in which the population reached quasi-extinction (QE) by 2060 (40 years, or 5 spectacled eider generations) and by 2100 (80 years, or 10 spectacled eider generations), was reported. We chose to define QE as 250 adults remaining in a breeding population, which was identified in the Plan as the level at which extreme measures such as captive breeding would be required to prevent population extirpation (USFWS 1996 p. 34). We based the length of projections on generation length, defined as the average age of parents of the current cohort (D’Elia and McCarthy 2010 pp. 753–754) (IUCN 2012 p. 11). Most spectacled eiders begin breeding at age three, and the lifespan of a spectacled eider is likely greater than four years (Petersen et al. 2000 p. 12); thus, generation length must be at least 4 years. The common eider, a related sea duck species, has an average lifespan of 7.36 years (SE = 0.97, Goudie et al. 2020). The International Union for Conservation of Nature (IUCN) assessment of spectacled eiders estimated generation time as 8.6 years, but no supporting information can be found to justify this number (BirdLife International 2018 p. 13). Based on the above, we assume generation time of spectacled eiders is approximately 8 years.

4.1.3. Results and Discussion

Under all scenarios for both populations, the number of simulations in which population size falls below 250 individuals is less than 1 percent before 2060 (Tables 7 and 8).

Table 7. Percent chance that the Y-K Delta breeding population of spectacled eiders will reach QE (250 adults) for future scenarios of lead exposure and sea ice extent in the Bering Sea wintering area from a population viability analysis.

Future Projections for Y-K Delta Breeding Population				
	RCP4.5, Lead 2026	RCP4.5, Lead Constant	RCP8.5, Lead 2026	RCP8.5, Lead Constant
by 2060 (5 generations)	0.05%	0.20%	0.06%	0.56%
by 2100 (10 generations)	1.72%	6.68%	3.50%	14.58%

Table 8. Percent chance that the ACP breeding population of spectacled eiders will reach QE (250 adults) under future scenarios of sea ice extent in the Bering Sea wintering area using a population viability analysis.

Future Projections for ACP Breeding Population		
	RCP4.5	RCP8.5
by 2060 (5 generations)	0.51%	0.70%
by 2100 (10 generations)	5.90%	8.78%

For the Y-K Delta breeding population, the QE probability increases as the duration of lead exposure lengthens, and the relationship between lead exposure and QE is more pronounced under the RCP8.5 scenario than the RCP4.5 scenario (Figure 13; Table 7). If lead deposition ceases in 2026, the QE probability at 2060 and 2100 for the Y-K Delta population remains less than 4 percent for both sea ice scenarios. If lead continues to be deposited into wetlands (models with constant lead), the QE extinction probability is approximately four times as high for both sea ice scenarios. In the scenario with constant lead deposition, and using RCP8.5 to project sea ice conditions, the QE probability is greater than 14 percent by 2100 (Table 7).

For the ACP breeding population, by 2100, the QE probability increases to almost 6 percent in the RCP4.5 scenario, and is greater than 8 percent in the RCP8.5 scenario (Table 8).

In both sea ice scenarios for both populations, the mean breeding population size is projected to decrease over time, indicating a negative relationship with decreasing number of extreme sea ice days (Figures 12 and 13). Based on the models for both Y-K Delta and ACP populations, model estimates suggest negative effects of extreme sea ice days at the posterior mean for juvenile and adult survival. The posterior of the quadratic ice effect on juvenile survival was negative in 57.8 percent Y-K Delta model outcomes (-0.017, 95 percent CRI: -0.192 – 0.169), and negative in 83.4 percent of the ACP model outcomes (-0.088, 95 percent CRI: -0.261 – 0.095). The posterior of the quadratic ice effect on adult survival was also negative, and similar for both populations; for the Y-K Delta the posterior was 65.9 percent negative (-0.039; 95 percent CRI -0.0225 – 0.150), and for the ACP the posterior was 69.0 percent negative (95 percent CRI = -0.234 – 0.142; Appendix B). In summary, these population projection models support previous studies suggesting a negative effect of decreasing extreme sea ice conditions on spectacled eiders; however, the uncertainty around this effect is large and includes zero. The near-term QE probability, given the model structure, assumptions and data, is relatively low through 2060, but increases by 2100.

Despite declining population sizes, projected population sizes are expected to remain robust, using the criteria set out to characterize population resilience (section 3.3.1). Under the four scenarios explored for the Y-K Delta population, there is at least a 41% chance of the population being over 10,000 in 2060 (at least 56%, if lead deposition ceases in 2026). Similarly, there is at least a 53% chance of the ACP population being over 5,000 in 2060 under either climate scenario explored.

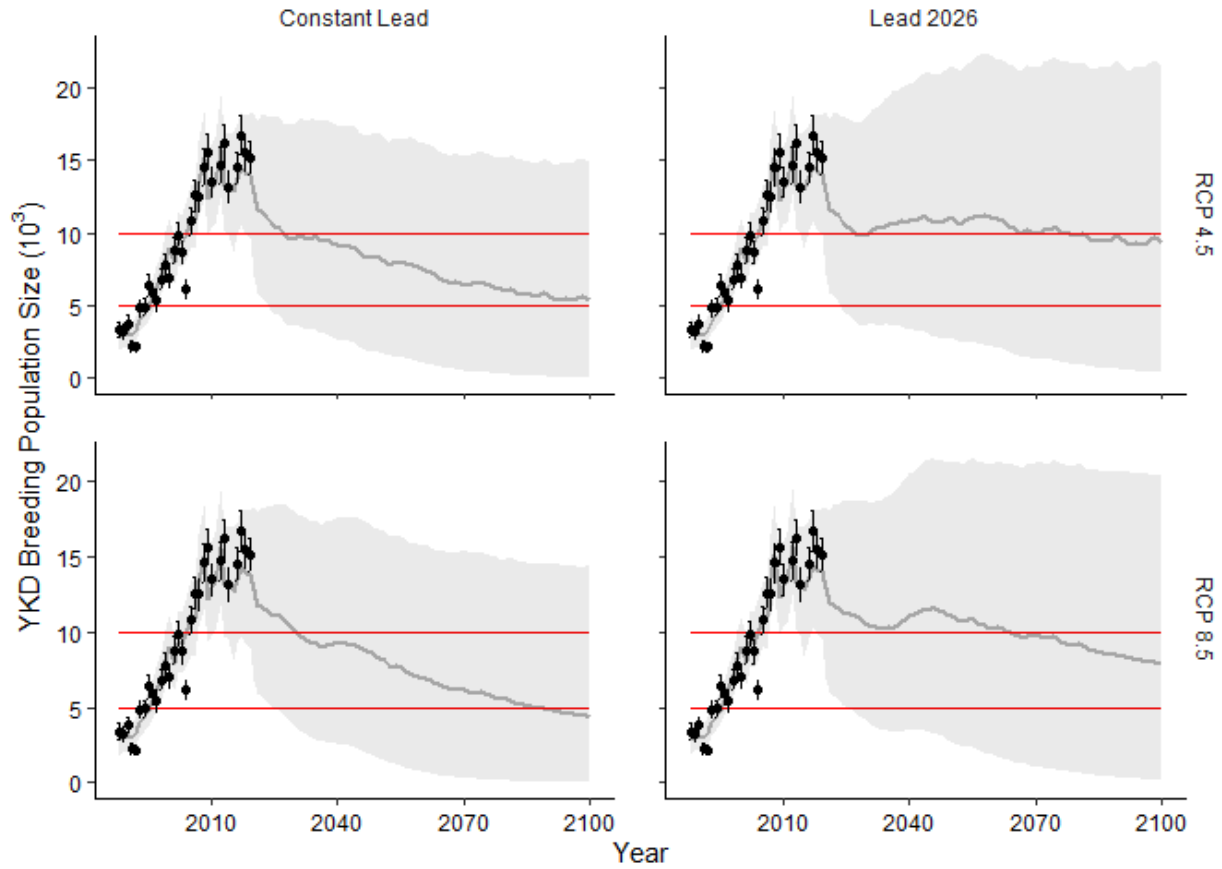


Figure 12. Estimated mean population projections for the Y-K Delta breeding population (dark grey line) under alternative lead (constant and declining after 2026) and winter sea ice condition (RCP 4.5 and RCP 8.5) scenarios. 95% credibility intervals given by the light grey areas. Observed counts (± 1 SE) are shown in each panel as points and error bars. Red lines indicate population levels associated with resilience criteria outlined Section 3.3.1 ($N = 5,000$ and $10,000$).

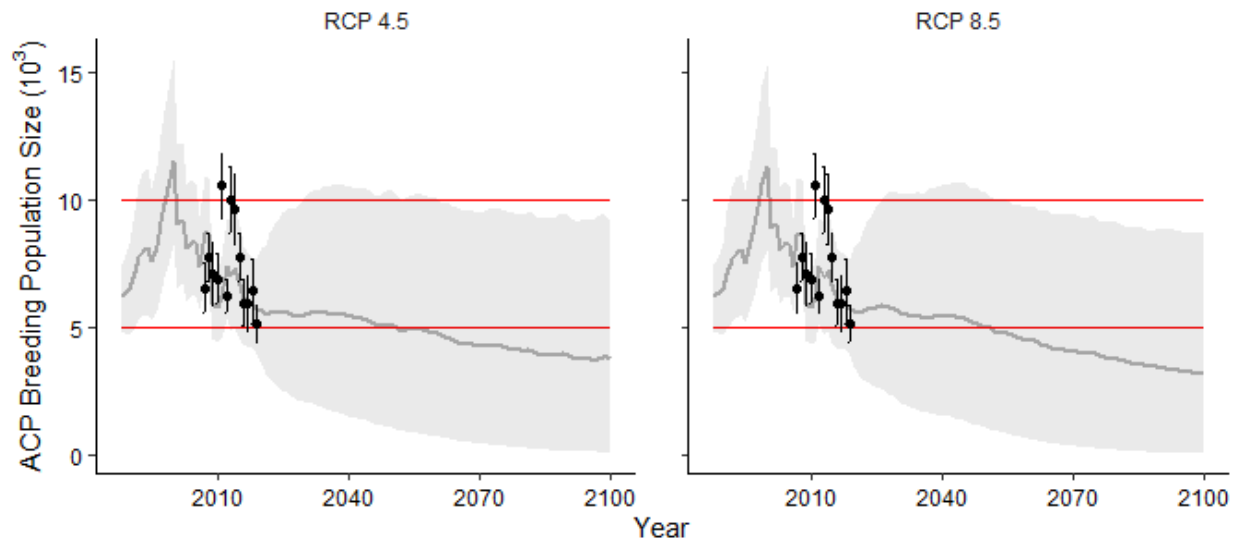


Figure 13. Estimated mean population projections for the ACP population (dark grey line) under alternative winter sea ice conditions (RCP 4.5 and RCP 8.5). 95% credibility intervals given by the light grey areas. Observed counts (± 1 SE) are shown in each panel as points and error bars. Red lines indicate population levels associated with resilience criteria outlined Section 3.3.1 ($N = 5,000$ and $10,000$).

4.1.4. Model limitations

The IPM-PVA model used to simulate future scenarios and calculate extinction probabilities is useful for informing decisions, but all models are only approximations of reality and are limited by their assumptions and the data used to inform parameters. The processes included in the model structure were identified by eider experts as important to population growth of spectacled eiders, and they were informed by data collected on the species or by formal expert elicitation when data were unavailable. The IPM-PVA model approach has several advantages; in particular, it can integrate multiple data sources such as population counts and mark-recapture data, resulting in increased precision of parameter estimates (Arnold et al. 2018 p. 272), and it incorporates model parameter uncertainties into population projections more fully (Oppel et al. 2014, Saunders et al. 2018).

Several assumptions are important to consider when evaluating the results of the IPM-PVA. These include:

- To determine the probability of lead exposure, we assumed that the spatial distribution of birds and lead deposition is proportionally constant over time on the Y-K Delta;
- Climate models that were used to project sea ice into the future introduce an unknown level of uncertainty. It can be observed in Figure 12 that recent levels of sea ice have been higher than

model projections indicated. If sea ice loss is more moderate than expected, this would dampen the deleterious effects of low sea ice projected in the model;

- The sea ice projections were based on sea ice concentration in the 1990s wintering area; however, spectacled eider distribution seems to be shifting away from that area, and therefore it is unknown if conditions there still affect spectacled eiders as they did in the past;
- Experts indicated a strong belief that winter sea ice conditions affect breeding propensity of spectacled eiders in the following breeding season, as observed in common eiders (Coulson 1984 pp. 531–532). We have few quantitative data to support this hypothesis; therefore, we developed a prior for the effect based on formal expert opinion;
- Due to the lack of data from the Arctic Russia breeding population, populations in this modeling approach were not treated as sub-populations in a metapopulation framework as they have been described in other analyses (e.g., Flint et al. 2016), likely resulting in a simplification of reality;
- The model was based on changes in lead exposure and sea ice conditions over time, but other environmental and anthropogenic factors are likely acting on spectacled eider populations such that true QE probabilities could be higher or lower than is estimated here (Section 3.1); and,
- The effect of sea ice on survival rates was evaluated based on a sample of spectacled eiders nesting on Kigigak Island, and caution should be used when extrapolating this to the entire population.

4.2 Arctic Russia Breeding Population

We do not have data with which to conduct a similar population viability analysis for the Arctic Russia breeding population. However, given the qualitative information we have about threats, distribution, and minimum population size, we can make predictions regarding the population's future condition. The available evidence suggests that lead ammunition has been used and will continue to be used in spectacled eider habitat in Russia (Section 3.1.1). We also expect that winter sea ice conditions will affect survival and breeding propensity of Arctic Russia eiders similarly to Alaska breeding populations as they are thought to intermix and winter in the same area. If we assume that other demographic parameters and threats are similar between Alaska and Russia breeding populations, which is a significant and largely untested assumption, we would expect future scenario models "RCP4.5 Constant Lead" and "RCP8.5 Constant Lead" above to best describe future conditions of the Arctic Russia population. If these assumptions are valid, the probability of QE for the Arctic Russia population would increase over time, at a similar rate as depicted in Figure 12 for the Y-K Delta population. However, as the Arctic Russia breeding population is currently larger than the Y-K Delta population (minimum population size of at least 45,000 birds relative to an abundance of 12,000 for the Y-K Delta), the probability of QE of the Russia population at 2060 and 2100 is likely lower than that estimated for the Y-K Delta population. In summary, given several assumptions, we expect the probability of QE of the

Arctic Russia population to remain low in the near-term (to 2060) but increase over time, resulting in decreasing resilience of the Arctic Russia population in the future.

4.3 Summary of Future Conditions and Viability

We expect significant changes to the marine and terrestrial habitats used by spectacled eiders in the future due to climate change (Section 3.1.10). In particular, climate models predict that sea ice extent will decrease significantly and the Bering Sea ecosystem will change due to changes to primary productivity, species distributions, and ocean acidification. However, it is uncertain how spectacled eiders will respond to such changes. We also expect some factors that negatively affect spectacled eiders to increase in the future, such as collision risk, contaminant exposure, and disturbance due to increasing vessel traffic in the Bering and Chukchi seas, and habitat loss due to industrial and community development. Others, such as lead exposure and harvest, may decrease with concerted management efforts in Alaska. However, we have no indication that such management efforts are planned or possible in Arctic Russia.

We acknowledge that several factors may influence resilience of spectacled eider populations, including those described in the above paragraph. Given the results of simulations based on plausible scenarios of sea ice conditions and lead exposure rates, factors that have been shown to affect demographic rates and of which we have data for future projections, the probability of QE remains relatively low through 2060, but increases over time for both Alaska-breeding populations in all scenarios. We do not have data to project future scenarios for the Arctic Russia population but suspect that the same factors will act on the resilience of that breeding population. Therefore, we predict that the resilience of all three populations will decrease over time, but to an unknown degree. However, because all three populations are likely to persist beyond 2100 based on model results, changes in redundancy are not expected. Both tundra and marine environments are likely to change significantly in the future. We do not have data with which to predict how genetically and/or behaviorally flexible the species will be in regards to these changes; however, we expect the species to be forced to alter its distribution, diet, and timing of key life history events (nest initiation and migration) in response. Therefore, we predict that the representation of spectacled eiders may be reduced as their habitats shift, particularly past 2060 and closer to 2100.

5. References

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Appendix A. Evaluation of Recovery Criteria in the 1996 Spectacled Eider Recovery Plan

In the 1996 Recovery Plan (Plan), the Recovery Team (Team) identified population abundance and growth rate criteria that, when met, indicate the point at which re-classifications under the ESA should be considered (i.e., endangered, threatened, or not warranted; USFWS 1996). The criteria were based upon the scientific evidence available at the time of Plan development, but also reflect levels of extinction risk and protection deemed acceptable to the Team at that time. Here, we first describe the basis for the recovery criteria developed in the Plan. We then evaluate these criteria in light of the most current information on the species, and summarize an analysis of whether the Y-K Delta and ACP populations have met the criteria.

Description of 1996 Recovery Plan Classification Criteria

In the Plan, classification criteria were developed for three breeding populations (Arctic Russia, ACP and Y-K Delta; USFWS 1996:Appendix 1, 3). For the *species* to be considered for delisting under the Plan, each breeding population must meet the criteria.

While difficult to decipher from the text in the Plan, it seems the criteria were developed using a step-wise process: first, determine the probability of extinction that corresponded to the definition of threatened and endangered populations using a population viability analysis, and then develop abundance thresholds that corresponded to those probabilities of extinction (Taylor et al. 1996, USFWS 1996).

At the time, little demographic or abundance information was available for spectacled eiders. Therefore, the Team first evaluated extinction risk at varying population growth rates based on data from common eiders (*Somateria mollissima*). Mean growth rate and variance in growth rate were used to project the population size through time using a simple model of exponential population growth (demographic data were not sufficient for a demographic model at the time; USFWS 1996:Appendix 1, 4). Simulations were conducted using uniform prior distributions of growth rate ranging from -0.01 to -0.16 and variation in growth rate ranging from 0.07 to 0.21 standard deviation. Initial population sizes ranged from 1000 to 5000, size was projected for 1000 years, and the time to quasi-extinction (QE) was recorded (USFWS 1996 p. Appendix 1, 11). The QE threshold was set at a population size of less than 250 adult spectacled eiders (equivalent to 125 pairs): the point at which the Team felt only the most radical conservation efforts could result in a full population recovery (e.g., captive breeding; USFWS 1996 p. Appendix 1, 10). Results demonstrated that a population with a growth rate of -0.05 has a 5 percent probability of reaching QE in 50 years, and the initial population size does not dramatically increase the amount of time to QE until the rate of decrease is more than 5 percent per year.

Therefore, the Team selected a population growth rate of ≤ -0.05 as the threshold for classification as endangered.

Table A1. Classification criteria for threatened spectacled eiders from the 1996 Spectacled Eider Recovery Plan (USFWS 1996, p. 36, 38). Note that criteria were developed for breeding populations, and because breeding populations are not designated Distinct Population Segments, all three populations must meet the following criteria to warrant a change in status under this Plan.

Criteria for Delisting from Threatened Status
A population will be considered for delisting from threatened status when the five factors for listing are reviewed for evidence of threats to the population and when:
(1) The population is increasing as judged by the following statistical measures:
The over protection loss exceeds the under protection loss, calculated using population trend data (based on at least 10 years (1 survey/year) of data but not exceeding a 15-year period) and where loss functions are symmetrical around $r = 0$ with a zero loss for both functions when $r = 0$; AND The minimum estimated population size⁴ is ≥ 6000 breeding pairs (12,000 individuals);
OR
(2) The minimum estimated population size is $\geq 10,000$ breeding pairs (20,000 individuals) over ≥ 3 surveys (1 survey/year, with surveys preferably being consecutive);
OR
(3) The minimum estimated population size is $\geq 25,000$ breeding pairs (50,000 individuals) in any one survey.
Criteria for Reclassifying from Threatened to Endangered
A population will be considered for reclassification from threatened to endangered when the five factors for listing are reviewed for evidence of threats and when:
(1) The population is declining by $\geq 5\%$ /year, as judged by the following statistical measures:
The under-protection loss exceeds the over-protection loss, calculated using trend data based on at least 5 years but not exceeding 15 year period, and loss functions where the loss when classifying is zero when $r < -0.05$ and the loss when not classifying is zero when $r \geq 0$;
AND
The minimum estimated population size is < 3000 breeding pairs for ≥ 1 year;
OR
The minimum estimated population size is < 2000 breeding pairs in any 1 year, unless ≥ 1 survey during the following 2 years produces an estimate of > 2000 breeding pairs.

⁴ Minimum estimated population size = the greater of two estimates, as determined by the best available data: (1) the lower limit of the 95% confidence interval (CI) of the population estimate (derived from using any subset of the data that yields the highest lower limit), including a visibility correction factor; or, (2) the actual number of birds counted during population surveys.

Based on available literature regarding minimum viable population size and using a model that did not account for potential density dependence (USFWS 1996 p. Appendix 1, 14), **the Team determined that if a population has a less than 5 percent chance of extinction in 500 years (65 spectacled eider generations) it should be considered for delisting.** In this case, extinction was defined as less than 1 pair remaining, not the QE threshold of 125 pairs as above (USFWS 1996 p. Appendix 1, 14).

Simulations were then conducted to determine the population size that corresponds with the probabilities of extinction chosen to represent each listing status. Stochastic population growth was simulated 10,000 times using an initial population size ranging from 50 to 25,600 pairs, and a mean population growth rate of $r = 0$ and standard deviation of 0.21, projected forward until less than 1 pair remain (USFWS 1996 p. Appendix 1, 14). Based on these simulations, if a population of 3000 is declining at greater than or equal to 5 percent per year, on average, then it has less than 50 years until it reaches QE. **Therefore, the following alternatives were chosen to represent endangered status: (1) a minimum population size of 3000, a population growth rate of -0.05, and the underprotection loss exceeds the overprotection loss; or, (2) a minimum population size of 2000** (USFWS 1996 p. 36).

Simulations indicated that a population size of 10,000 pairs corresponds to a less than 5 percent chance of extinction in 500 years (USFWS 1996 p. Appendix I, 15); **thus, the Team chose this as the abundance threshold for delisting (with the requirement that it be met over greater than or equal to 3 consecutive years).** The Team also provided two other alternatives for delisting from threatened status. First, in the case when population abundance data are available, the Team felt that a population abundance of 6,000 breeding pairs was sufficient (corresponding to a less than 6 percent chance of extinction in 500 years (USFWS 1996:Table I-2)), if the probability of overprotection error was greater than the probability of underprotection error (defined below). Second, estimated population abundance of greater than or equal to 25,000 breeding pairs in one survey, corresponding to the probability of extinction of 2.5 percent in 500 years, was deemed sufficient to consider delisting (USFWS 1996 p. Table I-2). For all alternatives, the lower 95 percent confidence limit of the estimated abundance, rather than the mean, was used for the threshold to ensure the population abundance was not overestimated (USFWS 1996 p. 36). Note that these abundance thresholds were determined by PVA analysis and were not based on historical size of breeding populations.

In the case of a population that is near the 6,000 breeding pair abundance threshold, the Plan required an additional analysis that ensures the risk of making a classification error is acceptably low. In species listing decisions, two possible classification decision errors could occur: (1) over protection: classifying a species to a protection category that it does not warrant; or (2) under protection: not classifying a species to a low-risk category, when it warrants greater protection (Taylor et al. 1996 p. 1078), Dunham et al. 2021).

A Bayesian decision analysis (BDA) was developed to assess the probability of classification error. BDAs use loss functions that explicitly assign a loss value to decisions (Taylor et al. 1996). For example, not classifying a population declining at 10 percent per year would be a greater loss than not classifying a population declining at 5 percent per year. Similarly, incorrectly classifying a population that was

actually increasing by 1 percent per year should be a greater loss than incorrectly classifying a population decreasing by 1 percent per year (Taylor et al. 1996 p. 1079). In the case of spectacled eiders, the risk of making an under protection error is measured as the probability of a breeding population decreasing to under 250 adults in 50 years (i.e., quasi-extinct). This definition of QE was chosen because it left only one management option: captive breeding (USFWS 1996 p. Appendix I, 10). The risk increases in size as the probability of QE nears 1.0. The Team determined that the consequences of under protection and over protection were equal when faced with a delisting decision (Dunham et al. 2021), and the over protection error must exceed under protection error to delist at the 6,000 breeding pair threshold. For decisions to re-classify as endangered, the underprotection loss must exceed the overprotection loss (USFWS 1996 p. 36).

Applicability of 1996 Recovery Criteria

Choosing decision criteria inherently involves both scientific analysis and value judgements about risk tolerance. As the SSA is intended to be a summary of the best available scientific information, we determined that the criteria were no longer appropriate scientific measures of current or future condition of the species given the current state of knowledge of spectacled eiders. Yet, the evaluation of whether the species has met the 1996 criteria may be useful in the 5-year review. We do caution, however, that the criteria developed in the 1996 plan are problematic as they are difficult to decipher, understand, and explain as written, are based on outdated information, and may not represent the risk tolerance of current decision makers, as we explain below.

In 2009, the Team determined that, based on indications that the Y-K Delta breeding population was increasing, the USFWS should evaluate whether populations have met recovery criteria. Even with the effort of multiple experts (established decision analysts, biometricians and population ecologists), a decade was required to decipher the analysis in the Plan and conduct it based on current data. In addition, the criteria are based on outdated information. Abundance criteria were based on simple models using population growth rate data from a different (but related) species. We now have 25 years of data on demographic rates and population abundance of spectacled eiders that would be more appropriate to use, and could be incorporated into contemporary analyses (e.g., the IPM-PVA described in Appendix B).

Furthermore, while the concept of using decision analysis and evaluating loss in endangered species listing problems is sound as it makes value judgements explicit and decisions more transparent and accounts for uncertainty (Cummings et al. 2018 p. 1248), the structure of this decision analysis may warrant modification, as suggested in Dunham et al. (2021). First, it is unclear if loss functions described in the criteria are consistent with current decision makers' risk tolerance, and developing loss functions based on true risk tolerance using expert elicitation may be preferable over loss functions that are symmetrical around zero (indicating that under and over protection are equally risky, Dunham et al. 2021). Another area that may warrant evaluation of decision makers' risk tolerance is basing the abundance criterion for delisting on a probability of extinction in 500 years. This represents 65 generations of spectacled eiders, and is longer than more contemporary recovery criteria and listing

decisions under the ESA, which generally base decisions on projections of 5 – 20 generations (D’Elia and McCarthy 2010 p. 754). Also, the Team chose to use the lower 95 percent confidence limit of the estimate, rather than the mean, as the abundance threshold. This implies that over estimates are much worse than under estimates, which is contradictory to the assignment of a symmetrical loss function around zero in the decision analysis (Dunham et al. 2021).

Other concerns include the choice to set loss equal to zero when population growth equals zero, as it does not accurately reflect extinction risk, because there is still a non-zero risk of extinction when population growth is greater than zero, given stochastic processes. In addition, abundance and misclassification loss criteria are somewhat redundant, as they are both based on abundance data (Dunham et al. 2021). And lastly, the abundance threshold criterion is sensitive to the posterior variance in abundance. Because the current estimates of abundance are based on a constant estimate of observation error, the posterior variance may be underestimated, biasing the results. Results could be improved by including annual estimates of observation error.

In summary, the current recovery criteria could be improved so they are clear, reflect current data, and use methods to explicitly quantify risk tolerance.

Have Alaska breeding populations met the 1996 criteria?

While the criteria have many caveats explained above, decision-makers may find it useful in the 5-year status review discussion to understand if the Alaska breeding populations have met the 1996 recovery criteria in the Plan, while acknowledging the caveats and assumptions made during the development of the criteria. Here we summarize Dunham et al.’s (2021) application of the BDA methods described in Taylor et al. (1996) and USFWS (1996) in evaluating if the Alaska breeding populations have met the delisting criteria in the Plan.

First, abundance, population growth rate, and process variation were estimated for both populations by fitting 2007-2019 detection-adjusted aerial survey count data to Bayesian state-space models (Dunham et al. 2021 Table A1). Four alternative models for the Y-K Delta population and two alternative models for the ACP population were developed to account for uncertainties in detection processes. All models indicated that the Y-K Delta population met the 6,000 breeding pair abundance threshold, and the ACP population did not (Table A1).

To determine if the populations met the classification error criterion, Dunham et al. (2021) then generated the under protection loss function by projecting abundance for 50 years over all possible values of population growth to determine the probability of extinction. The probability of committing a classification error is calculated as the sum of the expected loss when population growth rate (r) is less than zero (under protection) or r is greater than or equal to zero (over protection), represented by the area under the loss curve (Figure A1), multiplied by the probability of that value of r from the posterior distribution of mean population growth. The expected loss for each error type is a single value generated after integrating out all possible values of population growth rate (Dunham et al. 2021). The optimal decision minimizes under protection loss in favor of maximizing over protection loss.

For the ACP population, over protection loss is less than under protection loss using both models, indicating that the population has not met the criterion (Table A1, Figure A1). For the Y-K Delta population, the estimated over protection loss is greater than under protection loss using all four models (Table A1); therefore, the population meets the misclassification error criterion in the Plan. Given the preferred model (YKD4), the risk of committing an over protection error for the Y-K Delta population was 6.18 times that of the risk of committing an under protection error (0.067 and 0.011, respectively; Dunham et al. 2021, Figure A1). However, note that the risk of classification errors in both cases were close to zero, because the mean population growth rate was near zero and estimated with precision as indicated by the narrow posterior distribution, and because the loss curves were chosen to be symmetrical around zero. These factors indicate some uncertainty, and should be considered while making decisions about population resilience and species classification.

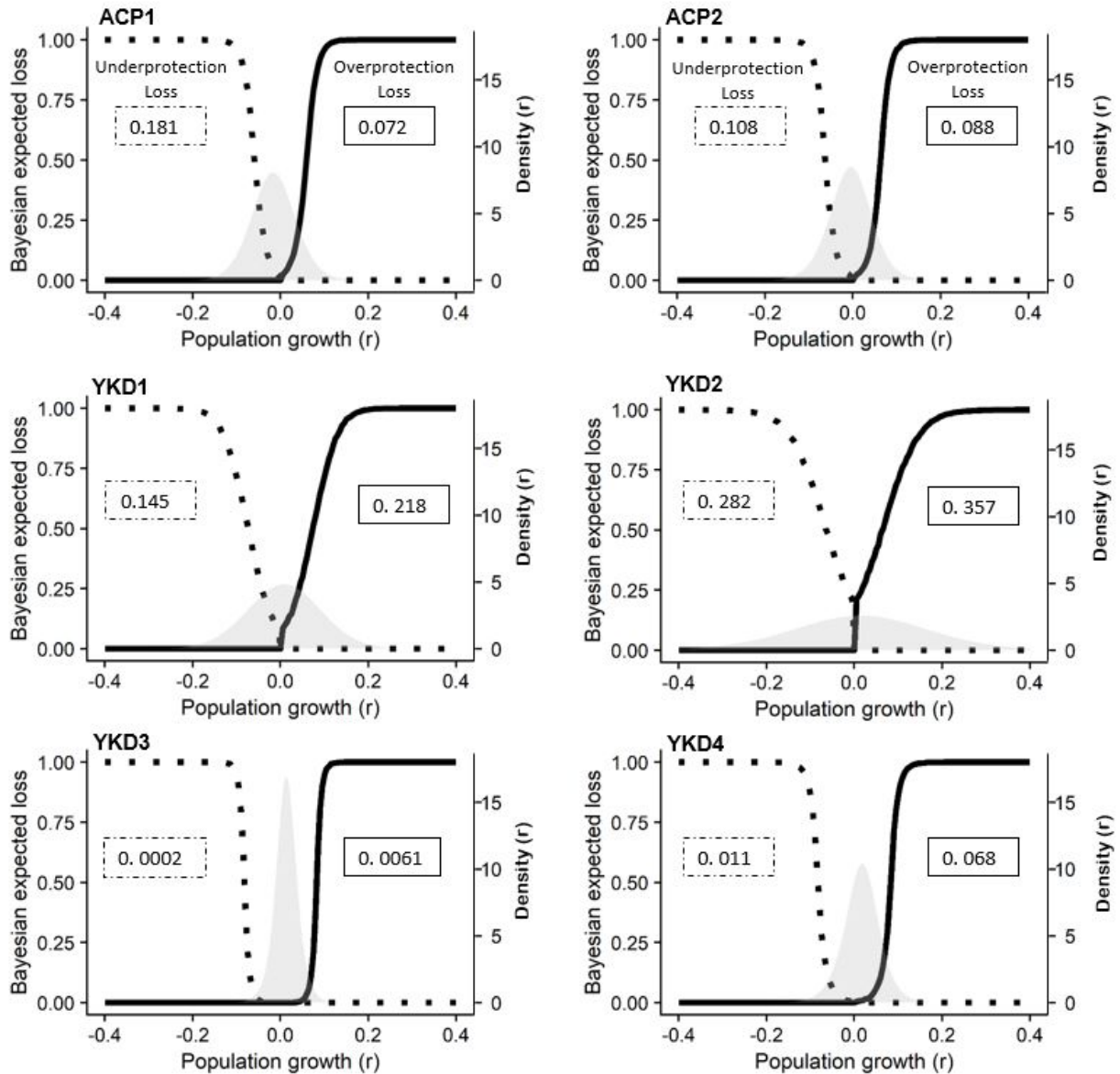


Figure A1. From Dunham et al. (2021): “Loss functions and posterior distributions of population growth rate (r) generated from state-space models of abundance for spectacled eiders breeding on the Arctic Coastal Plain (ACP) and Yukon Kuskokwim Delta (YKD). Loss functions were generated using the probability of QE ($n < 250$) given population size, growth rate, and process variation. The dotted line represents the under protection loss function (i.e. loss if decision were to delist given negative population growth) and the solid line is the over protection loss function (i.e. loss if the decision were to maintain status given positive population growth). Gray distributions show the posterior density of population growth rate (r) estimated by a Bayesian state-space model for the time series from 2007 to 2019. As part of the recovery criteria, spectacled eiders will be considered for delisting if overprotection (value in solid line box) is greater than underprotection (value in dashed line box). Greater overprotection error indicates that we are more likely to provide too much protection to the species than we are to provide too little protection to the species. The ACP population has not met this criteria because

underprotection error is greater than overprotection error. Each alternative model for the YKD population results in overprotection being greater than underprotection.”

Table A2. From Dunham et al. (2021): “Posterior estimates of population metrics and misclassification error for both Alaskan breeding populations of spectacled eiders (*Somateria fischeri*). Consideration for reclassification from threatened to recovered requires that both populations must reach or exceed the abundance threshold ($N \geq 12,000$ breeding birds), and overprotection loss must be greater than underprotection loss. Abundance estimates and misclassification error rates for the ACP population do not support the decision to delist. Alternatively, all four models for the YKD population support delisting based on population metrics meeting the reclassification criteria in the species recovery plan.”

	ACP1 ^a	ACP2 ^a	YKD1 ^b	YKD2 ^b	YKD3 ^b	YKD4 ^b
Abundance						
Posterior mean	5355	6401	15054	15047	15388	16113
Posterior SD	629	1510	1104	1118	908	2249
95% CRI	4106 - 6589	3766 - 9750	12903 - 17212	12863 - 17253	13595 - 17175	12313 - 21352
Pop. growth ‘r’						
Posterior mean	-0.016	-0.005	0.009	0.013	0.013	0.016
Posterior SD	0.043	0.043	0.068	0.137	0.023	0.037
95% CRI	-0.103 – 0.072	-0.092 – 0.082	-0.124 – 0.0143	-0.263 – 0.287	-0.035 – 0.062	-0.065 – 0.091
Process variation						
Posterior mean	0.158	0.142	0.323	0.479	0.073	0.123
Posterior SD	0.061	0.064	0.064	0.137	0.038	0.073
95% CRI	0.057 – 0.293	0.039 – 0.288	0.219 – 0.468	0.284 – 0.814	0.017 – 0.161	0.026 – 0.305
Loss						
Underprotection	0.181	0.108	0.145	0.282	0.0002	0.011
Overprotection	0.072	0.088	0.218	0.357	0.0061	0.068

^a ACP metrics refer to the Arctic Coastal Plain breeding population.

^b YKD metrics refer to the Yukon-Kuskokwim Delta breeding population.

Appendix B. An Integrated Population Model-Population Viability Analysis for Spectacled Eider

An Integrated Population Model - Population Viability Analysis for Spectacled Eiders (*Somateria fischeri*)

Catherine Bradley and Erik Osnas

20 November, 2020

Introduction

Here we present a Bayesian population viability analysis (PVA) derived from an integrated population model (IPM) to evaluate the resilience of the Yukon-Kuskokwim Delta (YKD) and Arctic Coastal Plain (ACP) populations of spectacled eiders (*Somateria fischeri*). Benefits to building a PVA from an IPM include utilizing both population abundance data and demographic data to more completely describe population dynamics and incorporating model parameter uncertainties into population projections more fully (Oppel et al. 2014; Saunders et al. 2018). For each population, the model generates a distribution of population sizes over time under various scenarios pertaining to lead exposure rates and winter sea ice conditions, factors that previous work suggest are significant to the species. From distribution of population size, the probability of quasi-extinction, as defined by the Spectacled Eider Recovery Plan (USFWS 1996), can be estimated. These results will be incorporated into the Species Status Assessment (Smith et al. 2018) for spectacled eiders and used to support a discussion of the populations' current status and resilience to future environmental change.

Life History Model

The life history model (Figure 1) is derived from Flint et al. (2016) and Dunham (2019) and describes a pulsed-birth system monitoring pre-breeding females. It tracks four stages: one year old (n_1), two year old non-breeders (n_{2nB}), two year old breeders (n_{2B}), and three years and older (n_{3+}). Survival parameters are represented by ϕ . With the exception of ϕ_0 , survival is annual; ϕ_0 describes survival from day 30 to 1 year old. ω represents a correction factor for the estimated survival probabilities to account for permanent emigration from the mark-recapture study area and into the larger population (local dispersal). Because natal site dispersal is thought to be higher than breeding site dispersal, ω_J and ω_A are estimated separately for sexually immature and sexually mature classes, respectively. The fecundity parameter, f , is composed of breeding propensity, clutch size (female offspring assuming a 0.5 sex ratio), nesting success, and duckling success (survival to day 30). The probability that an n_1 female becomes a breeding n_2 female, or maturation probability, is given by ϕ_1 and three years and older females are assumed to be sexually mature ($\phi_{3+} = 1$).

Winter sea ice conditions in the overwintering area may impact the survival of juveniles (ϕ_0) differently from all other age classes (ϕ_1 , ϕ_2 , and ϕ_{3+}). Only individuals on the breeding grounds (n_{2B} and n_{3+}) are exposed to the deleterious effects of lead through ϕ_a . The fecundity in year t is impacted by the winter sea ice conditions in the previous winter through the breeding propensity of mature females.

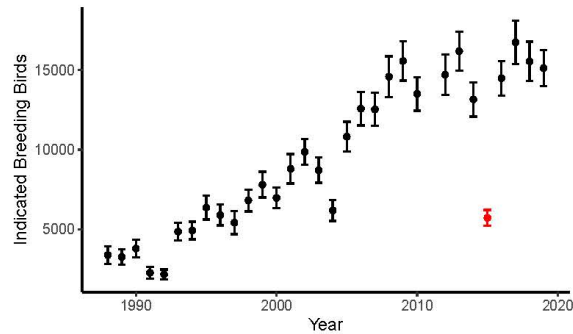


Figure 2. Estimated number of indicated breeding spectacled eiders on the Yukon Kuskokwim Delta, 1988-2019. Error bars represent ± 1 SE. The low estimate in 2015 is represented in red.

Count estimates for the ACP were based on aerial survey observations from 2007 to 2019 of indicated breeding birds (Wilson et al. 2018) and adjusted for single and pair-specific detection probabilities estimated from a double observer method (Wilson et al., *pers. comm.*). On the ACP survey, estimates come from two observers, the pilot and a right-seat observer, that make up an observer team. The estimated detection probabilities were estimated from two observers over three years (2015-2017) and the average detection probabilities were applied to all 7 unique teams from 2007 to 2019 to estimate indicated breeding spectacled eiders (Osnas, *unpublished code*, Figure 3). For the model using the ACP data, we used observer team as a covariate.

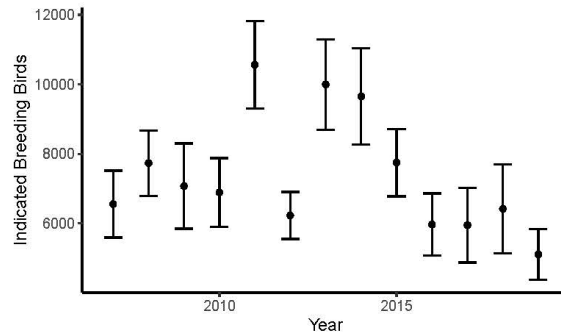


Figure 3. Estimated number of indicated breeding spectacled eiders on the Arctic Coastal Plain, 2007 - 2019. Error bars represent ± 1 SE.

The IPM uses data obtained from Kigigak Island to estimate survival probabilities and to assert a prior for average annual fecundity, less breeding propensity. The Kigigak Island time series is longer than that available from Kashunuk (YKD) or Utqiagvik (ACP) and lead exposure on Kigigak Island is much more consistent with that estimated for the entire YKD compared to Kashunuk, based on Flint et al. (2016). Adjustments for parameters from the Kigigak Island study area to the wider YKD and the ACP were made through the inclusion of parameters such as local dispersal (ω_i) and expert elicitation.

Annual estimates of nest survival and clutch size on Kigigak Island are available from 1992-2004 (Flint et al. 2016) and 1992-2015 (Gabrielson and Spragens 2013; Moore and Sowl 2015; Figure 4). Those efforts

utilized different models for daily survival rates; however, nest success estimates from Flint et al. (2016) and Gabrielson and Spragens (2013) are strongly correlated ($\rho = 0.97$). The available estimate of duckling survival (0.67) is from a number of observations over two years on Kigigak Island (Flint et al. 2016) and is considered high, likely due to the small sample size. Early iterations of the IPM directly incorporated these data sources. To simplify the model structure, rather than using this data explicitly in the IPM, it was used to develop an informed prior in the final model for the composite variable μ_f . This variable scales fecundity to fit the observed count and survival data.

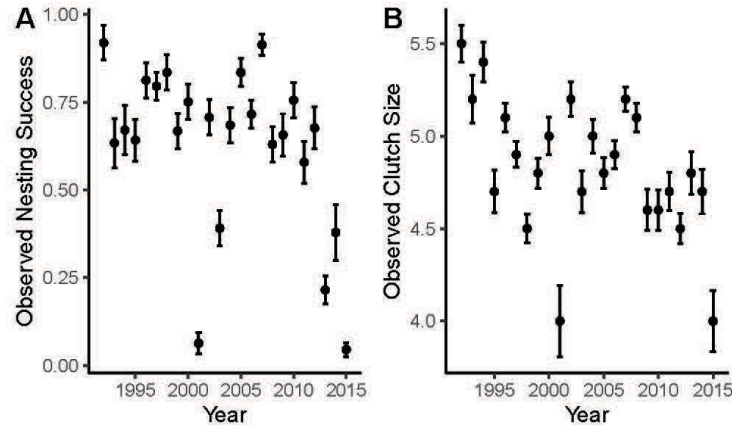


Figure 4. Annual estimates of spectacled eider nest survival (A) and clutch size (B) on Kigigak Island, 1992-2015 (from Gabrielson and Spragens 2013; Moore and Sowl 2015). Error bars represent ± 1 SE.

Mark-recapture efforts were conducted on Kigigak Island from 1992-2015 and the resultant capture history data is available as part of the [Dryad repository](#) from Christie et al. (2018). Females were marked as ducklings or as breeding adults. All birds initially marked as adults were assumed to be three years old or greater. One year old and two year old non-breeders are unobserved states in the survival analysis.

Environmental conditions hypothesized to have potential impacts to spectacled eider population resilience include lead exposure on the breeding grounds (Flint et al. 2016) and sea ice extent in the primary overwintering location south of Saint Lawrence Island (Christie et al. 2018). Lead exposure is thought to be of particular importance to the YKD population, due to the relatively high human population density and hunting pressure exerted there. Both are expected to decline going forward, with positive effects of the decline in lead on eider survival and uncertain effects of declining sea ice. The rate of decline for both factors is unknown. For lead, uncertainty is due to unknown future deposition rates on the breeding grounds and uncertainty in the rate that lead becomes unavailable to eiders. For sea ice extent, uncertainty is due to which future climate scenario will be realized and the annual stochastic variations in sea ice extent. These are included in the model structure and proposed variants in the future status of these variables are explored in the PVA as alternative scenarios.

The probability of lead exposure on the YKD, defined as blood Pb > 0.2 ppm, in the breeding female population at time of hatch from 1994-1999 was estimated in Flint et al. (2016) as 0.118. The model linked observed lead exposure rates at point locations to the distance to villages by land or waterways and extrapolated the observe relationship spatially across the YKD. Using this value as a baseline, θ_0 , we assume that the spatial distribution of breeding birds and lead deposition is proportionately constant throughout time. In alternative scenarios considered for the YKD population, three future lead exposure curves are modeled by varying the year at which new lead deposition on the YKD ceases, presumably through management action (2008, 2026, or remaining constant), and assuming a 10 year decay rate in the availability

of lead, such that $\theta_t = (\theta_0)^{dt}$ where θ is the exposure rate and d is the decay rate (Flint et al. 2016; Figure 5). A fourth scenario, to model current conditions, proposes declining lead levels from 2008-2019 and constant levels thereafter for future projections. Priors are declared for θ_0 and d to reflect the uncertainty in both variables. By asserting that all difference in adult survival between Kigigak Island and Kashunuk was due to differential lead exposure rates, Flint et al. (2016) estimated adult survival with and without exposure and found that survival with exposure was reduced by approximately one-half. A prior is declared for this effect, κ , accounting for sampling variance.

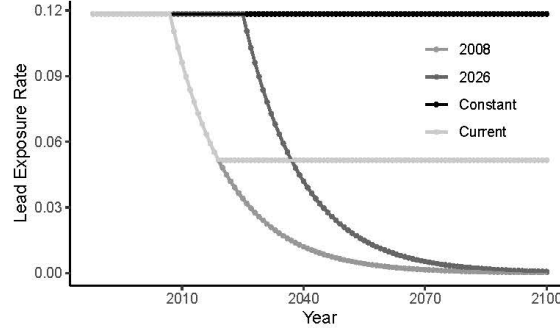


Figure 5. Population-level lead exposure rates asserted in the spectacled eider PVA assuming a 10-year half life of lead availability in the habitat and varying times at which new lead deposition in the environment ceases.

The number of observed extreme sea ice days (>95% coverage) in the overwintering area were obtained through the Christie et al. (2018) Dryad repository (link given above) and updated through winter 2017-2018 (David Douglas, USGS, *pers. comm.*). As in Christie et al. (2018), future estimates of extreme ice days relative to RCP4.5 and RCP8.5 climate scenarios are considered in the PVAs for the YKD and ACP populations. However, the model averaged predictions of extreme ice days under either climate scenario exhibit much less interannual variability (RCP4.5: $\sigma^2 = 141.03$ and RCP8.5: $\sigma^2 = 161.42$) than the observed record ($\sigma^2 = 725.76$; 1980-2018). To simulate future ice conditions with interannual variability more consistent with the observed record, we fit the observed data to the climate scenario estimates using a negative binomial regression, which models overdispersed error scaling with the mean. The observed data is most consistent with draws from a negative binomial distribution with mean equal to the model estimate and dispersion parameter, ζ , equal to 3.75 (RCP4.5) and 3.5 (RCP8.5). We use this distributional assumption to draw estimates of future extreme sea ice conditions (Figure 6) based on the model averaged predictions. The covariate is standardized before use in the linear models for survival and breeding propensity.

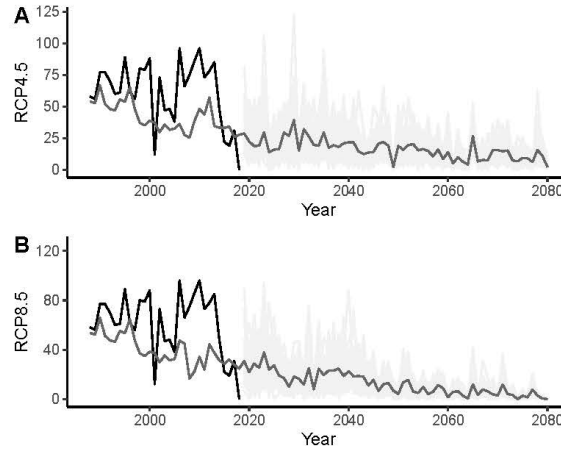


Figure 6. RCP4.5 (A) and RCP8.5 (B) extreme sea ice day data generation. Observed (black) and RCP model-averaged estimate (dark grey) of extreme sea ice days in the overwintering area. Light grey lines represent random draws from the RCP4.5 (A) and RCP8.5 (B) model averaged estimates using a negative binomial distribution with dispersion parameter 3.75 and 3.5, respectively.

To model current conditions, generation of future sea ice data consistent with the observed record, 1980-2018, was conducted in a similar manner, using draws from a negative binomial distribution with mean equal to the observed average ($\mu = 54.03$) and dispersion parameter 4.3, consistent with the observed variance ($\sigma^2 = 725.76$) where

$$\zeta = \frac{\mu^2}{\sigma^2 - \mu}.$$

Expert Elicitation

Several model parameters required expert elicitation, as there is no data currently available to inform their estimates. We followed the IDEA protocol outlined in Hemmings et al. (2018) to gather the necessary quantitative data. Seven experts with a collective breadth of knowledge in sea duck ecology, particularly eiders, and in demographic modeling began the exercise and six ultimately completed all steps. We developed ten questions concerning survival differences between the YKD and ACP populations, juvenile and adult local dispersal, winter sea ice effects on breeding propensity, the carrying capacity of the YKD and ACP populations, two year old maturation probability, and count survey interannual variability within observers. For each question, respondents were asked to give a minimum plausible value, maximum plausible value, most likely plausible value, and their confidence in the interval containing the true value. These responses were aggregated, summarized, and presented to the experts for a group discussion focused on ensuring consistent interpretation of the questions posed and allowing experts to communicate information or research that informed individual responses with the rest of the group. No attempt at consensus was made. Following this discussion, experts were given a second opportunity to respond to the questions, if they felt that revision was warranted. This second round of responses were again aggregated, summarized, and, ultimately, used to develop informed priors for model parameters.

Details on the questions, responses, and translation of responses into informed priors is available in an accompanying appendix.

Model Structure

The IPM is comprised of four parts or submodels: (1) a latent population projection or process model for the female eider population based on the Lefkovitch transition matrix described above, (2) a fecundity model informed by productivity studies on Kigigak Island, (3) survival estimates from a multi-state mark-recapture model based on banded adults and ducklings on Kigigak Island, and (4) an observation model that relates population projections to observed count data. In addition, all parameters in each submodel have prior distributions which are an important part of the model. For those submodels that have data, there is a likelihood of the data, given the parameter values. For all parameters, we used informed priors that were based on other published work using this same data, our own estimates obtained outside of the IPM, expert elicitation, or biological plausibility on the parameter scale.

Yukon-Kuskokwim Delta Population

Process Submodel

The process model is a simple age- and stage- (breeding status) structured matrix projection model for the female population based on the Lefkovitch transition matrix, $\mathbf{A}$ above. The female population in year t is

$$\mathbf{N}_{t+1} = \mathbf{A}_t \mathbf{N}_t,$$

where $\mathbf{A}_t$ has year-specific elements determined from the survival and fecundity models below and $\mathbf{N}_t$ is a vector of female population size for each age and stage (Figure 1),

$$\mathbf{N}_t = \begin{bmatrix} n_{1,t} \\ n_{2nB,t} \\ n_{2B,t} \\ n_{3+,t} \end{bmatrix}.$$

The female breeding population is then defined as $n_{B,t} = n_{2B,t} + n_{3+,t}$. The process model is used to determine the likelihood for the count model, describe below.

Fecundity Submodel

The annual per capita rate of fecundity is modeled relative to the estimated survival of immature females adjusted for juvenile dispersal from Kigigak Island to the wider YKD (ϕ_0^*) and as a function of the female breeding population size, $n_{B,t}$, and breeding propensity, BP_t , at time t

$$\log(f_t) = \log(\mu_f) + \log(\phi_{0,t}^*) - DD_f * n_{B,t} + \log(BP_t)$$

where μ_f is the average number of female offspring surviving to day 30 after hatch with a prior given on the log scale

$$\log(\mu_f) \sim \text{Normal}(-0.47, 0.1).$$

The prior for $\log(\mu_f)$ is consistent with an average nest success 0.65, female clutch size 2.4, and duckling survival 0.4 as observed in the Kigigak Island fecundity data time series.

The estimated survival of immature females ($\phi_{0,t}^*$), accounting for local dispersal, is

$$\phi_{0,t} + (1 - \phi_{0,t}) * \omega_J$$

where $\phi_{0,t}$ is estimated from the mark-recapture data and the prior for natal site dispersal, ω_J , was obtained from expert elicitation as

$$\omega_J \sim \text{Beta}(1, 9).$$

We assume a log-linear negative density-dependence effect on fecundity with prior

$$\log(\beta_{DDf}^{-1}) \sim \text{Normal}(7.95, 3.31)$$

informed from expert elicitation of the carrying capacity on the breeding grounds and using the deterministic Leftovich projection model to find the prior distribution for density-dependence consistent with the elicited carry capacity (see appendix). Specifically, the prior is consistent with a carrying capacity of approximately 20,000 breeding females at the prior mean and gives a prior distribution from 8,500 to 37,500 at the 10th and 90th percentiles, respectively.

We modeled density-dependence as a function of the breeding population although, mechanistically, density-dependence is expected to be exerted primarily in the overwintering area as a function of the total global population of spectacled eiders. There is currently no reliable data on the world population size or its trend. We initially attempted to model this unobserved population as a latent state in this IPM to scale density-dependence to the total (unobserved) population size. However, without data to suggest the trend in global population counts differed from the breeding population or strong priors that imply different demographic rates, the total population changed proportionally with the observed population. Therefore, modeling density-dependence as a function of the breeding population size is a simplification that assumes the global and local populations are synchronous so that each increase to a world equilibrium population size (carrying capacity) or, alternatively, the world population decreases and the YKD population increases as it is released from density-dependent regulation. Without data on the world population these two patterns cannot be distinguished, so modeling density-dependence based on breeding population size represent a phenomenological way to incorporate population size regulation and not a mechanistic belief that regulation occur on the breeding area.

Breeding propensity of mature females in year t is modeled on the logit scale as a function an intercept and extreme winter sea ice days at $t - 1$, including a squared term of sea ice to reflect expert opinion on the functional shape of the effect on the real scale, and a random year effect

$$\text{logit}(BP_t) = BP_0 + \beta_{BP,ice} * ice_{t-1} + \beta_{BP,ice^2} * ice_{t-1}^2 + \epsilon_{BP,t}$$

where

$$BP_0 \sim \text{Normal}(0.472, 0.22)T(-0.77, 3.22)$$

$$\beta_{BP,ice} \sim \text{Normal}(2.83, 0.11)T(-1, 3)$$

$$\beta_{BP,ice^2} \sim \text{Normal}(-1.17, 0.69)T(-1.21, 0.33).$$

Random year effects are modeled as

$$\epsilon_{BP,t} \sim \text{Normal}(0, \sigma_{BP})$$

$$\sigma_{BP} \sim \text{Uniform}(0, 1).$$

This parameter set is consistent with a breeding propensity around 0.90 (80% CI: 0.75-0.92) when there are 60 winter extreme sea ice days, declining to 0.78 (80% CI: 0.50-1) when there are around 35 winter extreme sea ice days and to 0.72 (80% CI: 0.46-0.93) when there are 85 days of extreme sea ice. There is no direct likelihood for the fecundity model. All information is expressed through the priors or through the likelihood in the survival model for $\phi_{0,t}$.

Survival Submodel

The mark-recapture data is summarized in an m-array format in the IPM and modeled using a multi-state survival framework. Ducklings, two year old breeding, and three year or older females are directly observed; one and two year old non-breeding females are unobserved states. Informed priors are asserted for μ_{pha} and μ_{phio} based on a multi-state survival model fit outside of the IPM.

The recapture probability, p_t is a function of nesting success in year t , under the assumption that greater nesting success increases the likelihood of observing a previously marked female

$$\text{logit}(p_t) = \text{logit}(\mu_p) + \beta_{ns} * nest_t + \epsilon_{p,t}$$

where

$$\begin{aligned}\mu_p &\sim \text{Beta}(1,1) \\ n_s &\sim \text{Normal}(0,0.1)\end{aligned}$$

and random year effects

$$\begin{aligned}\epsilon_{p,t} &\sim \text{Normal}(0, \sigma_p) \\ \sigma_p &\sim \text{Uniform}(0,1).\end{aligned}$$

To account for sampling error in the annual estimate of nesting success, the covariate in the model for detection probability was drawn from beta distributions using the mean and variance estimated in each year.

We modeled the impacts of the number of extreme sea ice days in the overwintering area, both linear and quadratic effects (Christie et al. 2018), change in lead exposure rate, and a linear negative density-dependence effect of the female breeding population to the survival of two year old breeding and three year and older females (ϕ_a) on the logit scale

$$\text{logit}(\phi_{a,t}) = \text{logit}(\mu_{\phi_a}^*) + \phi_{a,ice} * ice_t + \phi_{a,ice^2} * ice_t^2 - DD\phi * n_{B,t} + \epsilon_{\phi_a,t}$$

where $\mu_{\phi_a}^*$ is a mixture of the survival of breeding females exposed and unexposed to deleterious levels of lead

$$\begin{aligned}\mu_{\phi_a}^* &= [\mu_{\phi_a} * (1 - \theta_t)] + [\mu_{\phi_a} * \kappa * \theta_t] \\ \mu_{\phi_a} &\sim \text{Beta}(90,10).\end{aligned}$$

Input values for lead exposure rate, θ_t , and the relative impact to survival from lead exposure, κ , were generated as described in the *Data Sources and Use* section. The prior for the baseline exposure rate was

$$\theta_0 \sim \text{Beta}(6,45),$$

and the prior for lead exposure decay rate was a truncated normal with lower bound of one

$$d \sim \text{Normal}(10, 3.2)T(1,).$$

The impact of lead on survival was determined by

$$\kappa \sim \text{Beta}(20,20).$$

The priors for the ice covariates are

$$\begin{aligned}\phi_{a,ice} &\sim \text{Normal}(0,0.1) \\ \phi_{a,ice^2} &\sim \text{Normal}(0,0.1).\end{aligned}$$

The density-dependence is modeled as

$$\log(\text{beta}_{DD\phi}^{-1}) \sim \text{Normal}(7.95, 3.31)$$

in the same manner as described above. Annual random effects are

$$\begin{aligned}\epsilon_{\phi_a,t} &\sim \text{Normal}(0, \sigma_{\phi_a}) \\ \sigma_{\phi_a} &\sim \text{Uniform}(0,1).\end{aligned}$$

We assume that the survival of two year old non-breeders, ϕ_2 , and one year old females, ϕ_1 , are subject to the same effects of ice and density-dependence, but with no exposure to lead such that

$$\text{logit}(\phi_{2,t}) = \text{logit}(\mu_{\phi_a}) + \phi_{a,ice} * ice_t + \phi_{a,ice^2} * ice_t^2 - DD\phi * n_{B,t} + \epsilon_{\phi_a,t}$$

(note the lack of mixture term in μ_{ϕ_a}) and

$$\phi_{1,t} = \phi_{2,t}.$$

ϕ_0 is modeled as a function of ice and random year effects

$$\text{logit}(\phi_{0,t}) = \text{logit}(\mu_{\phi_0}) + \phi_{0,ice} * ice_t + \phi_{0,ice^2} * ice_t^2 + \epsilon_{\phi_0,t}$$

where

$$\begin{aligned}\mu_{\phi_0} &\sim \text{Beta}(15, 45) \\ \phi_{0,ice} &\sim \text{Normal}(0, 0.1) \\ \phi_{0,ice^2} &\sim \text{Normal}(0, 0.1).\end{aligned}$$

Random year effects are modeled as

$$\begin{aligned}\epsilon_{\phi_0,t} &\sim \text{Normal}(0, \sigma_{\phi_0}) \\ \sigma_{\phi_0} &\sim \text{Uniform}(0, 1).\end{aligned}$$

Priors for the maturation probability of 2 year old females, ω_a , and breeding site dispersal, ω_a were informed by expert elicitation and are given by

$$\sim \text{Gamma}^{-1}(7.1, 1.95)T(1,)$$

and

$$\omega_a \sim \text{Beta}(1, 35).$$

The likelihood for the mark-recapture model is

$$\mathbf{m}_t \sim \text{Multinomial}(\mathbf{p}_t, R_t),$$

where $\mathbf{m}_t$ is the t -th row of a 'multistate M-array,' a matrix with a row for each initial release occasion and state, a column for each combination of recapture occasion and state, and a column of total releases never recaptured for each release occasion and state. Then, each element of the matrix is the number of recaptures in each state, including those never observed after the initial release. The vector $\mathbf{p}_t$ is a vector of probabilities of being observed at each recapture occasion in each state, and these probabilities are determined by the parameters and functions above. R_t is the total number of individuals marked and released during capture occasion t . The total likelihood is the product over all rows of the M-array.

Observation Submodel

Observed population estimates from the aerial survey on the YKD are derived assuming constant detection across years and observers (Lewis et al. 2019). Therefore, we incorporated observer- and year-specific effect in the observation model. Observed population estimates were modeled as an expectation of the observed number of females, multiplied by two to account for both sexes,

$$\hat{Y}_t \sim \text{Normal}(2\mu_t, \widehat{\sigma_{y,t}})$$

where $\hat{Y}_t$ is the point estimate from the YKD survey and $\widehat{\sigma_{y,t}}$ is the standard error of the estimate. This is the likelihood of the count model.

The expected number of females was a function of the breeding population from the projection model, random effects due to observer, and an unobserved annual random effect due to year-specific detection variation

$$\log(\mu_t) = \log(n_{B,t}) - o_{i(t)} - d_t$$

with $n_{B,t}$ the number of breeding females on the YKD, $o_{i(t)}$ an average log detection effect due to observer i , and d_t a log detection effect for year t that essentially acts as a source of observation variance not accounted for in the estimated $\widehat{\sigma_{y,t}}$. Priors are

$$o_{i(t)} \sim \text{Normal}(0, \sigma_o)$$

$$\begin{aligned}\sigma_o &\sim \text{Gamma}(1, 10) \\ d_t &\sim \text{Normal}(0, \sigma_d)T(-0.69, 0.69) \\ \sigma_d &\sim \text{Gamma}(2, 13).\end{aligned}$$

The parameters in the gamma prior for σ_d were informed by expert elicitation on interannual detection variability within observers.

Finally, we specified a prior for the initial population size based on the observed estimates in 1988 and analysis of the stable age distribution of the transition matrix, above, using YKD mean parameter values:

$$\begin{aligned}n_{1,1} &\sim \text{Uniform}(100, 300), \\ n_{2B,1} &\sim \text{Uniform}(100, 350), \\ n_{2B,1} &\sim \text{Uniform}(100, 250),\end{aligned}$$

and

$$n_{3+,1} \sim \text{Uniform}(200, 2850).$$

Arctic Coastal Plain Population

We used the same model for the ACP population with the following modifications. For the count model we used the ACP population count data and standard errors as described above in the section *Data sources and Use* and treated count data from 1988 to 2006 as missing. For the unobserved annual detection variation, d_t , we used the prior

$$\begin{aligned}d_t &\sim \text{Normal}(0, \sigma_d)T(-0.28, 0.28) \\ \sigma_d &\sim \text{Gamma}(1, 13).\end{aligned}$$

This gives variation in detection consistent with the expert elicitation. The truncation was to insure that deviations in detection did not exceed 0.25 on the probability scale (assuming mean detection at 0.75) or the multiplicative effect on detection was >0.75 and <1.33 . Note that the mean σ_d is much lower than for the YKD, reflecting a belief that detection varies less between years within the same observer for this survey.

For the survival and fecundity models, we used the same structure and data as for the YKD, except that we removed all lead parameters from the survival model (essentially equivalent to setting $\theta_0 = 0$) and specified ACP-specific priors for the survival intercepts ($\mu_{\phi_0}, \mu_{\phi_a}$) and density-dependence parameters ($DD\phi, DDf$). This was done to reproduce the likelihood for the YKD mark-recapture model and reflect the belief that survival and fecundity parameters must be correlated because of the shared over-winter distribution of the YKD and ACP populations. We used the following ACP-specific priors based on the expert elicitation:

$$DDf, DD\phi \sim \text{Gamma}(3, 3300),$$

This prior is consistent with a carrying capacity of approximately 12,000 breeding females at the prior mean and gives a prior distribution from 6,900 to 34,000 at the 10th and 90th percentiles, respectively.

$$\mu_{\phi_0} \sim \text{Beta}(2.3, 4),$$

and

$$\mu_{\phi_a} \sim \text{Beta}(26, 3).$$

Finally, we use priors for the initial (1988) population size that, when the breeding population was summed, approximated those reported in Amundson et al. (2019) after assuming a detection rate of 0.75:

$$\begin{aligned}n_{1B,1} &\sim \text{Uniform}(250, 500), \\ n_{2B,1} &\sim \text{Uniform}(250, 500), \\ n_{3B,1} &\sim \text{Uniform}(500, 1000),\end{aligned}$$

and

$$n_{4B,1} \sim \text{Uniform}(1500, 3000).$$

Model Implementation and Diagnostics

Models were run using JAGS v.4.3.0 (Plummer 2017) in the R package jagsUI v.1.5.1 (Kellner 2019) with R v.4.0.2 (R Core Team 2020) for 30,000 iterations on 4 chains with a 20,000 iteration burn-in. We monitored derived parameters that were directly needed for the population viability analysis (breeding population size), influential to the results (survival and associated parameters), or those where achieving convergence was challenging (breeding population size and some of the variance parameters). For the models presented, these parameters all converged within 20,000 iterations ($R_{hat} < 1.1$) and were sampled until the effective sample size of the breeding population estimate in year 2100 was greater than 300. Models were developed using the RCP4.5 ice projections and, for the YKD population, assuming lead levels began declining in 2008.

Seven models were run for the YKD population: all combinations of proposed lead exposure rates (Lead2008, Lead2026, Constant) and ice projections (RCP4.5 and RCP8.5), and one model of current conditions. Similarly, three models were run for the ACP population, varying the ice projections (RCP4.5, RCP8.5, and current conditions).

Quasi-extinction and Time Horizons

As defined in the Spectacled Eider Recovery Plan (USFWS 1996), 250 adults in a breeding population constitutes quasi-extinction. *A priori*, the probability of quasi-extinction within the next 40 years (approximately 5 generations) and between 40 and 80 years in the future (approximately 5-10 generations) were identified as the derived quantities of primary interest from the PVA (Kate Martin, USFWS, *pers. comm.*).

For each of the alternative scenarios explored in the PVA, 40,000 samples were taken from the posterior distribution of the breeding population size projections ($2n_{B,t}$). To quantify the probability of quasi-extinction within each time frame, the year when the population projection first reaches QE, if ever, in each sample was retained and the total QE events observed in each time frame (< 2060 or $2060-2100$) is expressed as a percentage.

Results

The PVA projects population abundances to 2100, or 80 years, generating thousands of samples of the future population trajectory. For example, figure 7 depicts simulated population trajectories for the YKD population, assuming lead decreases beginning in 2008 and ice conditions are consistent with RCP4.5 model projections. There is wide variation in the predicted trajectories, as would be expected in a stochastic population projection. In figures 9 and 10, only the mean of population trajectories is shown in order to emphasize the average effect of each scenario; however, the variation across trajectories is similar to that shown in figure 7. Importantly, no realized population trajectory will mimic the mean trajectory. Of the 55 simulated draws shown here, five fall below the QE level of 250 breeding individuals before 2100.

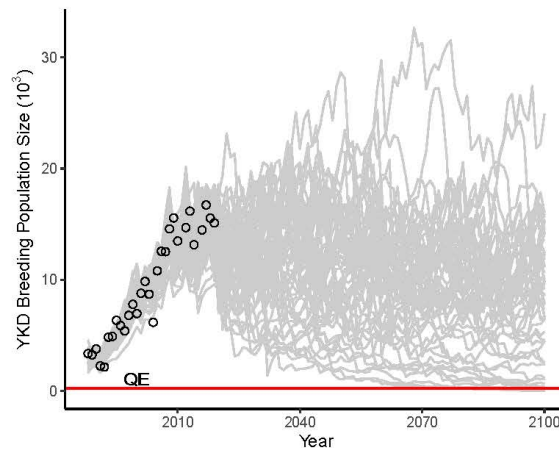


Figure 7. Breeding population size for the YKD population of spectacled eiders, assuming lead levels begin to decline in 2008 and ice conditions are consistent with RCP4.5 model predictions. Observed population sizes (open circles) and a subsample ($n = 55$) of population model projections (light grey lines) are shown. The quasi-extinction (QE) threshold is shown by the red line.

As described above, the first year when each population projection falls to, or below, 250 breeding individuals is retained to calculate the probability of quasi-extinction. As an example, figure 8 depicts the distribution of the year at which QE is met for the YKD population model, assuming lead declines in 2008 and ice conditions are consistent with RCP4.5 model predictions.

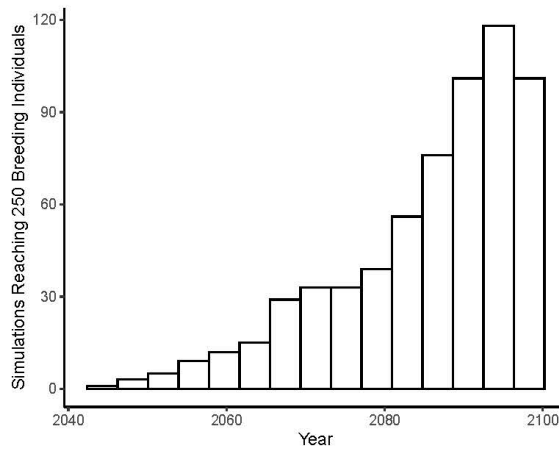


Figure 8. Distribution of year at which simulations that reached quasi-extinction (QE) for the YKD population model of spectacled eiders, assuming lead levels begin to decline in 2008 and ice conditions are consistent with RCP4.5 model predictions. The figure is limited to only those simulations where QE was met. Of 40,000 model simulations, 39,369 population trajectories did not meet QE.

The mean annual expected breeding population size under each scenario for the YKD and ACP populations are depicted below (Figures 9 and 10, respectively).

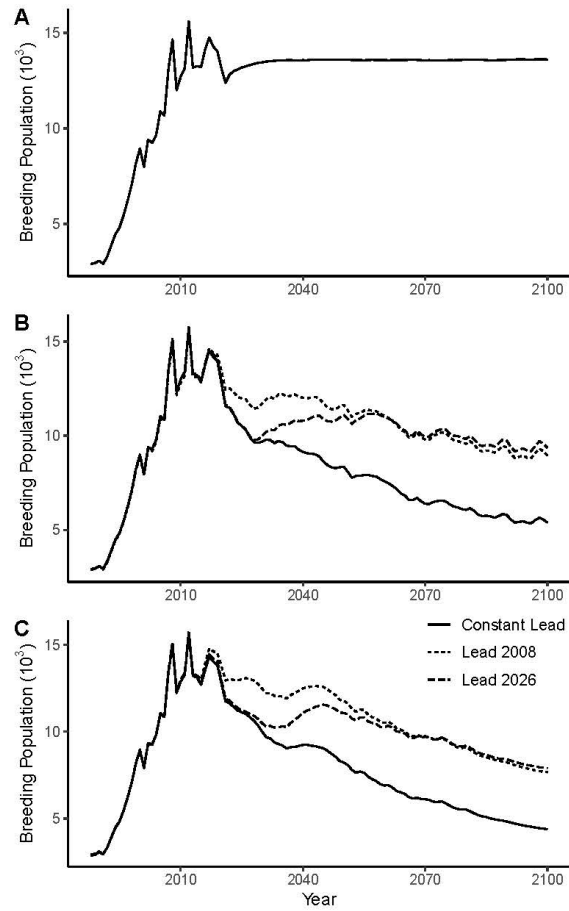


Figure 9. Estimated mean population projections for the YKD population under current conditions (A), RCP4.5 (B), and RCP8.5 (C).

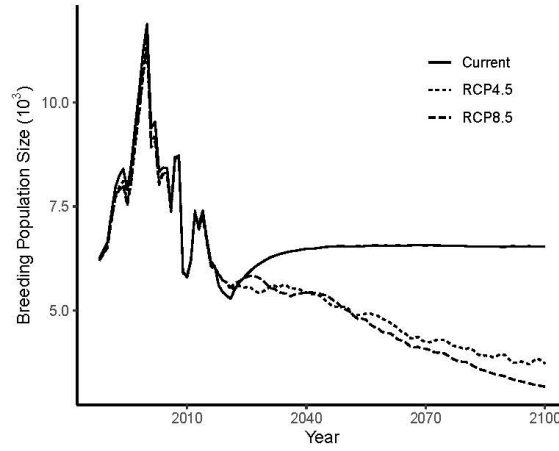


Figure 10. Estimated mean population projections for the ACP population under alternative extreme winter sea ice conditions.

One finding of particular interest is the weight of evidence for the quadratic effects of ice on survival and breeding propensity. Based on the current conditions model, for both the YKD and ACP populations, model estimates suggest negative effects at the posterior mean in all scenarios. Specifically, the posterior of ϕ_{0,ice^2} , the quadratic ice effect on juvenile survival, was 57.8% negative in the YKD model (-0.017, 95% CI: -0.192 - 0.169) and 83.4% negative in the ACP model (-0.088, 95% CI: -0.261 - 0.095). The posterior of the quadratic ice effect on adult survival, ϕ_{a,ice^2} , was similar in the YKD and ACP models (YKD: 65.9% negative, -0.039, 95% CI: -0.225 - 0.150; ACP: 69.0% negative, -0.048, 95% CI: -0.234 - 0.142). The negative effect in BP_{ice^2} , the quadratic ice effect on breeding propensity, was also similar between the two populations; however, that parameter was informed by a prior provided by the expert elicitation.

The estimated average growth rate for each population 1988-2019 is a second derived quantity of interest. For the YKD population, average growth rate is estimated as 1.053, 95% CI: 1.035 - 1.069. Across that same time period, the average growth rate for the ACP population is estimates as 0.995, 95% CI: 0.982 - 1.008.

Tables 1 and 2 summarize the probability of QE (%) before 2060 and between 2060 and 2100 for the YKD and ACP populations, respectively, under the alternative scenarios explored in the PVA.

Table 1. Quasi-extinction (QE) probabilities expressed as a percentage of runs where the population declined below 250 breeding individuals before 2060 and 2060-2100 for the YKD population under alternative scenarios of ice projections and lead exposure.

	QE before 2060 (%)	QE 2060-2100 (%)
Current Conditions	0.00	0.00
RCP4.5 - Lead 2008	0.07	1.51
RCP4.5 - Lead 2026	0.05	1.67
RCP4.5 - Lead Constant	0.20	6.48
RCP8.5 - Lead 2008	0.03	2.85
RCP8.5 - Lead 2026	0.06	3.44
RCP8.5 - Lead Constant	0.56	14.02

Table 2. Quasi-extinction (QE) probabilities expressed as a percentage of runs where the population declined

below 250 breeding individuals before 2060 and 2060-2100 for the ACP population under alternative scenarios of ice projections and lead exposure.

	QE before 2060 (%)	QE 2060-2100 (%)
Current Conditions	0.50	0.65
RCP4.5	0.51	5.39
RCP8.5	0.70	8.08

Under all conditions, the probability of QE for both populations is less than 1% before 2060. Consistent with expectations, the QE probability of the YKD population increases as the duration of high lead exposure rates lengthens. This relationship between lead exposure rates and QE is more pronounced under the RCP8.5 climate scenario than the RCP4.5 scenario.

Under current conditions, the QE probability of both populations remains low through 2100 (0% for the YKD population and 0.65% for the ACP population). However, reductions in winter sea ice, as predicted in both RCP4.5 and RCP8.5 scenarios, have moderate impacts to the QE probabilities in both cases.

Tables 3 and 4 summarize estimated population sizes (95% CI) for the YKD and ACP populations, respectively, at 2060 and 2100 under all model scenarios explored.

Table 3. Estimated breeding population size (95% CI) at 2060 and 2100 for the YKD population under alternative scenarios of ice projections and lead exposure.

	2060 Breeding Population	2100 Breeding Population
Current Conditions	13,298 (6,246 - 22,527)	13,313 (6,233 - 22,726)
RCP4.5 - Lead 2008	10,777 (2,482 - 21,436)	8,562 (434 - 20,352)
RCP4.5 - Lead 2026	10,669 (2,181 - 22,107)	8,933 (423 - 21,562)
RCP4.5 - Lead Constant	7,253 (1,122 - 16,203)	4,751 (60 - 14,860)
RCP8.5 - Lead 2008	10,156 (2,333 - 21,114)	6,921 (220 - 19,787)
RCP8.5 - Lead 2026	9,895 (1,812 - 21,141)	7,172 (160 - 20,422)
RCP8.5 - Lead Constant	6,474 (711 - 15,736)	3,263 (10 - 14,362)

Table 4. Estimated breeding population size (95% CI) at 2060 and 2100 for the ACP population under alternative scenarios of ice projections and lead exposure.

	2060 Breeding Population	2100 Breeding Population
Current Conditions	6,428 (1,804 - 12,207)	6,429 (1,566 - 12,203)
RCP4.5	4,595 (712 - 10,048)	3,510 (81 - 9,235)
RCP8.5	4,249 (612 - 9,727)	2,785 (39 - 8,735)

The full YKD model, written in JAGS, is given below. Note that in JAGS, normal distributions are specified in terms of precision and not standard deviation as in R (R Core Team 2020).

```
model {
  ## Priors
  # state-space
  sigma.obs[24] ~ dunif(800, 1400) # bounds consistent with s.e. observed since 2005
  sigma.obs[28] ~ dunif(800, 1400)

  # adding noise to the ice predictions s.t. interannual variation in the
```

```

# future is consistent with the observed time series (1980 - 2018)
for (i in 1:(K+1)){
  nb.prob[i] <- nb.size/(obs.ice[n.occasions + BEFORE + AFTER - 1 + i] + nb.size)
  ice.draw[i] ~ dnegbin(nb.prob[i], nb.size)
} # i

ice.new <- c(obs.ice[1:(n.occasions - 1 + BEFORE + AFTER)], ice.draw[])
ice <- (ice.new - mean(ice.new))/sd(ice.new)

# lead exposure and decay rate
theta_0 ~ dbeta(6, 45) # mean of 0.12, with 90% between 0.5 and 0.2
decay ~ dnorm(10, 0.1) T(1,) # assumes highly likely decay rate is between 5-15 years

### COMMENT OUT LEAD SCENARIOS DEPENDING ON RUN (DECLINE IN 2008, DECLINE IN
### 2026, OR CONSTANT LEAD)

# constant lead
#for (i in 1:(BEFORE+n.occasions-1+AFTER+K)){
#  lead[i] <- theta_0
#}

# decline in lead beginning in 2008 (21st year)
#for (i in 1:20){
#  lead[i] <- theta_0
#} # i
#for (i in 21:(BEFORE+n.occasions-1+AFTER+K)){
#  lead[i] <- 0.5^(1/decay)*lead[i-1]
#} # i

# decline in lead beginning in 2026 (39th year)
for (i in 1:38){
  lead[i] <- theta_0
} # i
for (i in 39:(BEFORE+n.occasions-1+AFTER+K)){
  lead[i] <- 0.5^(1/decay)*lead[i-1]
} # i

# productivity
mean.log.F ~ dnorm(-0.47,100)
mean.logit.BP ~ dnorm(0.472, 0.22) T(-0.77, 3.22)
betaF.ice ~ dnorm(2.83, 0.11) T(-1, 3)
betaF.ice2 ~ dnorm(-1.17, 0.69) T(-1.21, 0.33)
tau.BP <- pow(sigma.BP, -2)
sigma.BP ~ dunif(0,1)
for (i in 1:2){
  log.invDD[i] ~ dnorm(7.95, 3.31) # from EE
  invDD[i] <- exp(log.invDD[i])
  betaN[i] <- 1/invDD[i] #DD for phi and F
} #i

# survival and breeding propensity
mean.alpha.inv ~ dgamma(7.1, 1.95) T(1,) # from EE
mean.logit.alpha <- logit(1/mean.alpha.inv)

```

```

mean.phi0 ~ dbeta(15,45) # from cjs model
mean.logit.phi0 <- logit(mean.phi0)
tau.phi0 <- pow(sigma.phi0, -2)
sigma.phi0 ~ dunif(0, 1)

mean.phiA ~ dbeta(90,10) # from cjs model
tau.phiA <- pow(sigma.phiA, -2)
sigma.phiA ~ dunif(0, 1)

omegaA ~ dbeta(1, 35) # from EE
omegaJ ~ dbeta(1, 9) # from EE

mean.p ~ dbeta(1,1)
mean.logit.p <- logit(mean.p)
tau.p <- pow(sigma.p, -2)
sigma.p ~ dunif(0, 1)

for (i in 1:4){
  beta[i] ~ dnorm(0, 100)
} # i

beta.nest ~ dnorm(0, 100)

kappa ~ dbeta(20, 20) #relate phiA to phi2 by lead rate

## State-space model for count data
# Model for the initial population size
N[1,1] ~ dunif(100, 300) # n1, 1YO
N[2,1] ~ dunif(100, 350) # n2nB, 2YO non-breeding
N[3,1] ~ dunif(100, 250) # n2B, 2YO breeding
N[4,1] <- N[5,1] + N[6,1] # n3+, 3+YO breeding
N[5,1] ~ dunif(100, 350) # n3+ from n2nB
N[6,1] ~ dunif(100, 2500) # n3+ from n2B and n3+

# Process model
for (t in 1:(BEFORE+n.occasions-1+AFTER+K)){ # extended loop here
  N[1,t+1] <- F[t] * (N[3,t] + N[4,t])
  N[2,t+1] <- (phi1[t] + (1-phi1[t])*omegaJ)*(1-alpha[t])*N[1,t]
  N[3,t+1] <- (phi1[t] + (1-phi1[t])*omegaJ)*alpha[t]*N[1,t]
  N[4,t+1] <- N[5,t+1] + N[6,t+1]
  N[5,t+1] <- (phi2[t] + (1-phi2[t])*omegaA)*N[2,t]
  N[6,t+1] <- (phiA[t] + (1-phiA[t])*omegaA)*(N[3,t] + N[4,t])
} # t

# Observation model
tau.o <- pow(sigma.o, -2)
sigma.o ~ dgamma(1, 10)
tau.d <- pow(sigma.d, -2)
sigma.d ~ dgamma(2, 13) # from EE
for(i in 1:5){
  o[i] ~ dnorm(0, tau.o)
} # i
for (t in 1:(n.occasions+BEFORE+AFTER)){

```

```

d[t] ~ dnorm(0, tau.d) T(-0.69, 0.69)
count[t] ~ dnorm(2*exp(log(N[3,t] + N[4,t]) - o[obs[t]] - d[t]), tau.obs[t]) # count provided as data
tau.obs[t] <- pow(sigma.obs[t], -2) # sigma.obs provided as data
} # t

## Fecundity Model
# Process model
for (t in 1:(n.occasions+BEFORE+AFTER+K-1)){ # extended loop here
  logit.BP[t] <- mean.logit.BP + betaF.ice*ice[t] + betaF.ice2*ice[t]*ice[t]
  + eps.BP[t]
  eps.BP[t] ~ dnorm(0, tau.BP)
  BP[t] <- ilogit(logit.BP[t])
  F[t] <- exp(log(phi0[t] + (1-phi0[t])*omegaJ) + mean.log.F
    - betaN[1]*(N[3,t] + N[4,t]) + log(BP[t]))
} # t

## Multistate survival model
# process model
for (t in 1:(n.occasions-1)){
  nest[t] ~ dbeta(nest.obs.a[t], nest.obs.b[t])
}

nest.s <- (nest - mean(nest))/sd(nest)

for (t in (1+BEFORE):(BEFORE+n.occasions-1)){
  logit.p[t] <- mean.logit.p + beta.nest*nest.s[t-BEFORE] + eps.p[t]
  eps.p[t] ~ dnorm(0, tau.p)
  p[t] <- ilogit(logit.p[t])
} # t

for (t in 1:(n.occasions - 1 + BEFORE + AFTER + K)){
  logit.phi0[t] <- mean.logit.phi0 + beta[3]*ice[t+1]
    + beta[4]*ice[t+1]*ice[t+1] + eps.phi0[t]
  eps.phi0[t] ~ dnorm(0, tau.phi0)
  phi0[t] <- ilogit(logit.phi0[t])

  logit.phiA[t] <- logit(mean.phiA*(1-lead[t]*(1-kappa))) + beta[1]*ice[t+1]
    + beta[2]*ice[t+1]*ice[t+1]
    - betaN[2]*(N[3,t] + N[4,t]) + eps.phiA[t]
  logit.phi2[t] <- logit(mean.phiA) + beta[1]*ice[t+1]
    + beta[2]*ice[t+1]*ice[t+1]
    - betaN[2]*(N[3,t] + N[4,t]) + eps.phiA[t]
  eps.phiA[t] ~ dnorm(0, tau.phiA)
  phiA[t] <- ilogit(logit.phiA[t])
  phi2[t] <- ilogit(logit.phi2[t])

  phi1[t] <- phi2[t]

  logit.alpha[t] <- mean.logit.alpha
  alpha[t] <- ilogit(logit.alpha[t])
} # t

for (t in 1:(n.occasions - 1)){

```

```

# state transition and reencounter probabilities
psi[1,t,1] <- 0
psi[1,t,2] <- phi0[t+BEFORE]
psi[1,t,3] <- 0
psi[1,t,4] <- 0
psi[1,t,5] <- 0
psi[2,t,1] <- 0
psi[2,t,2] <- 0
psi[2,t,3] <- phi1[t+BEFORE]*(1-alpha[t+BEFORE])
psi[2,t,4] <- phi1[t+BEFORE]*alpha[t+BEFORE]
psi[2,t,5] <- 0
psi[3,t,1] <- 0
psi[3,t,2] <- 0
psi[3,t,3] <- 0
psi[3,t,4] <- 0
psi[3,t,5] <- phi2[t+BEFORE]
psi[4,t,1] <- 0
psi[4,t,2] <- 0
psi[4,t,3] <- 0
psi[4,t,4] <- 0
psi[4,t,5] <- phiA[t+BEFORE]
psi[5,t,1] <- 0
psi[5,t,2] <- 0
psi[5,t,3] <- 0
psi[5,t,4] <- 0
psi[5,t,5] <- phiA[t+BEFORE]
po[1,t] <- 0
po[2,t] <- 0
po[3,t] <- 0
po[4,t] <- p[t+BEFORE]
po[5,t] <- p[t+BEFORE]

# non-encounter probabilities, dq, and reshape the array for the
# encounter probabilities
for (s in 1:ns){
  dp[s,t,s] <- po[s,t]
  dq[s,t,s] <- 1-po[s,t]
} # s
for (s in 1:(ns-1)){
  for (m in (s+1):ns){
    dp[s,t,m] <- 0
    dq[s,t,m] <- 0
  } # m
} # s
for (s in 2:ns){
  for (m in 1:(s-1)){
    dp[s,t,m] <- 0
    dq[s,t,m] <- 0
  } # m
} # s
} # t

# multinomial likelihood

```

```

for (t in 1:(n.occasions-1)*ns){
  marr[t,1:(n.occasions*ns-(ns-1))] ~ dmulti(pr[t,], rel[t])
} # t

# Define the cell probabilities of the multistate m-array
# Define matrix U: product of probabilities of state-transition and non-encounter (this is just done be
for (t in 1:(n.occasions-2)){
  U[(t-1)*ns+(1:ns), (t-1)*ns+(1:ns)] <- ones
  for (j in (t+1):(n.occasions-1)){
    U[(t-1)*ns+(1:ns), (j-1)*ns+(1:ns)] <- U[(t-1)*ns+(1:ns), (j-2)*ns+(1:ns)] %**% psi[,t,] %**% dq[,t]
  } # j
} # t
U[(n.occasions-2)*ns+(1:ns), (n.occasions-2)*ns+(1:ns)] <- ones
# Diagonal
for (t in 1:(n.occasions-2)){
  pr[(t-1)*ns+(1:ns), (t-1)*ns+(1:ns)] <- U[(t-1)*ns+(1:ns), (t-1)*ns+(1:ns)] %**% psi[,t,] %**% dp[,t,]
  # Above main diagonal
  for (j in (t+1):(n.occasions-1)){
    pr[(t-1)*ns+(1:ns), (j-1)*ns+(1:ns)] <- U[(t-1)*ns+(1:ns), (j-1)*ns+(1:ns)] %**% psi[,j,] %**% dp[,.,]
  } # j
} # t
pr[(n.occasions-2)*ns+(1:ns), (n.occasions-2)*ns+(1:ns)] <- psi[,n.occasions-1,] %**% dp[,n.occasions-1,]

# Below main diagonal
for (t in 2:(n.occasions-1)){
  for (j in 1:(t-1)){
    pr[(t-1)*ns+(1:ns), (j-1)*ns+(1:ns)] <- zero
  } #j
} #t

# Last column: probability of non-recapture
for (t in 1:(n.occasions-1)*ns){
  pr[t,(n.occasions*ns-(ns-1))] <- 1-sum(pr[t,1:(n.occasions-1)*ns])
} #t

## Derived parameters
# Total population size and breeding population size
for (t in 1:(n.occasions + BEFORE + AFTER + K)){ # extended loop here
  Ntot[t] <- 2*(N[1,t] + N[2,t] + N[3,t] + N[4,t])
  Nb[t] <- 2*(N[3,t] + N[4,t])
}

# Check whether the population goes extinct in future
for (t in 1:K){ # extended loop here
  extinct[t] <- 1- step(Nb[n.occasions + BEFORE + AFTER + t] - 251) # quasiaextinction of 250
}
}
")

```

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Appendix C. Expert Elicitation in Support of the Spectacled Eider Population Viability Analysis.

Expert Elicitation in Support of the Spectacled Eider (*Somateria fischeri*) Population Viability Analysis

Erik Osnas and Catherine Bradley

15 December, 2020

Introduction

We developed an integrated population model-population viability analysis (IPM-PVA; see accompanying appendix) to support the discussion of the current status and future resilience of spectacled eider (*Somateria fischeri*) populations in the Yukon-Kuskokwim Delta (YKD) and Arctic Coastal Plain (ACP) in a Species Status Assessment (Smith et al. 2018). Several parameters in the model required expert elicitation to derive informed priors, as there is no data currently available to inform their estimates. We followed the IDEA protocol outlined in Hemmings et al. (2018) to gather the necessary quantitative data. Seven experts with a collective breadth of knowledge in sea duck ecology, particularly of eiders, and in demographic modeling began the exercise and five ultimately completed all steps. We developed ten questions concerning survival differences between the YKD and ACP populations, juvenile and adult local dispersal, winter sea ice effects on breeding propensity, the carrying capacity of the YKD and ACP populations, two year old maturation probability, and count survey interannual variability within observers. Responses were aggregated, summarized, and presented to the experts for a group discussion focused on ensuring consistent interpretation of the questions posed and allowing experts to communicate information or research that informed individual responses with the rest of the group. No attempt at consensus was made. Following this discussion, experts were given a second opportunity to respond to the questions, if they felt that revision was warranted. This second round of responses were again aggregated, summarized, and, ultimately, used to develop informed priors for model parameters.

Here we present the materials shared with the experts, questions used for elicitation, summaries of the final responses, and a discussion of how these responses were incorporated into the IPM-PVA.

Invitations

Invitations were sent to 12 experts in sea duck ecology and/or demographic modeling. We aimed for a group composed of both individuals with deep knowledge of the Alaska populations of spectacled eiders and individuals that could provide diverse perspectives, grounded in a strong background of relevant experience. The invitation included the purpose of the exercise, a brief description of the method we would employ, and an anticipated timeline. Seven individuals agreed to participate in the initial round.

Questions and Instructions

Questions were provided to the experts with instructions to use any research, knowledge, or experiences to inform their answers, but to answer the questions independently of one another. For each question, respondents were asked to give a minimum plausible value, maximum plausible value, most likely plausible value, and their confidence in the interval containing the true value. Participants had the option to abstain

from answering a question if they felt they could not provide an informed response. For illustrative purposes, an example question and answer set was given.

Question syntax changed slightly between the first round and second round of responses, following discussion of the intent of each question. For clarity, we present the final question set here.

Q1: Adult mean survival probability on the Arctic Coastal Plain (ACP)

As a reminder, the Arctic Coastal Plain (ACP) and Yukon-Kuskokwim Delta (YKD) spectacled eiders are separate breeding populations that winter in the same area. We have mark-recapture data from Kigigak Island during 1992 - 2015 to inform estimated survival probabilities for the population of adult spectacled eiders breeding on the YKD. Kigigak Island is a small island (about 8 km across and 32 km²) that represents about 5% of the core breeding area on the YKD. No such dataset exists for the ACP population. For the purposes of this question, adults are at least 3 years old. Our best estimate of mean annual adult apparent survival on Kigigak Island is 0.88 (95% CI: 0.82 - 0.92). Assuming there is no movement between the breeding populations, what is the mean annual adult true survival for the breeding population on the ACP? (domain [0,1])

Q2: Juvenile mean survival probability on the Arctic Coastal Plain (ACP)

As a reminder, the Arctic Coastal Plain (ACP) and Yukon-Kuskokwim Delta (YKD) spectacled eiders are separate breeding populations that winter in the same area. We have mark-recapture data from Kigigak Island during 1992 - 2015 to inform estimated survival probabilities for juveniles in the population of spectacled eiders breeding on the YKD. Kigigak Island is a small island (about 8 km across and 32 km²) that represents about 5% of the core breeding area on the YKD. No such dataset exists for the ACP population. For the purposes of this question, the juvenile period is from 40 days old to 1 year old. Our best estimate of mean juvenile apparent survival on Kigigak Island is 0.21 (95% CI: 0.14 - 0.29). Assuming there is no movement between the breeding populations, what is the mean juvenile true survival for the breeding population on the ACP? (domain [0,1])

Q3: Small-scale natal site dispersal

The mark-recapture data that provides an estimate of juvenile spectacled eider apparent survival is from a study on Kigigak Island. Kigigak Island is a small island (about 8 km across and 32 km²) that represents of the total Yukon-Kuskokwim Delta that represents about 5% of the core breeding area on the YKD. What proportion of juveniles born on Kigigak Island may never return to the Island to breed but may breed elsewhere on the YKD? (domain: [0,1])

Q4: Small-scale breeding site dispersal

The mark-recapture data that provides an estimate of adult spectacled eider apparent survival is from a study on Kigigak Island. Kigigak Island is a small island (about 8 km across and 32 km²) that represents about 5% of the core breeding area on the YKD. What proportion of adults that breed on Kigigak Island may never return to the Island to breed but may breed elsewhere on the YKD? (domain: [0,1])

Q5: Effect of winter sea ice conditions on productivity

The severity of the winter sea ice in the principal overwintering area south of St. Lawrence Island may impact the productivity of spectacled eiders in the following breeding season. Here, productivity refers to the per capita average annual production of fledglings (age 40 days). This could occur by influencing breeding propensity, nesting success, egg survival, clutch size, or duckling survival. We characterize the severity of

the winter sea ice by the number of days during which the principal overwintering area is 95% or more ice-covered (extreme sea ice days). From 1988 - 2018 the average number of extreme sea ice days was 60 (SD = 25, range = 0 - 96). We will ask two questions to address any possible non-linearity in expert judgements of the effect of winter sea ice conditions on productivity.

Assume that the average productivity of spectacled eiders, under average winter sea ice conditions, is 0.8 fledglings/breeding female. In a year with 85 extreme sea ice days (1 SD above the mean), what is the productivity following that winter? (domain [0, Inf])

Assume that the average productivity of spectacled eiders, under average winter sea ice conditions, is 0.8 fledglings/breeding female. In a year with 35 sea ice days (1 SD below the mean), what is the productivity following that winter? (domain [0, Inf])

Q6: Maximum equilibrium population size of the YKD population

Over the most recent 12 year period, estimates of the population size of breeding spectacled eiders on the YKD have been relatively stable around 15,000 birds. Assuming recent environmental conditions are maintained, what is the maximum equilibrium population size of breeding spectacled eiders on the YKD? (domain [0, Inf])

Q7: Maximum equilibrium population size of the ACP population

Over the most recent 12 year period, estimates of the population size of breeding spectacled eiders on the ACP have been relatively stable around 7,000 birds. Assuming recent environmental conditions are maintained, what is the maximum equilibrium population size of breeding spectacled eiders on the ACP? (domain [0, Inf])

Q8: Breeding propensity of 2 year old females

Female spectacled eiders may first attempt to breed at 2 years old. On average, what proportion of 2 year old females return to the breeding ground and attempt to breed? (domain [0,1])

Q9 Annual variation in aerial detection on the YKD

Average detection probability of breeding pairs of eiders on the YKD from aerial surveys is approximately 0.50 across observers, eider densities, and years. On the YKD, detection probability is determined from the ratio of eider nests estimated from ground-based plot surveys to observed eiders from the air using the standard North American aerial waterfowl survey protocol. Considering all years and factors affecting the ratio of aerial observations to nests, how much could detection probability differ among years within the same observer? For example, if you believe that an observer's detection probability most likely varies by 0.10 and that it is 0.50 on average, you are stating that the observer's detection probability could most likely be between 0.40 and 0.60 in any given year. (domain: [0, 0.5])

Q10: Annual variation in aerial detection on the ACP

Average detection probability of breeding pairs of eiders on the ACP from aerial surveys is approximately 0.75 when measured across three observers and years. Detection probability here is defined from a double observer mark-recapture experiment from two observers in the plane. Importantly for the ACP, detection is estimated as detection of birds and not nests. In all other respects the observation protocol is the same as for the YKD. Considering all factors that could affect detection of eiders from the air on the ACP, how much could detection probability differ among years within the same observer? For example, if you believe that an observer's detection probability could vary by 0.10 and that it is 0.75 on average, you are stating that the observer's detection probability could be between 0.65 and 0.85. (domain: [0, 0.25])

Summarization and Discussion

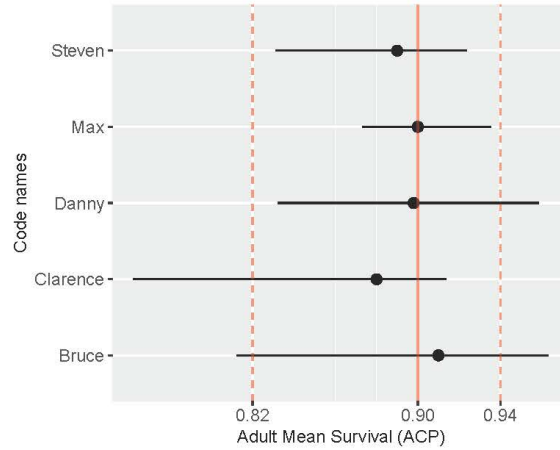
We reviewed the experts' first set of responses and followed up with those that chose values outside the domain provided or other minor issues that could be clarified before the group discussion (e.g., minimum value exceeding maximum value). Experts also made comments on some question syntax which gave us the opportunity to anticipate where language would need to be revised in the final question set and where there may be discrepancies in interpretation during the discussion.

Expert responses were standardized such that responses provided 80% confidence intervals (Hemmings et al. 2018) and figures were produced showing each respondent's interval and the average interval across responses (e.g., below). These figures were shared with participants prior to a discussion where we stepped through each question, discussed its intent and any ambiguities, and allowed the experts to share how they arrived at their responses. The revised question set, provided above, was shared with the experts following the discussion, and they had an opportunity to change their responses, if desired.

Final Responses and Prior Development

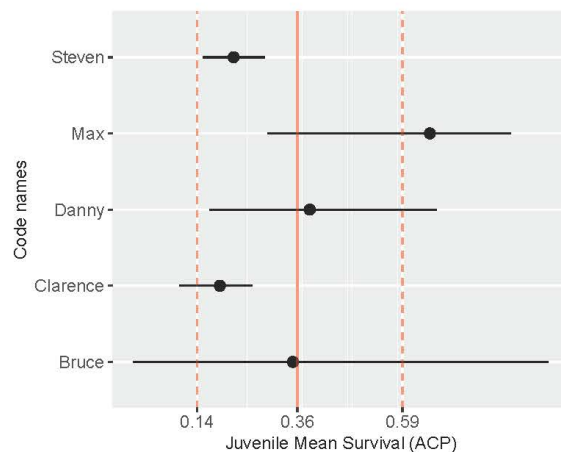
Five experts completed all stages of the expert elicitation. Below are the final response figures, where code names are used for the experts. For each solicited parameter, each individual's response, standardized to 80% confidence intervals, is shown in black and averaged responses for the most likely values (solid line) and bounds (dashed lines) are shown in red and indicated on the x-axis. Following each is a discussion of how this response was translated into a prior used in the IPM-PVA as well as any challenges that were encountered in doing so. We give a measure of central tendency (mean or median) and the 80% confidence interval of the asserted prior to compare to those elicited from the experts.

Q1: Adult mean survival probability on the Arctic Coastal Plain (ACP)



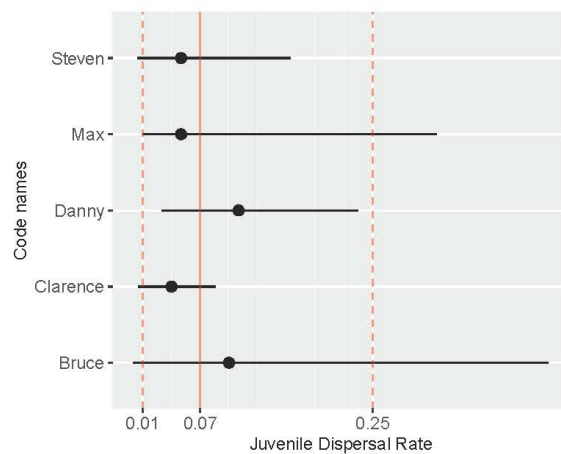
A $\text{beta}(26, 3)$ prior for mean true adult survival, μ_{ϕ_a} , was asserted to reflect expert responses. This distribution has a mean of 0.90 (80% CI: 0.82-0.96), and is bounded by 0 and 1. Because this reflects true survival, adult breeding site dispersal, ω_A , was not estimated in the model for the ACP.

Q2: Juvenile mean survival probability on the Arctic Coastal Plain (ACP)



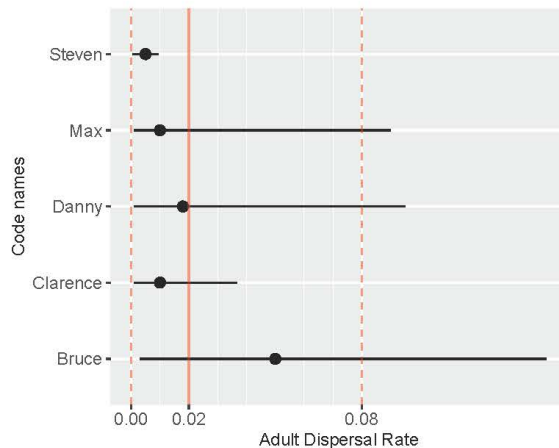
A $\text{beta}(2.3, 4)$ prior for mean true juvenile survival, μ_{ϕ_0} , was asserted to reflect expert responses. This distribution has a mean of 0.36, 80% bounds of 0.14 and 0.61, and is bounded by 0 and 1. As above, because this reflects true survival, natal site dispersal, ω_J , was not estimated in the model for the ACP.

Q3: Small-scale natal site dispersal



Because the tails of the elicited distribution were strongly asymmetrical, we interpreted the most likely value as a median. A $\text{beta}(1, 9)$ prior for natal site dispersal, ω_J , was asserted to reflect expert responses. This distribution has a median of 0.07, 80% bounds of 0.01 and 0.23, and is bounded by 0 and 1.

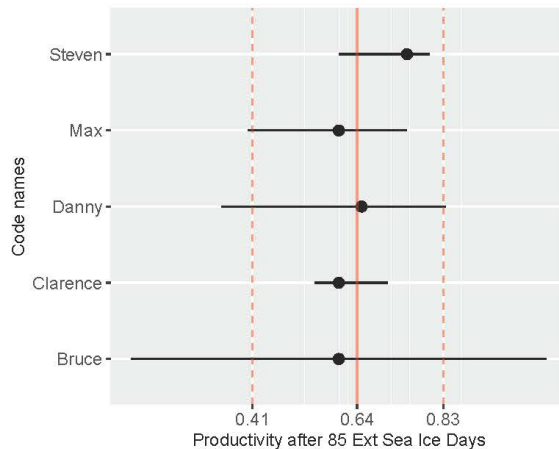
Q4: Small-scale breeding site dispersal



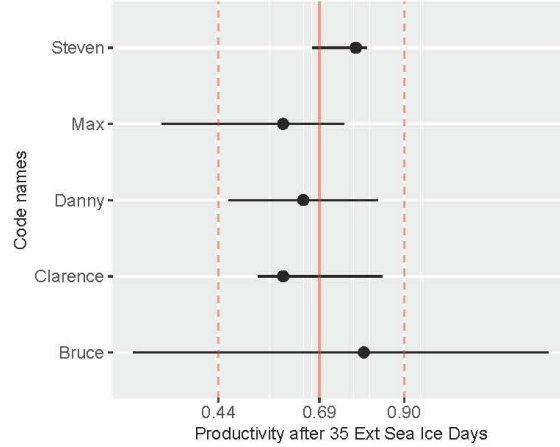
Again, because the tails of the elicited distribution were strongly asymmetrical, we interpreted the most likely value as a median. A $\text{beta}(1, 35)$ prior for natal site dispersal, ω_J , was asserted to reflect expert responses. This distribution has a median of 0.02, 80% bounds of 0.003 and 0.06, and is bounded by 0 and 1.

Q5: Effect of winter sea ice conditions on productivity

Assume that the average productivity of spectacled eiders, under average winter sea ice conditions, is 0.8 fledglings/breeding female. In a year with 85 extreme sea ice days (1 SD above the mean), what is the productivity following that winter? (domain $[0, \text{Inf}]$)



Assume that the average productivity of spectacled eiders, under average winter sea ice conditions, is 0.8 fledglings/breeding female. In a year with 35 sea ice days (1 SD below the mean), what is the productivity following that winter? (domain [0, Inf])



Expert responses suggested that there was a non-linear response in productivity to winter extreme sea ice days. However, during the discussion, experts stated that they would expect this relationship to be borne out through a reduction in breeding propensity rather than in any loss in fecundity. This required the addition of a new parameter to the model, as the breeding propensity of adult females had been assumed to be 1.0, consistent with previous demographic models of spectacled eiders (e.g. Flint et al. 2016). Assuming breeding propensity was 0.9 in average winter conditions, the experts' responses suggested that it would decline 20%, or to 0.72 (80% CI: 0.46-0.96), in high ice conditions (85 days) and 13.75%, or to 0.78 (80% CI: 0.50-1.0), in low ice conditions (35 days). However, note that the confidence interval in these effects are wide and a positive effect of low (and high) sea ice conditions were considered possible.

We assumed a model with linear and quadratic effects of sea ice on the logit-linked breeding propensity. We then created a large grid of possible covariate values for the model, fit each possible model to the standardized sea ice data, and calculated the sum of the absolute error between the expected values of breeding propensity at 35, 60, and 85 days and the predicted value using each model. We selected the model with the lowest absolute error. For example, the model

$$\text{logit}(BP) = 0.472 + 2.83 * ice - 1.17 * ice^2$$

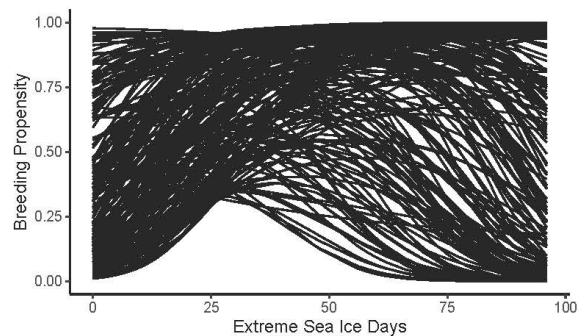
results in predicted breeding propensities of 0.77 with 35 extreme sea ice days, 0.90 with 60 extreme sea ice days, and 0.72 with 85 extreme sea ice days. This was repeated to generate a linear model of the lower and upper bounds of expected breeding propensity. Normal priors were asserted for the covariate values, truncating these distributions to reflect the upper and lower bounds

$$BP_0 \sim \text{Normal}(0.472, 0.22)T(-0.77, 3.22)$$

$$\beta_{BP,ice} \sim \text{Normal}(2.83, 0.11)T(-1, 3)$$

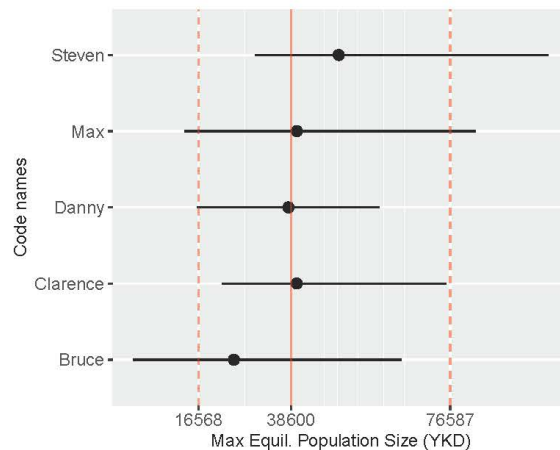
$$\beta_{BP,ice^2} \sim \text{Normal}(-1.17, 0.69)T(-1.21, 0.33).$$

Prior predictive plots demonstrate the uncertainty implied by these priors in the relationship between extreme sea ice days and breeding propensity. Below are 200 simulated curves using the priors specified above and observed extreme sea ice days from 1988-2018 and the RCP4.5 model-averaged expected extreme sea ice from 2019-2100 as the predictor.



Because future predictions in the IPM-PVA were made primarily at low extreme sea ice days, we prioritized fit to the expert elicited values at 35 days over fit at 85 days. Across a larger sample of 10,000 simulations from the priors, breeding propensity at 35 days is estimated to be 0.78 (80% CI: 0.48-0.95) which is consistent with the elicited most plausible value and 80% CI range.

Q6: Maximum equilibrium population size of the YKD population



The maximum equilibrium population size as elicited included males and females, so these values were first divided by two to fit the female-only population dynamics model used in the IPM-PVA. This constraint on population size was incorporated into the IPM-PVA by asserting negative linear density-dependence effects on survival and fecundity. Though modeled using different link functions, the logit and log, respectively, a simplifying assumption was that the effect size, on the link scale, was equal in the linear models for both survival and fecundity (i.e., the prior used in both linear models was the same).

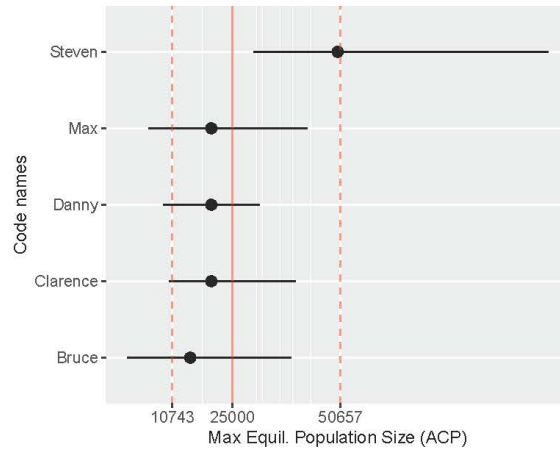
To derive the effect size needed to represent the elicited values of equilibrium population size, we built a deterministic population dynamics model based on the transition matrix, A , presented in the accompanying appendix. This model was initiated using the same initial population size and structure assumed in the IPM-PVA and mean values of breeding propensity, survival, and fecundity were asserted. The population

trajectory was simulated for 200 years under a variety of proposed density-dependence effect sizes and we identified effect sizes that resulted in equilibrium population sizes roughly equivalent to those proposed by the experts for the mean and 80% confidence interval bounds.

For the YKD population, a density-dependence effect of 0.00032 resulted in a population stabilizing at approximately 20,000 females. A lower bound of approximately 8,500 females was achieved with an effect of size 0.00076 and an upper bound of approximately 37,500 females was obtained with an effect size of 0.00017.

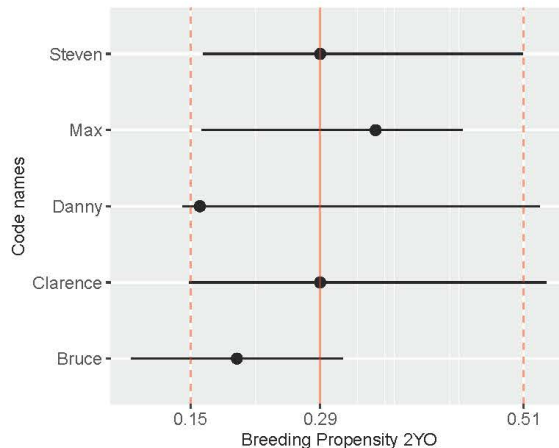
Standard probability distributions used to approximate small, non-negative values near zero such as the gamma or inverse-gamma were not able to represent these values well. However, the inverse of a log-Normal distribution with mean = 7.95 and standard deviation = 0.55 has a median value of 0.00036, lower 80% CI bound of 0.00018 and upper 80% CI bound of 0.00072. This prior was implemented in the IPM-PVA for the YKD population.

Q7: Maximum equilibrium population size of the ACP population



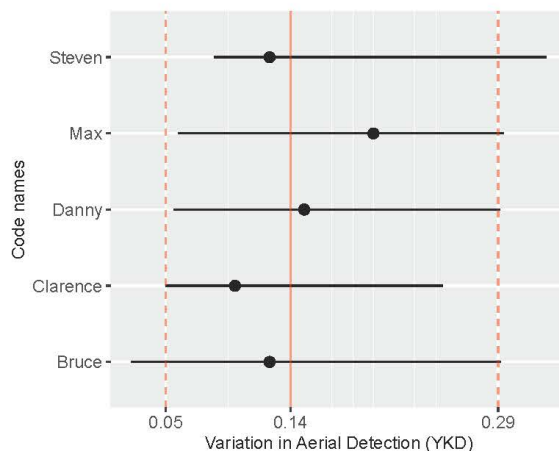
For the ACP population, we used the same method to translate the elicited population size into a density-dependence parameter. A density-dependence effect of 0.00089 resulted in a population stabilizing at approximately 12,500 females. A lower bound of approximately 5,400 females was achieved with an effect of size 0.00208 and an upper bound of approximately 25,000 females was obtained with an effect size of 0.00044. Therefore, we used a gamma(3, 3300) that gave a mean of 0.00091 and standard deviation = 0.00053 has a median value of 0.00081, lower 80% CI bound of 0.00033 and upper 80% CI bound of 0.0016. This prior was implemented in the IPM-PVA for the ACP population.

Q8: Breeding propensity of 2 year old females



To distinguish this parameter from adult breeding propensity (Q5), it was termed maturation probability, α , in the IPM-PVA. An inv-gamma(7.1, 1.95), truncated at 1, was used to represent the expert elicitation. This distribution has a median of 0.29 (80% CI: 0.18-0.49).

Q9 Annual variation in aerial detection on the YKD



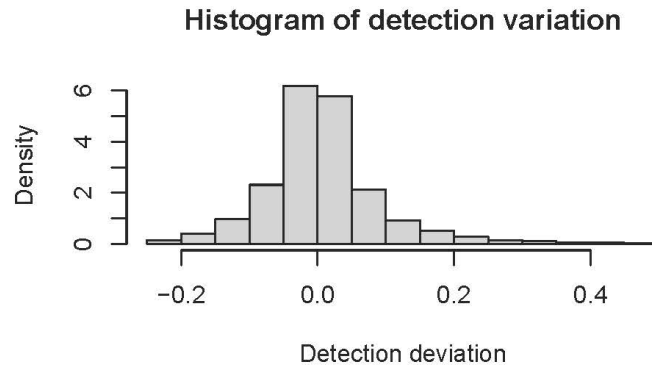
There was some ambiguity in the interpretation of this quantity for the YKD and ACP. We elicited the quantity on the probability scale because we felt that would be easier to discuss and for the experts to express their professional judgement. This quantity, however, was implemented as the prior distribution for the standard deviation of the log of the difference between average detection and year-specific detection. Therefore, we simulated from various priors for the standard deviation of the log difference in detection and

compared resultant distributions on the probability scale. We also assumed the deviation in year-specific detection was bounded to an absolute value of 0.5 for the YKD. Therefore, the year-specific deviation was simulated from a truncated normal distribution. For the YKD, we used

$$\sigma_d \sim \text{Gamma}(2, 13),$$

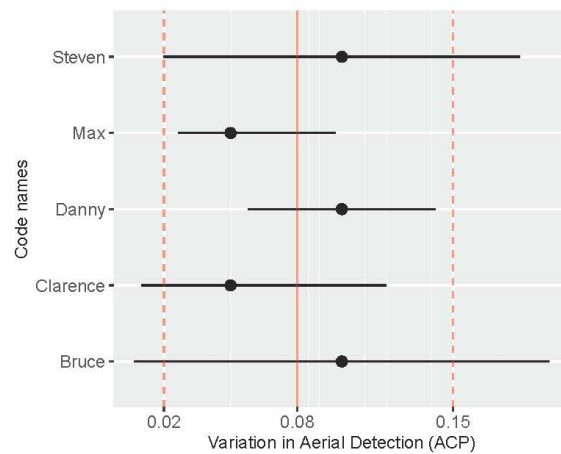
$$d_t \sim \text{Normal}(0, \sigma_d)T(-0.69, 0.69),$$

where d_t is the log difference in detection from the average detection. Given the above prior and hyperprior, the distribution of detection deviation is as in the figure below.



The above distribution gives a median of 0.04 and 80% CI of 0.01 - 0.14 for the absolute value of the deviation. This is significantly lower than that elicited. We will likely update this in future model versions.

Q10: Annual variation in aerial detection on the ACP

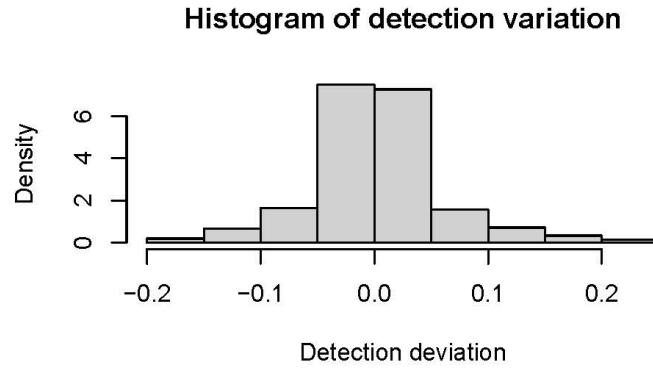


For the ACP, we used

$$\sigma_d \sim \text{Gamma}(1, 13),$$

$$d_t \sim \text{Normal}(0, \sigma_d)T(-0.28, 0.28),$$

where d_t is the log difference in detection from the average detection. Given the above prior and hyperprior, the distribution of detection deviation is as in the figure below.



The above distribution gives a median of 0.02 and 80% CI of 0 - 0.1 for the absolute value of the deviation. **This is significantly lower than that elicited. We will likely update this in future model versions.**