JOURNAL OF
AVIAN BIOLOGY

*Research article*

# Patterns in avian reproduction in the Prudhoe Bay Oilfield, Alaska, 2003–2019

**Rebecca L. McGuire**[1], **Martin Robards**[1] **and Joseph R. Liebezeit**[1,2]

[1]Wildlife Conservation Society, Arctic Beringia Program, Fairbanks, Alaska
[2]Audubon Society of Portland, Portland, Oregon, USA

**Correspondence: Rebecca L. McGuire (rlmcguire@alaska.edu)**

**Journal of Avian Biology**
**2023: e03075**
doi: 10.1111/jav.03075
Subject Editor:
Ronald C. Ydenberg
Editor-in-Chief: Staffan Bensch
Accepted 20 February 2023

The Arctic Coastal Plain is one of the most important avian breeding grounds in the world; however, many species are in decline. Arctic-breeding birds contend with short breeding seasons, harsh climatic conditions, and now, rapidly changing, variable, and unpredictable environmental conditions caused by climate change. Additionally, those breeding in industrial areas may be impacted by human activities. It is difficult to separate the impacts of industrial development and climate change; however, long-term datasets can help show patterns over time. We evaluated factors influencing reproductive parameters of breeding birds at Prudhoe Bay, Alaska, 2003–2019, by monitoring 1265 shorebird nests, 378 passerine nests, and 231 waterfowl nests. We found that nest survival decreased significantly nearer high-use infrastructure for all guilds. Temporally, passerine nest survival declined across the 17 years of the study, while there was no significant evidence of change in their nest density. Shorebird nest survival did not vary significantly across years, nor did nest density. Waterfowl nest density increased over the course of the study, but we could not estimate nest survival in all years. Egg predator populations varied across time; numbers of gulls and ravens increased in the oilfields 2003–2019, while Arctic fox decreased, and jaeger numbers did not vary significantly. Long-term datasets are rare in the Arctic, but they are crucial for understanding impacts to breeding birds from both climate change and increasing anthropogenic activities. We show that nest survival was lower for birds nesting closer to high-use infrastructure in Arctic Alaska, which was not detected in earlier, shorter-term studies. Additionally, we show that Lapland longspur nest survival decreased across time, in concert with continent-wide declines in many passerine species. The urgency to understand these relationships cannot be expressed strongly enough, given change is continuing to happen and the potential impacts are large.

Keywords: Arctic, infrastructure, nest density, nest survival, passerines, shorebirds, waterfowl

Arctic Alaska is a breeding ground for millions of birds of over 30 species (BLM 2020). These birds migrate along the four major North American flyways, as well as the East Asian-Australasian Flyway and the Central Pacific Flyway. During their travels, birds

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reach virtually every state in the nation, as well as Central and South America, Russia, China, Japan, Africa, Australia, New Zealand, and Antarctica ( ASG 2019). Recent work by Rosenberg et al. (2019) demonstrates a loss of nearly three billion birds to the North American avifauna since 1970; importantly, this work detailed a nearly 25% decrease in birds from Arctic tundra regions of North America and reinforces earlier reports of declines in Arctic-breeding birds, particularly shorebirds (Stehn et al. 1993, Morrison et al. 2006, Andres et al. 2012). These declines are due to a host of reasons throughout their annual cycles. However, the breeding grounds, by definition, are the only place where populations can be replenished. The breeding grounds in northern Alaska contain bird nurseries of global note for their productivity and high shorebird species diversity. These Arctic nurseries are also at risk due to the rapid escalation in impacts from a climate changing at least three times faster than that of most of the globe (AMAP 2021). Simultaneously, human infrastructure, and particularly infrastructure associated with oil and gas extraction, is increasing throughout the circumpolar Arctic.

There is a host of scientifically documented climate-related stressors that may impact nest survival of tundra-nesting birds. Ecological niche models suggest suitable breeding conditions for shorebirds will decrease with time, forcing birds to adapt, or more likely move north and east to areas in the Eurasian and Canadian Arctic islands (Wauchope et al. 2017). Important habitats, such as low-centered polygons and low-lying basins (Cunningham et al. 2016), are already drying up (Liljedahl et al. 2016), possibly reducing suitability for nesting shorebirds. Rising sea levels, more severe coastal storms, and glacial melt rates are modifying sediments in Arctic river deltas, which reduce invertebrates, a critical food for thousands of post-breeding shorebirds (Churchwell 2015, Churchwell et al. 2016). Shrubs and predators (e.g., red fox) are increasing in the Arctic tundra (Tape et al. 2006, Elmhagen et al. 2017, Parrett et al. 2022), making habitats less suitable and potentially more dangerous. Earlier emergence of invertebrates resulting from earlier warm weather, may be creating a phenological mismatch (McKinnon et al. 2012, Saalfeld and Lanctot 2017, Machín et al. 2018, Kwon et al. 2019, Saalfeld et al. 2019) that can reduce chick survival (Senner et al. 2017), and may ultimately drive long-term population declines (van Gils et al. 2016). Lastly, greater variability in seasonal weather and dates of snowmelt can dramatically reduce breeding success in some years (Meltofte et al. 2007, Richter-Menge et al. 2019, Saalfeld et al. 2019, Schmidt et al. 2019, McGuire et al. 2020). There is also some evidence that nest predation of shorebirds has increased threefold over the last 70 years and that the larger increase in the Arctic relative to the tropics indicates a link to climate change (Kubelka et al. 2018). However, there are some concerns about the dataset and analyses used to come to this conclusion (Bulla et al. 2019), and the question as to how climate change is impacting nest survival of Arctic-nesting birds has not been answered adequately, although it is clearly of great importance.

There are also a variety of factors associated with industrial development that may directly or indirectly affect nesting birds in northern Alaska, including habitat degradation via hydrology alteration and road dust, vehicle and aircraft traffic, noise, air pollution, and increased nest predator populations associated with development in the Prudhoe Bay oil fields. (NRC 2003, Liebezeit et al. 2009). Populations of avian nest predators have increased in the oilfields (NRC 2003), in part because of the availability of artificial breeding and perching sites (buildings, towers, culverts, and similar structures), heat sources, and year-round availability of anthropogenic food subsidies (via landfills, dumpsters, and hand-outs; NRC 2003, Stickney et al. 2014). Known egg predators in the Prudhoe Bay region include glaucous gulls *Larus hyperboreus*, common ravens *Corvus corax*, Arctic fox *Alopex lagopus*, red fox *Vulpes vulpes*, and jaegers *Stercorarius* spp.. Previous work demonstrated that Arctic foxes have been the most significant nest predators in this region (Liebezeit and Zack 2008), but more recent studies have documented an increase of red foxes and a decrease of Arctic foxes in the oilfields (Stickney et al. 2014, Parrett et al. 2022). Liebezeit et al. (2009) showed that nest productivity of some shorebird species is lower close to industrial infrastructure, presumably due to subsidized predators or disturbance. Liebezeit et al. (2009) relied on an unbalanced gradient of nests that were either relatively close or very far from infrastructure and it is possible additional effects of infrastructure exist but were not detected. Bentzen et al. (2017) did not find a significant relationship between nest survival and distance to high-use infrastructure or amount of infrastructure near the nest, however they relied on only three years of data and used an experimental design with primarily artificial nests.

Nest predation is reported to be a significant cause of nest failure for shorebirds, passerines, and waterfowl in many locations within the Alaskan Arctic (Pamplin 1986, Liebezeit et al. 2009). Predation pressures on tundra-nesting birds are thought to have changed in recent decades as human development has led to increases in subsidized predator populations by altering predator distribution, productivity, and survival (NRC 2003, Liebezeit et al. 2009, Stickney et al. 2014). However, predation pressures on Arctic-nesting birds may also be changing due to climate change (Kubelka et al. 2018).

Different groups of birds likely respond to climate change differently; waterfowl, shorebirds and passerines have very different life history strategies and habitat requirements. Even within the shorebird guild, there is great diversity in reproductive strategies. It has been suggested that one overarching ecological factor that might explain the ability of shorebirds to adapt to changing climatic conditions on Arctic breeding grounds is their reproductive strategy (e.g. conservative and opportunistic; Saalfeld and Lanctot 2015, Saalfeld and Lanctot 2017, McGuire et al. 2020). Briefly, conservative shorebirds typically display nest-site fidelity and territoriality, consistent population densities, relatively even individual spacing, and monogamous mating systems with bi-parental incubation. In contrast, opportunistic shorebirds display the opposite traits, and a polygamous mating system with uniparental incubation conservative. A primary facet of this classification is bi-parental versus uniparental incubation

(McGuire et al. 2020). Three studies on shorebirds have evaluated whether being conservative or opportunistic affected breeding response to seasonal variation in environmental conditions (Saalfeld and Lanctot 2015, Saalfeld and Lanctot 2017, McGuire et al. 2020). Saalfeld and Lanctot (2015, 2017) found that conservative species tended to have low variability in annual nest densities, and that there were some phenotypically flexible adjustments in most species to snowmelt, although opportunistic species appeared to adjust better than conservative species. However, McGuire et al. (2020) found that shorebirds of both strategies, across multiple sites in northern Alaska, bred earlier and in higher numbers in early, warm springs relative to historic levels, with opposite trends being observed in late springs. The degree to which shorebirds have the behavioral flexibility to optimally track changing environmental conditions in response to variable Arctic conditions is unknown.

Long-term ecological datasets (> 10 years) for migratory birds are rare in the Arctic and are vitally important for evaluating potential impacts from industry, and how these impacts can be separated from the impacts of climate change. In this study, we analyzed a long-term data set to evaluate the patterns of nest survival, nest density, nest initiation, and egg predator numbers in relation to infrastructure and during two decades of rapid climate change at Prudhoe Bay, the heart of oil industrial development in northern Alaska.

### Study area

We monitored shorebird, passerine and waterfowl nests over 17 field seasons (2003–2019) on Alaska's Arctic Coastal Plain in the Prudhoe Bay oilfields (148°20′W, 70°15′N). The study site was in an area developed with oil industry buildings and other structures along with an associated road network and human activities. The site consists of a mosaic of low, wet tundra habitats and higher, well-drained upland habitats (Kessel and Cade 1958, Brown et al. 1980). Wetland community types included wet sedge *Carex* spp. meadows, moist sedge–dwarf shrub (e.g. willow *Salix* spp.) meadows, and emergent sedge and pendant grass *Arctophila fulva* on the margins of lakes and ponds (Anderson et al. 1999).

## Methods

Twelve permanent study plots, each measuring 100 × 1000 m, were established in 2003 at Prudhoe Bay and were monitored each year as part of a long-term monitoring study. The plots were placed > 300 m from roads and buildings to minimize any potential influence of human activity (Liebezeit et al. 2009). We located nests by conducting area searches or dragging ropes across the tundra to flush adults from nests and by following birds exhibiting behaviors indicative of nesting back to their nests (Brown et al. 2014).

We estimated nest initiation date (date first egg laid) based on the number of eggs if nests were found during egg-laying (assuming one egg laid per day for all taxa, although plovers may take 1.5 days, Colwell 2006), or by back-calculating from known hatch date using standard incubation duration. If these two methods could not be used, we employed an egg-floatation technique to estimate nest initiation (Sandercock 1998, Liebezeit et al. 2007, Brown et al. 2014). This technique relies on the fact that eggs lose mass as the embryos inside develop, causing them to sink initially and later float in water. Nests were revisited generally every five days during incubation. We recorded a nest as hatched if at least one chick was observed in the nest, or if eggshell fragments indicative of hatching or an egg tooth were found in the nest within four days of the expected hatch date (Brown et al. 2014). In the case of waterfowl, detached membranes found in the nest were also considered to be evidence of hatching. We recorded the hatch date as 1) the day that downy chick(s) were first found in the nest, 2) the day after eggs were observed with pipped holes in the shells, or 3) two days after eggs were observed with star-cracks in the shells (Brown et al. 2014). We classified nests as unsuccessful or failed if we found broken eggshells indicative of predation in the nest, if the clutch disappeared more than four days before the predicted hatch date, or if the eggs remained unattended by parents for ≥ 3 days. We recorded nest fate as unknown if there was unclear or conflicting evidence at the nest site (Brown et al. 2014). If fate was unknown ($n = 107$), nests were considered successful until the last day they were known to be active (a metric important for the nest survival analysis). We did not include nests found at hatch for nest survival analyses. For each year we calculated estimates for nest density (i.e. cumulative number of nests found throughout study plot(s) divided by the total area of the plot(s) in km$^2$).

We recorded the dominant landform and vegetation type within a 5 m radius centered on the nest during the final nest visit (when no longer active) following the classification in the Geobotanical Atlas of the Prudhoe Bay Region, Alaska (Walker et al. 1981). For the analysis, we grouped these categories as wetland, moist habitat, or dry habitat. Wetland habitats included Walker et al. (1981) 'non-patterned ground', 'low-centered polygons', and 'strangmoor and/or disjunct polygon rims' landforms, and a vegetation type of 'wet' or 'emergent'. Moist habitats included Walker et al. (1981) 'mixed landform' and 'moister' vegetation type. Upland habitats included Walker et al. (1981) 'pingo', and 'high-centered polygons' and the 'driest' vegetation types. Occasionally landform and vegetation types did not agree, in which case we classified by vegetation type.

We determined the loss of snow at our sites in each year by estimating the percent daily snow cover on plots every 2–5 days between late May and the end of June or until 10% snow cover remained. We extrapolated snow cover within each plot between sample days using linear regression and averaged each day across plots. We present the average snow cover across all survey plots 7–15 June. Average nest initiation across all years and for all species was 11 June.

In each year, we conducted three sessions (early, mid, and late season) of timed (10 min) point counts between 12 June and 23 July following the methods of Liebezeit et al.

(2008). All counts were performed between 8:00 and 20:00 h Alaska Standard Time (AST). A point count session on each plot consisted of recording all observations of potential nest predators that were detected during 10 min periods from three fixed locations (at least 200 m apart) with at least 10 min between consecutive counts. We conducted these counts following methods described by Ralph et al. (1993). We estimated predator distance from the observer by using rangefinders, by judging the distance using the plot marker stakes as reference points, or by pacing the distance on foot. We counted only species that have been implicated as potential nest predators in previous studies. We followed the infrastructure classification system of Liebezeit et al. (2009), which included five categories based on potential to provide nest predators with food or sites for perching, nesting, or denning based on potential for food availability and a combination of vertical height and structural complexity. We calculated the distance to 'high-use' infrastructure (i.e. infrastructure with a high potential to support subsidized predators); this included infrastructure that was of 'high' or 'medium' food attraction and 'high' structural attraction using information supplied by BP Alaska, Inc. in 2013; this dataset is appropriate as the infrastructure (pads, roads, and buildings) did not change substantially over the course of the study. Additionally, we calculated the area of the underlying anthropogenic gravel footprint within circular 2 km buffers centered on each nest (referred to as infrastructure density in text; see Liebezeit et al. 2009) and the distance to the nearest road. We used ArcMap ver. 10.2 to measure the proximity of nests to oil field structures and to calculate the area of infrastructure within the defined buffers.

### Analysis

We used generalized linear models and a logit link (Program MARK; White and Burnham 1999, Dinsmore et al. 2002) to evaluate daily nest survival. To reduce the number and complexity of the models considered, we examined a hierarchical model set. Models of daily nest survival varied by year, guild (shorebird, waterfowl or passerine), habitat (wetland, moist, upland), initiation date, and incubation strategy (biparental or uniparental, shorebirds only), distance to infrastructure, area of infrastructure within a 2 km radius of the nest, and distance to nearest road. We ran all main effects models, then combined 'strong' effects additively and multiplicably. We included one post-hoc model to further evaluate the relationship between spring conditions and incubation strategy where we held passerines and waterfowl constant but allowed shorebird daily survival rate to vary by individual year and incubation strategy ($\mathrm{DSR}_{\mathrm{year\times strategy}}$). We compared models using $\mathrm{AIC}_c$ (Akaike's Information Criterion corrected for small sample sizes), in which we considered the model with lowest $\mathrm{AIC}_c$ value to be the best-fitting, and models with $\Delta\mathrm{AIC}_c < 2$ that did not add to model complexity to be plausible (Arnold 2010, Burnham and Anderson 2002).

We calculated estimates of nest density (i.e. the cumulative number of nests found throughout the study plots divided by the total area of the plots (km$^2$)) by year, guild, and incubation strategy (shorebirds only).

Values are reported as means ± standard error.

## Results

We monitored 1265 shorebird nests, 231 waterfowl nests, and 378 passerine nests between 2003 and 2019 (Table 1). Biparental shorebirds included American golden-plover *Pluvialis dominica*, black-bellied plover *P. squatarola*, dunlin *Calidris alpina*, long-billed dowitcher *Limnodromus scolopaceus*, ruddy turnstone *Arenaria interpres*, semipalmated sandpiper *C. pusilla*, and stilt sandpiper *C. himantopus*. Uniparental shorebirds included buff-breasted sandpiper *C. subruficollis*, pectoral sandpiper *C. melanotos*, red phalarope *Phalaropus fulicarious*, and red-necked phalarope (*Ph. Lobatus*; Table 1). Shorebird nests ranged from 0.8 to 12.2 km from the nearest high-use infrastructure (4.1 ± 0.07 km), waterfowl nests from 0.8 to 26.6 km (3.9 ± 0.17 km), and passerine nests from 0.8 to 12.1 km (4.0 ± 0.13 km). Area of infrastructure within 2 km of the nests ranged from 0 to 1.24 km$^2$ for shorebirds (0.34 ± 0.01 km$^2$), 0 to 1.2 km$^2$ for passerines (0.35 ± 0.01 km$^2$), and 0 to 1.2 km$^2$ for waterfowl (0.33 ± 0.01 km$^2$). Distance to the nearest roads varied from 0.1 to 2.4 km for shorebirds (1.1 ± 0.01 km), 0 to 2.4 km for passerines (1.1 ± 0.03 km), and 0 to 2.2 km for waterfowl (1.1 ± 0.03 km).

The top a priori model included an interaction between guild and year (shorebirds and passerines only), guild and initiation date, an additive effect of biparental incubation strategy for shorebirds and distance to infrastructure ($\mathrm{DSR}_{\mathrm{Guild+guild\times year+guild\times init+biparental1+distinfra}}$). $\mathrm{DSR}_{\mathrm{Guild+guild\times year+guild\times init+biparental1+distinfra}}$ was 0.49 $\mathrm{AIC}_c$ units from the next best a priori model ($\mathrm{DSR}_{\mathrm{Guild+guild\times year+guild\times init+biparental+areainfra}}$; Table 2) and carried 0.20 $\mathrm{AIC}_c$ model weight. The top four models carried 0.57 of total $\mathrm{AIC}_c$ model weights and varied only by which infrastructure variables were included (Table 2). Nest survival increased at greater distances to infrastructure (ß$_{\mathrm{Dist\_infra}} = 0.04$, SE = 0.02, 95% CL 0.001–0.08; Fig. 1), overall, and for each guild, although effects with guilds separated were insignificant (Waterfowl, ß$_{\mathrm{Dist\_infra}} = 0.09$, SE = 0.06, 95% CL -0.027 to 0.20; Shorebirds, ß$_{\mathrm{Dist\_infra}} = 0.03$, SE = 0.03, 95% CL -0.02 to 0.08; Passerines, ß$_{\mathrm{Dist\_infra}} = 0.04$, SE = 0.04, 95% CL -0.04 to 0.11). The effect size of ß$_{\mathrm{Dist\_infra}}$ results in an estimated 63% (95% CI: 0.61–0.65) nest survival at 1 km from the nearest high-use infrastructure and a 68% (95% CI: 0.67–0.71) nest survival if the same nest is located at 6 km from the nearest high-use infrastructure (estimated for a biparental shorebird with a 22-day incubation period (dunlin), that initiated on day 10 in the 9th year of the study).

While there was some support for the model containing the variable area infrastructure ($\mathrm{DSR}_{\mathrm{Guild+guild\times year+guild\times init+biparental+areainfra}}$; Table 2), the estimate that nest survival decreased with increasing infrastructure within 2 km of the nest did not reach significance

Table 1. Number of nests found at Prudhoe Bay, Alaska, 2003–2019. Species included in the other category are unidentified eiders, Pacific loon, tundra swan and sandhill cranes.

| | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 | Species total |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Bi-parental shorebirds | | | | | | | | | | | | | | | | | | |
| American golden-plover | 1 | 5 | 1 | 2 | 0 | 0 | 2 | 3 | 1 | 2 | 1 | 0 | 1 | 1 | 0 | 2 | 0 | 22 |
| Black-bellied plover | 2 | 3 | 2 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 9 |
| Dunlin | 4 | 11 | 3 | 2 | 1 | 1 | 5 | 4 | 3 | 4 | 1 | 1 | 2 | 2 | 4 | 3 | 2 | 53 |
| Long-billed dowitcher | 4 | 4 | 0 | 1 | 1 | 4 | 3 | 1 | 4 | 4 | 5 | 3 | 2 | 10 | 3 | 2 | 2 | 53 |
| Ruddy turnstone | 0 | 2 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| Semipalmated sandpiper | 25 | 51 | 42 | 32 | 26 | 19 | 23 | 35 | 42 | 31 | 25 | 31 | 23 | 25 | 29 | 21 | 19 | 499 |
| Stilt sandpiper | 5 | 7 | 3 | 5 | 2 | 7 | 6 | 5 | 8 | 5 | 3 | 2 | 4 | 2 | 4 | 4 | 1 | 73 |
| Uniparental shorebirds | | | | | | | | | | | | | | | | | | |
| Buff-breasted sandpiper | 1 | 0 | 0 | 2 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 1 | 0 | 0 | 0 | 2 | 0 | 10 |
| Pectoral sandpiper | 16 | 22 | 23 | 26 | 12 | 10 | 8 | 21 | 25 | 4 | 12 | 29 | 28 | 10 | 8 | 12 | 25 | 291 |
| Red phalarope | 15 | 12 | 5 | 10 | 4 | 4 | 7 | 7 | 2 | 3 | 3 | 7 | 4 | 2 | 2 | 3 | 0 | 90 |
| Red-necked phalarope | 6 | 10 | 8 | 13 | 10 | 11 | 7 | 4 | 5 | 12 | 12 | 14 | 12 | 15 | 7 | 12 | 2 | 160 |
| Waterfowl | | | | | | | | | | | | | | | | | | |
| Greater scaup | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 2 |
| Greater white-fronted goose | 7 | 14 | 12 | 13 | 0 | 0 | 6 | 11 | 16 | 7 | 10 | 17 | 8 | 12 | 10 | 13 | 26 | 182 |
| King eider | 3 | 1 | 2 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 0 | 11 |
| Canada and cackling geese | 3 | 1 | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 2 | 3 | 2 | 3 | 0 | 3 | 22 |
| Long-tailed duck | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 1 | 0 | 5 |
| Northern pintail | 1 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 0 | 0 | 1 | 5 |
| Other | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 4 |
| Passerines | | | | | | | | | | | | | | | | | | |
| Lapland longspur | 24 | 36 | 21 | 39 | 17 | 21 | 32 | 32 | 19 | 15 | 19 | 24 | 18 | 18 | 15 | 18 | 10 | 378 |
| Nest density by year (nests/km$^2$) | 67.5 | 52.5 | 75.0 | 102.5 | 58.3 | 62.5 | 76.7 | 95.8 | 95.0 | 74.2 | 81.7 | 112.5 | 92.5 | 80.8 | 70.0 | 75.8 | 75.0 | |

Table 2. Models of daily survival rate (DSR) of shorebird, passerine and waterfowl nests found at Prudhoe Bay, Alaska, 2003–2019. Variables included year, guild (shorebird, waterfowl or passerine), habitat (wetland, moist, upland), initiation date (init), and incubation strategy (biparental, shorebirds only), distance to infrastructure (distinfra), area of infrastructure within a 2 km radius of the nest (areainfra), and distance to nearest road (distroad). Shown are are all models within 6 $AIC_c$ of the top model and the null model. Waterfowl nests were not found on-plot in all years and there we did not model waterfowl daily nest survival by year.

| DSR model | $AIC_c$[b] | $\Delta AIC_c$[c] | $w_i$[d] | $K$[a] | Deviance |
|---|---|---|---|---|---|
| Guild + guild×year[1] + guild×init + biparental + distinfra | 3426.74 | 0.00 | 0.20 | 9 | 3408.73 |
| Guild + guild×year + guild×init + biparental + areainfra | 3427.23 | 0.49 | 0.16 | 9 | 3409.22 |
| Guild + guild×year + guild×init + biparental + distinfra + distroads | 3427.83 | 1.09 | 0.12 | 10 | 3407.82 |
| Guild + guild×year + guild×init + biparental + areainfra + distinfra | 3428.37 | 1.63 | 0.09 | 10 | 3408.36 |
| Guild + guild×year + guild×init + biparental | 3428.84 | 2.10 | 0.07 | 8 | 3412.83 |
| Guild + guild×year + guild×init + biparental + areainfra + distroads | 3428.98 | 2.24 | 0.07 | 10 | 3408.96 |
| Guild + guild×year + guild×init + distroads | 3429.30 | 2.56 | 0.06 | 9 | 3411.29 |
| Guild + guild×year + guild×init + biparental + guild*distinfra | 3429.81 | 3.07 | 0.04 | 11 | 3407.80 |
| Guild + guild×year + guild×init + biparental + areainfra + distroads + distinfra | 3429.81 | 3.07 | 0.04 | 11 | 3407.80 |
| Guild + guild×year + guild×init + biparental + guild×distroads | 3430.20 | 3.46 | 0.04 | 11 | 3408.19 |
| Guild + guild×year + guild×init + biparental + guild×areainfra | 3430.47 | 3.73 | 0.03 | 11 | 3408.46 |
| Guild + guild×year + biparental | 3430.74 | 4.00 | 0.03 | 6 | 3418.74 |
| Guild + guild×year + init + biparental | 3432.69 | 5.95 | 0.01 | 7 | 3418.68 |
| Guild + guild×init + biparental | 3432.75 | 6.01 | 0.01 | 6 | 3420.74 |
| (.) | 3505.69 | 78.95 | 0.00 | 1 | 3503.69 |

($ß_{Area\_infra}$ = -0.37, SE = 0.19, 95% CL -0.74 to 0.01), and the model improved by less than 2 $AIC_c$ units with the inclusion of the variable. Similarly, the variable distance to road is in the 3rd ranked model, 1.09 $AIC_c$ units from the top model; however, although nest survival appears to increase at a greater distance from roads, the beta estimate is also insignificant ($ß_{Dist\_roads}$ = 0.08, SE = 0.09, 95% CL -0.09 to 0.25).

Shorebirds with a biparental incubation strategy had a significantly higher daily nest survival than uniparental shorebirds ($ß_{Biparental}$ = 0.35, SE = 0.11, 95% CL 0.13–0.57; Fig. 2). Daily survival rate for uniparental and biparental shorebirds varied between years, and the difference varied between years (Post-hoc model; Fig. 3). Nest survival of Lapland longspur *Calcarius lapponicus,* the only passerine, decreased across the 17 years of the study ($ß_{Passerine×year}$ = -0.05, SE = 0.02, 95% CL -0.08 to -0.01; Fig. 2). Assuming a 12-day incubation period, nest success for passerines ranged from an estimated 0.59 in 2003 to 0.34 in 2019. There was no evidence of a change in shorebird nest survival across the years ($ß_{shorebird×year}$ = 0.01, SE = 0.01, 95% CL -0.01 to 0.03; Fig. 2). Waterfowl were not found on the plots in all years of the study and therefore were not included in the models of change over the years of the study. Nest survival decreased with later initiation date for shorebirds ($ß_{Shorbird×init}$ = -0.01, SE = 0.01, 95% CL -0.03 to 0.01) and it increased for passerines ($ß_{Passerine×init}$ = 0.02, SE = 0.01, 95% CL -0.002 to 0.04), but neither was significant.

Habitat variables did not occur in any models within 6.5 $AIC_c$ units from the top a priori model and did appear to be important ($ß_{Moist}$ = 0.01, SE = 0.17, 95% CL -0.33 to 0.34; $ß_{Wetland}$ = -0.14, SE = 0.17, 95% CL -0.484 to 0.195).

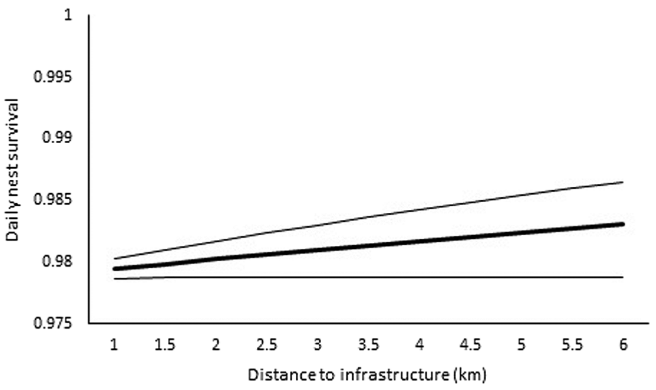

Figure 1. Daily survival rate (DSR) of biparental shorebird nests found at Prudhoe Bay, Alaska, 2003–2019 (lower and upper 95% confidence intervals). Survival is modeled for nests found in the ninth year of the study (2011), initiated on 4 June.

Figure 2. Daily survival rate (DSR) of biparental and uniparental shorebird nests and passerine nests found at Prudhoe Bay, Alaska, 2003–2019. Survival is modeled for nests initiated on 4 June, at the average distance from infrastructure for shorebirds (4.1 km) and passerines (3.9 km) and year is a linear variable.

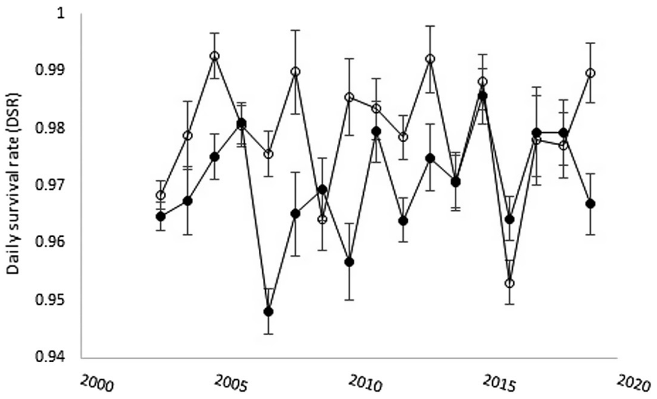

Figure 3. Daily survival rate (DSR) of biparental (open circles) and uniparental (closed circles) shorebird nests at Prudhoe Bay by year, Alaska, 2003–2019.

On average, shorebird density (52.8 nests km$^2$ ± 2.64) was much higher than either waterfowl (10.5 nests km$^2$ ± 1.32) or passerines (17.8 nests km$^2$ ± 1.13). Shorebird density did not change significantly over the course of the entire study (2003–2019; $F=0.01$, $R^2=0.06$, $p=0.92$; Fig. 4). Waterfowl density increased over the course of the study ($F=12.66$, $R^2=0.46$, $p=0.002$; Fig. 4). Passerine nest density did not vary significantly over the course of the study ($F=0.01$, $R^2=0.001$, $p=0.92$; Fig. 4).

Average snow cover during 7–15 June varied between years with 2018 standing out as much snowier than average (Fig. 5).

Waterfowl initiated incubation on average on 7 June (± 0.60; range 24 May to 6 July) between 2002 and 2019; however, samples sizes were zero in some years (sample size range: 0–22). Shorebird initiated incubation slightly later than waterfowl (average: June 13 ± 0.19; range 27 May to 20 July), while passerines initiated incubation at approximately the same time (average: 8 June ± 0.39; range 26 May to 5 July) between 2002 and 2019. Initiation dates did not vary significantly across the years of the study for shorebirds ($F=0.05$, $df=1224$, $p=0.83$) or passerines ($F=0.89$, $df=363$, $p=0.35$; Fig. 5); we did not include waterfowl due to small sample sizes.

The average number of predators detected during 30 min surveys at Prudhoe Bay increased between 2003 and 2019 ($F=10.64$, $p=0.005$; Fig. 6). This increase was driven by glaucous gulls ($F=28.30$, $p < 0.001$) and common ravens ($F=7.20$, $p=0.02$), as Arctic fox decreased ($F=8.34$, $p=0.01$), and jaegers did not vary significantly ($F=0.29$, $p=0.59$). Red fox were not detected during these surveys, although they are present in the oilfields.

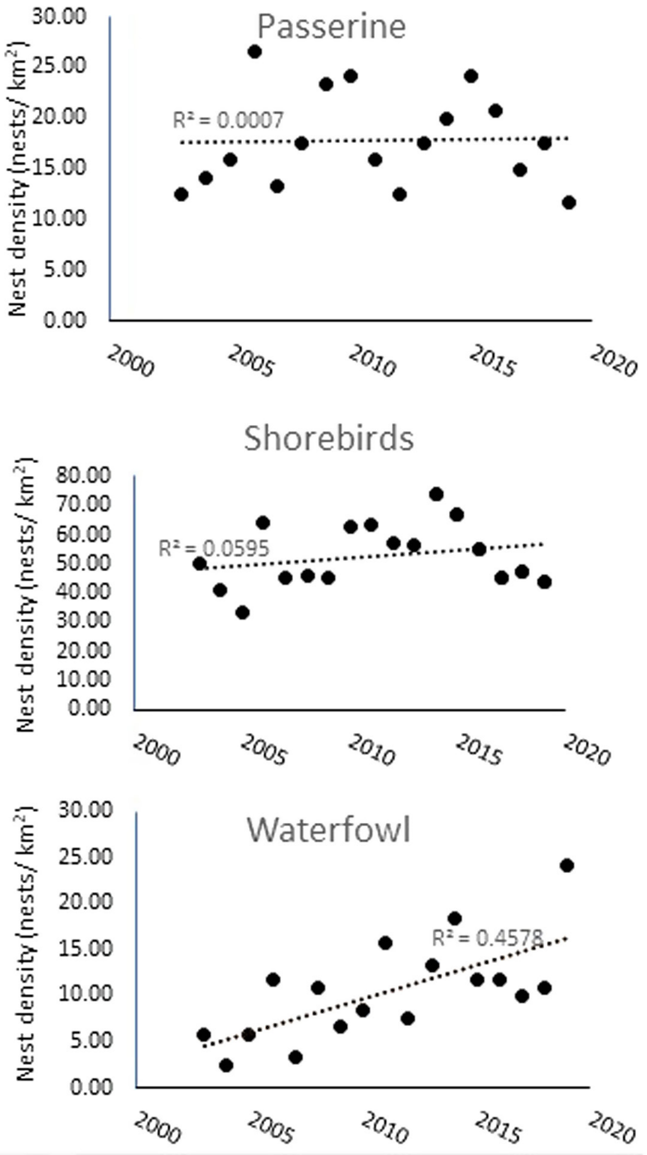

Figure 4. Nest density (nest km$^2$; ± SE) trends across time for shorebird, waterfowl, and passerines at Prudhoe Bay, Alaska, 2003–2019.

## Discussion

Long-term datasets are rare in the Arctic, but they are crucial for understanding impacts to breeding birds from both climate change and increasing anthropogenic activities and related infrastructure in the Arctic. We found that nest survival was lower for birds nesting closer to high-use infrastructure. Additionally, although insignificant, there was some indication that nest survival decreased in areas with a greater density of infrastructure, and that nest survival was higher at a greater distance from roads. Nest survival in the Arctic is always highly variable making it difficult to detect factors influencing nest survival without very large data sets, which may explain why previous studies have had mixed results. At the same site, Prudhoe Bay, Alaska, Bentzen et al (2017) did not find an effect of infrastructure on nest survival of shorebird nests, however, sample size was low ($n=186$). At a larger scale, comparing nest survival within the oilfields to sites well outside them, Liebezeit et al. (2009) showed that

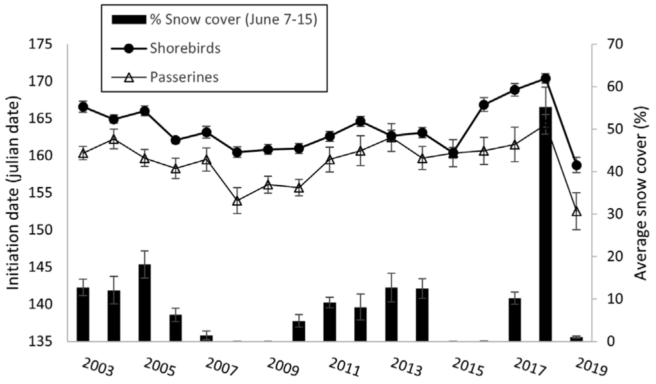

Figure 5. Average % snow cover on plots at Prudhoe Bay, Alaska, 7–15 June, and average nest initiation dates (± SE) for shorebirds and passerines at Prudhoe Bay, Alaska, 2003–2019.

human infrastructure had a negative effect on nest survival for passerines, but did not find a similar effect for shorebirds. However, within the oil fields, some individual species (red and red-necked phalaropes) had lower productivity closer to infrastructure, while others (semipalmated and pectoral sandpipers) did not (Liebezeit et al. 2009). Liebezeit et al. (2009) suggested that high variability in environmental conditions, nest survival, and predator numbers between sites and years may have contributed to these inconsistent results. Our results show that within Prudhoe Bay, with a very large dataset (n = 1874), nest survival of shorebirds, passerines, and waterfowl were significantly lower in areas closer to high-use infrastructure.

Spring conditions, described here as percent snow cover in the first week of June, are quite variable between years (Fig. 5), as has been shown previously (Meltofte et al. 2007, Schmidt et al. 2019). Although the earliest initiation dates for both passerines and shorebirds are in years with lower snow cover in early June, snow cover does not explain all the variability in nest initiation dates (Fig. 5). For example, 2016 stands out as having had very low snow cover but very late average initiation for shorebirds, and relatively late average initiation for passerines. Snow cover has been shown to be a driver in nest initiation previously at a study encompassing a larger geography (Liebezeit et al. 2014), but clearly there are other factors influencing the timing of initiation. Similarly, if you compare nest density (Fig. 4) with percent snow cover in June (Fig. 5), there are no correlations between low snow cover in some years (e.g. 2007, 2008, 2009, 2015, 2016, and 2019) and nest densities for any of the guilds. Shorebird nest survival (Fig. 3) varied dramatically between years, but it does appear that 'early' springs resulted in either higher or lower daily nest survival.

Saalfeld and Lanctot (2015, 2017) found that conservative species (typically biparental incubators) tended to have low variability in annual nest densities, and that there were some phenotypically flexible adjustments in most species to snowmelt, although opportunistic species (typically uniparental incubators) appeared to adjust better than conservative species. McGuire et al. (2020) showed that shorebirds of both strategies bred earlier and in higher numbers in early, warm springs relative to historic levels; opposite trends were observed in late springs, but nest survival was unrelated to spring type. Similarly, in this study, there is no clear pattern between 'early' and 'late' springs for either uniparental or biparental birds. Shorebirds with a biparental incubation strategy had a significantly higher daily nest survival than uniparental shorebirds but the difference between the two varied between years (Fig. 3). There was no clear pattern between years with low snow cover in the spring and those with higher snow cover on nest survival for either uniparental or biparental shorebirds (Fig. 3). 2016 stood out as an anomaly as uniparental shorebirds had higher nest survival than bi-parental shorebirds and nest survival was particularly low for bi-parental species. In other respects, 2016 appeared fairly typical (nest density, nest initiation, predator numbers). There were higher numbers of long-billed dowitchers (biparental incubators) than typical and lower numbers of pectoral sandpipers (uniparental; Table 1). Nest survival did not vary significantly across the years of the study for either strategy.

Evaluating trends in nest density highlights the importance of long-term data sets. There is no significant trend in

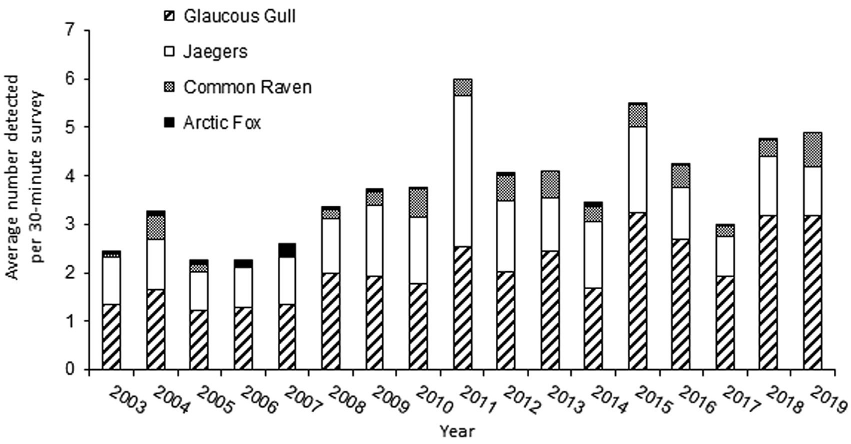

Figure 6. Average number of predators detected during 30 min surveys at Prudhoe Bay, Alaska, 2003–2019.

nest density for shorebirds or passerines between 2003 and 2019. However, during the last five to six years of the study (Fig. 4) there is a strong decline in numbers of nests from both guilds on our plots. We are unable to determine whether numbers will bounce back as they have in the past, but the high variability documented in Arctic studies (Liebezeit et al. 2009, McGuire et al. 2020) suggests it would not be unprecedented. The increase in geese breeding on our plots is mirrored across the Arctic Coastal Plain; since 1986 populations of both greater white-fronted geese and snow geese have increased (Wilson et al. 2018), possibly due to warmer temperatures in Arctic breeding grounds and climate-driven coastal subsidence (Hupp et al. 2017, Fondell et al. 2021).

Numbers of glaucous gulls and common ravens increased in the oilfields over the nearly two decades of this study. Garbage disposal and other sources of anthropogenic food (lunch sacks, feeding wildlife, etc.) improved drastically since the 1980s and there was no clear trend in abundance of gulls in the lagoons at Prudhoe Bay during the period of oilfield development (1978–2001; Noel et al. 2006), directly before our study began. Avian predator numbers within the oilfields have not been monitored systematically since then, and there is no information, other than this study, on whether their numbers decreased when practices were improved, and then increased over the past two decades. However, Wilson et al. (2018) found a long-term increase in glaucous gull numbers across the Arctic Coastal Plain (1992–2017), and Parrett et al. 2021 showed an increase in their annual growth rate in the Colville Delta, 2005–2020 (Parrett et al. 2021). Given that this mirrors the increase we see over nearly the same timeframe, the increase within Prudhoe Bay may not be due to anthropogenic food sources or perch sites within the oilfields. Wilson et al. (2018) found no increase in jaeger species between 1986 and 2017, supporting our results. However, they found no change in the population of commons ravens between 1992 and 2017 across the coastal plain (Wilson et al. 2018) while we did see an increase within the oilfields. Arctic fox decreased over the nearly two decades of the study, possibly due to displacement by red fox (Stickney et al. 2014). Although predator point counts underestimate the importance of Arctic fox as nest predators (Liebezeit and Zack 2008), presumably detection rates have not changed between 2003 and 2019, and previous studies have also shown a decline in Arctic fox numbers (Stickney et al. 2014), and an increase in red fox (Parrett et al. 2022).

Lapland longspur nest survival decreased across the 17 years of the study. They had an estimated nest success of 0.59 in the early years and 0.34 at the end of the study. A previous study in northern Alaska estimated nest survival at 51% (35.6–88.2% among years; Mayfield method) for Lapland longspur between 1967–1973 (Custer and Pitelka 1977). Although Custer and Pitelka (1977) used the Mayfield method, and we used generalized linear models implemented in Program MARK (White and Burnham 1999, Dinsmore et al. 2002) to evaluate daily nest survival, estimates of nest success are comparable. Our estimates of nest success by the mid-to late 2020s, are low compared to the seven-year average reported by Custer and Pitelka (1977), while our early estimates are very similar to theirs. Lapland longspur nest density varied between 11 and 27 nests km$^2$ between years over the course of the study but there was no obvious trend across time (Fig. 5). Custer and Pitelka (1977) also found high interannual variation in nest density at Point Barrow, Alaska, in the 1970s. Across North America, breeding density has decreased in southern parts of range, possibly due to climate change (Hussell and Montgomerie 2020). There is little information on the primary nest predators for passerines in the Arctic (but see Liebezeit and Zack 2008); however, the increase in avian nest predators may have disproportionately affected Lapland longspurs leading to the decline in nest survival for this species but not shorebirds, although this is speculative.

Lapland longspurs are the most abundant passerine in northern Alaska and are not a species of conservation concern (Hussell and Montgomerie 2020), but there has been little recent research into their breeding biology or demographics. To our knowledge there are no estimates of nest survival, nest density, or other demographic parameters from North American breeding areas since the 1970s, and no population trend analysis. However, passerines as a group have declined across North America drastically since the 1970s (Rosenberg et al. 2019). Also, in northern Alaska, a recent study by Boelman et al. (2015) suggested that increasing shrub dominance will diminish the habitat quality for Lapland longspurs, which breed in open tundra. We suggest that that they warrant focused study and concern given the decline in nest survival over the past two decades, increasing shrubs in much of their Arctic breeding grounds, and the fact that passerines have declined significantly since the 1970s.

In conclusion, we found no obvious pattern between timing of snow melt in early June and nest initiation, nest density, or shorebird daily nest survival. However, we found that nest survival was lower for birds nesting closer to high-use infrastructure. Lapland longspur nest survival decreased across the 17 years of the study and we recommend that this species receives attention to determine their population status. Shorebird nest density and nest survival did not vary significantly across the nearly two decades of this study except with respect to high-use infrastructure.

*Acknowledgements* – We thank BP Exploration (Alaska), Inc., for their generous financial assistance, encouragement, and help with our field logistics. Bill Streever, Christina Pohl, Tatyana Venegas, and Kyla Choquette were instrumental in the success of this project. We would also particularly like to recognize the dedication and hard work of all the many fieldwork leaders and technicians over the 17 years of the study. We also thank Kevin Fraley for GIS assistance and Steve Zack for initiating the project in 2003.

*Funding* – This work was funded by BP Exploration (Alaska), Inc. and the Wildlife Conservation Society.

*Conflict of interest* – The authors declare that this study received funding from BP Exploration (Alaska) Inc.

## Author contributions

**Rebecca McGuire**: Conceptualization (equal); Data curation (equal); Formal analysis (equal); Funding acquisition (equal); Investigation (equal); Methodology (equal); Project administration (equal); Validation (equal); Visualization (equal); Writing – original draft (equal); Writing – review and editing (equal). **Martin Robards**: Funding acquisition (equal); Project administration (equal); Writing – review and editing (equal). **Joseph Liebezeit**: Conceptualization (equal); Data curation (equal); Funding acquisition (equal); Methodology (equal); Project administration (equal); Writing – review and editing (equal).

## Transparent peer review

The peer review history for this article is available at https://publons.com/publon/10.1111/jav.03075.

## Data availability statement

Data are available from the Dryad Digital Repository: https://datadryad.org/stash/share/WI-0_PnsRXkn8TwIAyEnXU7BbtrMi9BvwQRlBTJMTBg (McGuire et al. 2023).

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