*Research Paper*

# Bird trends from long-term observation data at sites in the Hudson Bay Lowlands

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ABSTRACT. Monitoring population trends for bird species in the Arctic and sub-Arctic can be difficult and cost prohibitive. Breeding populations of birds in these remote locations may be changing and have garnered much attention regarding their conservation. We analyzed data from bird lists (daily species observations) collected while conducting other targeted research to estimate trends in the probability of observing species at two sub-Arctic study sites. For Akimiski Island, Nunavut (52 species) we estimated positive trends with high confidence (95% credible limits do not include 0) for 14 species and negative trends with high confidence for 18. For Burntpoint Creek, Ontario (46 species) we estimated that 12 species had positive trends with high confidence and 12 had negative trends with high confidence. More than 60% of species at each site matched population trends for at least one of three larger geographic scales that we compared estimates with; regional, provincial and national. We hypothesized that trends may be due to climate change effects or because of effects related to intensive herbivory from nesting geese. Trends for most of the species where we made specific predictions from these hypotheses agreed with those predicted; however, well-designed experiments are needed to conclusively determine the driving mechanisms of these trends. Collecting bird list data while simultaneously conducting other research is not nearly as time consuming or expensive as targeted breeding bird surveys (e.g., point counts, call counts, nest searching, etc.). This analysis method may be of utility for other remote field study sites where gaps in trend data exist.

## Tendances des oiseaux à partir d'observations faites depuis longtemps sur des sites dans les basses-terres de la baie d'Hudson

RÉSUMÉ. Le suivi de la tendance des populations d'oiseaux dans l'Arctique et le subarctique peut être difficile et son coût prohibitif. Les populations d'oiseaux nicheurs dans ces endroits éloignés sont peut-être en train de changer et leur conservation a suscité beaucoup d'attention. Nous avons analysé les données de listes d'oiseaux (observations quotidiennes d'espèces) recueillies lors d'autres recherches ciblées afin d'estimer la tendance de la probabilité d'observer des espèces à deux sites d'étude subarctiques. Pour l'île Akimiski, au Nunavut (52 espèces), nous avons calculé des tendances positives avec un niveau de confiance élevé (les limites crédibles à 95 % ne comprennent pas les 0) pour 14 espèces et des tendances négatives avec un niveau de confiance élevé pour 18 autres. Pour le ruisseau Burntpoint, en Ontario (46 espèces), nous avons déterminé que 12 espèces présentaient des tendances positives avec un niveau de confiance élevé et 12 des tendances négatives avec un niveau de confiance élevé. La tendance de plus de 60 % des espèces à chaque site correspondait aux tendances de populations d'au moins une des trois grandes échelles géographiques avec lesquelles nous avons comparé les résultats : régionale, provinciale et nationale. Nous avons émis l'hypothèse selon laquelle les tendances pouvaient être imputables aux effets du changement climatique ou aux effets liés au broutement intensif des oies nicheuses. Les tendances pour la plupart des espèces pour lesquelles nous avons fait des prédictions spécifiques à partir de ces hypothèses corroboraient celles prédites; toutefois, des protocoles expérimentaux bien conçus sont nécessaires pour déterminer de manière concluante les mécanismes moteurs de ces tendances. La collecte d'observations au moyen de listes d'oiseaux, tout en menant simultanément d'autres recherches, est loin d'être aussi longue et coûteuse que les suivis ciblés d'oiseaux nicheurs (p. ex. dénombrement par points d'écoute, relevé de chants, recherche de nids, etc.). Cette méthode d'analyse peut être utile pour d'autres sites d'étude éloignés où les tendances sont inconnues.

## INTRODUCTION

Monitoring of population trends for bird species occurs through several government and non-government sponsored programs and at multiple spatial scales in North America (e.g., ECCC 2019, Sauer et al. 2004). Available evidence indicates that populations of most waterfowl species have increased and those of most shorebird species have declined (Rosenberg et al. 2019, Smith et al. 2020). Populations of most landbird species have been mostly stable but some have declined in recent decades, with some showing large declines (NABCI Canada 2019, Rosenberg et al. 2019, Smith et al. 2020).

Population indices from the breeding range are preferred for informing management and conservation as observations during migration (Ross et al. 2012) or on wintering ranges (Newbold et al. 2012) may reflect a mixture of animals from reproductively isolated populations. Indices from winter range counts may also be subject to overdispersion issues (Richards 2007) due to

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clumping and bias from uneven distribution and survey coverage. Species whose breeding ranges are not easily accessed are generally not well monitored, particularly in more northerly regions (ECCC 2018, Roy et al. 2019). Exceptions may include species that are the target of research or game species whose numbers are monitored for harvest management purposes, (e.g., Canada Geese (*Branta canadensis*): Leafloor et al. 1996 and Snow Geese (*Anser caerulescens*): Kerbes et al. 2014). Other programs provide long-term datasets but may not have enough geographic coverage to adequately survey all species breeding ranges (e.g., Sauer et al. 2013). Citizen science-based programs and incidental observations have proven to contribute valuable knowledge for tracking status and trends in populations, especially where standardized surveys or management-driven monitoring programs are lacking (Crewe et al. 2008, Bonter and Cooper 2012, Callaghan and Gawlik 2015, Walker and Taylor 2017). However, these programs rarely target those breeding populations that are in difficult to access areas such as the Arctic and sub-Arctic where status of breeding birds has received increasing management and conservation attention.

For species in decline, population change may be driven by factors on the breeding range, during migration, and in wintering areas. The drivers of change may manifest through negative impacts on juvenile and adult survival or through impacting reproductive success and recruitment. In Arctic and sub-Arctic regions, bird population change may be occurring through the direct or indirect effects of climate change (Butler 2000, Ims et al. 2019). Similarly, herbivory by overabundant geese (Abraham and Jefferies 1997, Flemming et al. 2016, Buij et al. 2017, Flemming et al. 2019) may also be causing bird population change due to the change these herbivores induce on their habitat. These two phenomena may independently cause change in bird populations in this region or they may interact resulting in little habitat change (Post 2013, Biuw et al. 2014) or in compounded change. Climate change effects are predicted to be most pronounced at higher latitudes (ACIA 2005) and are expected to impact the sub-Arctic and Arctic regions first (IPCC 2018). Habitat change in these regions has been observed including the encroachment of shrubs in tundra habitats (Post et al. 2019), drying of wetland habitats (Roach et al. 2011), or mismatch for some species between the timing of hatch and the availability of their primary food sources for young (McKinnon et al. 2012, Brook et al. 2015). Further, most Arctic and sub-Arctic breeding birds occupy intermediate trophic levels and may be more vulnerable to climate change effects than species at other trophic levels because they may be subjected to both top-down and bottom-up ecological change (Weiser et al. 2018a). There is evidence that Arctic breeding shorebirds are exhibiting declines worldwide (Studds et al. 2017, Weiser et al. 2020) and opportunities to enhance monitoring and trend assessments are needed to inform conservation.

Here we used list analyses methods (Roberts et al. 2017, Szabo et al. 2010) to estimate change in long-term (17 to 23 years) bird list datasets for species at two sub-Arctic study sites. We only included species where we deemed observation effort was consistent through time to ensure that survey methods did not result in the detection of false trends. We hypothesized that any species-specific trends we detected would be consistent with trends reported from larger geographic scale assessments for those species. Trends that did not match those from larger scale

assessments may be due to more localized drivers at each study site; drivers that are not widespread or acting at the larger geographic scales. We hypothesized that habitat damage due to localized herbivory at the more southerly study site (Kotanen and Abraham 2013) could potentially compound the broader scale environmental impacts on habitat that could be driving some of the trends we observed. Further, we predicted that the presence of herbivores (Canada and Lesser Snow Geese) attracts both avian (American Crow (*Corvus brachyrhynchos*), Common Raven (*Corvus corax*)) and mammalian predators that may increase predation of eggs for other species, especially breeding ducks (Mallard (*Anas platyrhynchos*), American Black Duck (*A. rubripes*), Northern Pintail (*A. acuta*), and Green-winged Teal (*A. crecca*)) and breeding shorebirds (Semipalmated plover (*Charadrius semipalmatus*), Short-billed Dowitcher (*Limnodromus griseus*), and Least Sandpiper (*Calidrus minutilla*) thus causing negative trends for those species. We also predicted that, through encroachment of shrubs in open graminoid areas at the more northerly site (predominantly low-tundra habitat), species that would benefit from increasing shrub habitat would likely increase, including American Tree Sparrow (*Spizella arborea*), Lincoln's Sparrow (*Melospiza lincolnii*), Song Sparrow (*M. melodia*), and White-throated Sparrow (*Zonotrichia albicollis*).

## METHODS

Daily observations of all bird species were performed at Akimiski Island, Nunavut (N53.013, W81.321) from 1993 to 2015 (N = 23 yr) and at Burntpoint Creek, Ontario (N55.241, W84.318) from 2001 to 2005 and 2007 to 2018 (N = 17 yr). Although data were collected for variable periods annually, observations used in analysis were restricted to those between 21 May and 4 August at both locations.

Observations were done by researchers (from 4 to 20 per day) with varying bird identification experience (from minimal to advanced) while performing fieldwork on long-term Canada Goose, Lesser Snow Goose (Leafloor et al. 2000), and shorebird research projects (Nguyen et al. 2003, Pollock et al. 2012, Weiser et al. 2020). While in the field, observers recorded all bird species encountered by sight and sound, according to their abilities. At the end of each day, a composite list of bird observations was compiled from all observers.

We restricted data to include those species known to be within their breeding range based on available literature and documented evidence or where we had confirmation of breeding evidence at our local sites (Cadman et al. 2007, MNRF unpublished data). We excluded locally non-breeding and transient species that often occurred at our sites (e.g., Black-bellied Plover (*Pluvialis squatarola*)) but whose presence and abundance in any year may be affected more by weather than true population change. To reduce the potential effect of bias on trends, we also removed species observed in less than 20% of years and on less than 3% of observation days because of known issues with trend analysis. Szabo et al. (2010) found that rare species were reported at a rate unaccounted for by list length. Thus, we assumed that observations of these rare species as we defined them were likely more by chance than associated with effort and change over time. Some species that were the focus of intense research were also excluded due to the potential for a false trend because of increased observation effort for a limited number of years during the time

series; however, we did model one such species, Smith's Longspur (*Calcarius pictus*) at Burntpoint Creek, but excluded a single year (2007) when a Smith's Longspur project had occurred. We also removed extremely abundant species (observed >85% of observation days in each year) as any survey (day) without an observation was likely an anomaly due to lack of field work in a portion of the study area on that day (e.g., Lesser Snow Goose, where the colony was not visited daily). This would not necessarily result in a bias; however, we would still not likely be able to reliably detect a trend for these very abundant species. Species that bred in habitats not consistently surveyed (e.g., inland sedge meadows on Akimiski Island) were also excluded because of the same potential for false trends resulting from inconsistent surveying. The result of our exclusions and restrictions is a subset of species for which we have highest confidence in consistency of status (e.g., breeding) and observation effort (e.g., unbiased by numerical abundance or from increased detection by targeted research).

The geographic area covered at each site (proportion of typical study area) varied daily but was similar between years. We used a list length analysis to compensate for effort differences, which assumes the number of species (list length) observed in each survey (in our case, a day) increases with increasing effort (see Szabo et al. 2010). This also compensated for observer skill as we equated longer lists with observers with greater bird identification experience. Logistic regression was used to model the probability of a species presence as a function of the total number of species (the list length) and year surveyed (Link et al. 2006, Szabo et al. 2010). We assessed three additive trend models (all included a year effect): 1) a prevalence model (intercept + year), 2) an effort model (intercept + year + log(list length)), and 3) a date/effort model (intercept + year + log(list length) + day of year). Species detection probability may be density dependent which is a known potential complication for this method (Szabo et al. 2010); however, the mix of skills and number of observers contributing to daily lists mediates potential bias due to detection rate change. We included a model with day of year to account for any seasonal differences there might be for species detection probability (Walker and Taylor 2017). For each species, we estimated parameter coefficients for the three models in a Bayesian logistic model regression framework.

We fitted models using Markov Chain Monte Carlo (MCMC) simulation using uninformed priors (mean of 0 and standard deviation of 10000) with 80000 iterations and 20000 burn-in iterations and a thinning rate of five. As in Szabo et al. (2010), input variables were centered with a mean of 0 but were not rescaled to units of standard deviation to help with interpretation. We implemented JAGS 4.3.0 (Gelfand and Smith 1990, Plummer 2013) from program R (R Development Core Team 2018) using the package jagsUI (Kellner 2018). To determine whether each species models converged, we calculated Geweke z-scores (Geweke 1992) for each coefficient using R package ggmcmc (Fernández-i-Marín 2020) and also inspected plots of their traces. Any parameter coefficient with a Geweke z-score greater than 2.0 or less than -2.0 for any of three chains was deemed to have poor convergence and models were run again until there was acceptable convergence. We made further convergence assessment using the Gelman-Rubin diagnostic R (Gelman et al. 2004). Convergence was achieved for parameters (all R < 1.1) using three chains. For each species, we evaluated model fit by measuring the deviance from the null model

that was explained by the fitted model using Deviance Information Criterion (DIC: Spiegelhalter et al. 2002). Models with ΔDIC < 2 have similar strong support and we used the rules of parsimony to select models from which to make an inference. We present the results for the year effect regardless of whether the 95% credible limits included zero (a low confidence trend) and categorized trends where 95% credible limits did not include zero as a high confidence trend.

We compared our trend estimates (probability of observing each species) for each study site with trends from larger geographic scales. We directly compared our estimates of trend with those for the Hudson Bay Lowlands (which encompasses both study sites) reported in the second breeding bird atlas of Ontario (Cadman et al. 2007). In the second Ontario breeding bird atlas, probability of observation was compared between the first (1981-1985) and second (2001-2005) atlases and statistical significance (P < 0.1) of change was reported (Cadman et al. 2007). At the Ontario provincial scale, we compared our results with abundance trends estimated from an analysis of the Breeding Bird Survey of North America (Smith and Edwards 2020). We used custom trends calculated from the annual Canadian Wildlife Service analyses of the Breeding Bird Survey (Smith et al. 2020) for time intervals matching those for our trends from each study area. At the national scale, we used abundance trends from the Breeding Bird Survey of North America (Smith et al. 2020) also estimated for the same time periods as for each study area. In all cases, if the credible limit included zero for the year effect, we had little confidence in the estimated trend (a low confidence trend). To assess the consistency of trends between study sites and among scales, we estimated the correlation using a Pearson correlation including 95% confidence limits (significance was determined at alpha < 0.05).

## RESULTS
In total, 811 and 342 surveys (survey-days) were used for analysis for Akimiski Island and Burntpoint Creek, respectively. After screening species due to rarity, inconsistent surveying, outside of breeding range, and super abundance, we found the trend model (estimating list length and year effects) or the day model (estimating list length, day of year and year effects) to be the most parsimonious for most species. We compared trend (year effect covariate) results for 52 species observed at Akimiski Island (Fig. 1, Table 1) and 46 at the Burntpoint Creek study area (Fig. 2, Table 2). For Akimiski Island, we found 32 species with trends that did not include 0 within the 95% credible limits; 14 positive and 18 negative. For Burntpoint Creek, we found 24 species with trends that did not include 0 within the 95% credible limits; 12 positive and 12 negative. We identified trends for 33 (63.5%) and 31 (67.4%) species for Akimiski Island and Burntpoint Creek, respectively, that matched the direction of trends for at least one of the three larger geographic scales. Conversely, 19 (36.5%) species observed at Akimiski Island and 15 (32.6%) for the Burntpoint Creek study area (three in common) did not match the direction of trends for at least one of three larger geographic scales. There was a positive but insignificant correlation (0.20, 95% CL = -0.182 to 0.526, P = 0.31) between trends for species common to both study areas (N = 29, Figure 3); however, where there were high confident trends for species in common at both sites (N = 10), five species had the same trend direction (American

**Table 1**. Mean trends of the probability of observation (and 95% credible limits) estimated from daily bird count data for species surveyed annually at the Akimiski Island study site in the sub-Arctic, 1993 to 2015. Trend comparisons are with those at the 1) regional scale; the Hudson Bay Lowlands (HBL) Breeding Bird Atlas trend from estimating the probability of observation between the first (1981 to 1985) and second (2001 to 2005) Atlases in Ontario, the 2) provincial scale using trends in abundance from the North American Breeding Bird Survey (1993 to 2015) and with the 3) Breeding Bird Survey at the national scale (1993 to 2015).

| ORDER and common name | Scientific name | Akimiski Island | HBL Atlas[†] | Ontario BBS[‡] | Canada BBS[§] |
|---|---|---|---|---|---|
| **ANSERIFORMES** | | | | | |
| Mallard | *Anas platyrhynchos* | 0.072 (0.031– 0.113)* | No trend | -0.414 | -0.365 |
| American Black Duck | *Anas rubripes* | 0.004 (-0.029– 0.037) | No trend | -4.994* | -2.634 |
| Northern Shoveler | *Spatula clypeata* | 0.007 (-0.035– 0.049) | No trend | | |
| Northern Pintail | *Anas acuta* | -0.028 (-0.068– 0.013) | No trend | -4.298 | -0.953 |
| Green-winged Teal | *Anas crecca* | -0.101 (-0.141– -0.061)* | No trend | -1.374 | 0.12 |
| American Wigeon | *Mareca americana* | -0.126 (-0.177– -0.077)* | No trend | -1.766 | 0.298 |
| Common Merganser | *Mergus merganser* | -0.027 (-0.069– 0.016) | No trend | -0.05 | -2.126 |
| Red-breasted Merganser | *Mergus serrator* | -0.03 (-0.079– 0.019) | No trend | -3.555* | -3.343 |
| Hooded Merganser | *Lophodytes cucullatus* | -0.015 (-0.097– 0.069) | No trend | 3.255* | 2.994* |
| **GAVIIFORMES** | | | | | |
| Pacific Loon | *Gavia pacifica* | -0.194 (-0.282– -0.11)* | No trend | | |
| Common Loon | *Gavia immer* | 0.045 (0.013– 0.078)* | No trend | -0.408 | 0.694 |
| **PELECANIFORMES** | | | | | |
| American Bittern | *Botaurus lentiginosus* | 0.032 (-0.006– 0.071) | No trend | -0.546 | 0.746 |
| **ACCIPITRIFORMES** | | | | | |
| Northern Harrier | *Circus hudsonius* | 0.076 (0.041– 0.11)* | Increase | -2.187* | -2.326* |
| **STRIGIFORMES** | | | | | |
| Short-eared Owl | *Asio flammeus* | 0.072 (0.029– 0.115)* | Increase | | |
| **GRUIFORMES** | | | | | |
| Sandhill Crane | *Antigone canadensis* | 0.065 (0.032– 0.098)* | Increase | 11.044* | 6.182* |
| **CHARADRIIFORMES** | | | | | |
| Killdeer | *Charadrius vociferous* | 0.083 (0.041– 0.127)* | Increase | -4.433* | -2.202* |
| Semipalmated Plover | *Charadrius semipalmatus* | -0.093 (-0.133– -0.053)* | No trend | | |
| Greater Yellowlegs | *Tringa melanoleuca* | 0.152 (0.112– 0.194)* | Increase | 0.645 | 0.997 |
| Lesser Yellowlegs | *Tringa flavipes* | 0.008 (-0.027– 0.043) | No trend | | |
| Hudsonian Godwit | *Limosa haemastica* | -0.067 (-0.102– -0.032)* | No trend | | |
| Marbled Godwit | *Limosa fedoa* | -0.022 (-0.061– 0.015) | No trend | | |
| Least Sandpiper | *Calidris minutilla* | -0.047 (-0.083– -0.012)* | No trend | | |
| Short-billed Dowitcher | *Limnodromus griseus* | -0.045 (-0.087– -0.002)* | No trend | | |
| Wilson's Snipe | *Gallinago delicata* | -0.064 (-0.101– -0.027)* | Increase | -0.494 | 0.938 |
| Bonaparte's Gull | *Chroicocephalus philadelphia* | -0.089 (-0.126– -0.052)* | No trend | 0.995 | -1.874 |
| Arctic Tern | *Sterna paradisaea* | -0.052 (-0.086– -0.018)* | No trend | | |
| Caspian Tern | *Hydroprogne caspia* | 0.041 (-0.037– 0.121) | No trend | 0.052 | -0.325 |
| **PICIFORMES** | | | | | |
| Northern Flicker | *Colaptes auratus* | -0.054 (-0.089– -0.019)* | Increase | -0.43 | -1.033* |
| **PASSERIFORMES** | | | | | |
| Alder Flycatcher | *Empidonax alnorum* | 0.19 (0.119– 0.267)* | Increase | -0.556 | -0.69 |
| Canada Jay | *Perisoreus canadensis* | 0.042 (0.002– 0.083)* | Increase | 0.753 | -0.257 |
| Common Raven | *Corvus corax* | 0.054 (0.017– 0.092)* | Increase | -0.854 | 0.892* |
| American Crow | *Corvus brachyrhynchos* | 0.238 (0.195– 0.283)* | Increase | 0.161 | -0.311* |
| Horned Lark | *Eremophila alpestris* | -0.143 (-0.184– -0.103)* | No trend | -5.093* | -4.731* |
| Boreal Chickadee | *Poecile hudsonicus* | -0.072 (-0.136– -0.007)* | No trend | -0.789 | -0.454 |
| Ruby-crowned Kinglet | *Regulus calendula* | 0.052 (-0.002– 0.107) | Increase | 0.572 | 1.582* |
| Hermit Thrush | *Catharus guttatus* | 0.034 (0.002– 0.066)* | Increase | 0.446 | -1.184 |
| American Robin | *Turdus migratorius* | 0.004 (-0.032– 0.04) | Increase | -1.417* | -0.456* |
| Tennessee Warbler | *Leiothlypis peregrina* | 0.021 (-0.022– 0.064) | No trend | -7.476* | -1.136 |
| Orange-crowned Warbler | *Leiothlypis celata* | 0.018 (-0.04– 0.077) | Increase | 0.692 | -1.593* |
| Yellow Warbler | *Setophaga petechia* | 0.019 (-0.013– 0.052) | Increase | -2.189* | -1.047* |
| Yellow-rumped Warbler | *Setophaga coronata* | 0.005 (-0.035– 0.045) | Increase | -0.386 | -0.868 |
| Wilson's Warbler | *Cardellina pusilla* | 0.055 (-0.04– 0.155) | Increase | 0.135 | 0.281 |
| American Tree Sparrow | *Spizelloides arborea* | -0.129 (-0.196– -0.063)* | No trend | | |
| LeConte's Sparrow | *Ammospiza leconteii* | 0.08 (-0.021– 0.186) | Increase | -3.497* | -3.749* |
| Savannah Sparrow | *Passerculus sandwichensis* | -0.047 (-0.092– -0.002)* | Increase | -3.253* | -1.412* |
| Fox Sparrow | *Passerella iliaca* | -0.002 (-0.038– 0.033) | Increase | -1.973 | 0.254 |
| Lincoln's Sparrow | *Melospiza lincolnii* | -0.053 (-0.09– -0.017)* | Increase | -1.935* | -1.176 |
| Swamp Sparrow | *Melospiza georgiana* | -0.064 (-0.113– -0.016)* | Increase | 1.725* | 0.538 |
| Song Sparrow | *Melospiza melodia* | 0.087 (0.004– 0.172)* | Increase | -0.892* | -0.534* |
| White-crowned Sparrow | *Zonotrichia leucophrys* | 0.103 (0.066– 0.142)* | Increase | | |
| Dark-eyed Junco | *Junco hyemalis* | -0.008 (-0.042– 0.027) | Increase | -1.769 | -0.667 |
| Common Redpoll | *Acanthis flammea* | -0.12 (-0.162– -0.08)* | No trend | -2.029 | -1.743 |

[†]Regional scale Hudson Bay Lowlands trends from the Ontario Breeding Bird Atlas (Cadman et al. 2007)
[‡]Ontario provincial scale trends from the North American Breeding Bird Survey for the time period 1993 to 2015 (Smith and Edwards 2020)
[§]Canada national scale trends from the North American Breeding Bird Survey for the time period 1993 to 2015 (Smith and Edwards 2020)
*Indicates a species had a trend where zero was not within the 95% credible limits

**Fig. 1**. Mean coefficients for the effect of year (trend) on the probability of observation for bird species at the Akimiski Island, Nunavut study site based on daily bird count data from 1993 to 2015. Values are logit-transformed probability of observation and bars are 95% credible limits.



**Fig. 2**. Mean coefficients for the effect of year (trend) on the probability of observation for bird species trends at the Burntpoint Creek, Ontario study site based on daily bird count data from 2001 to 2018 (excluding 2006). Values are logit-transformed probability of observation and bars are 95% credible limits.
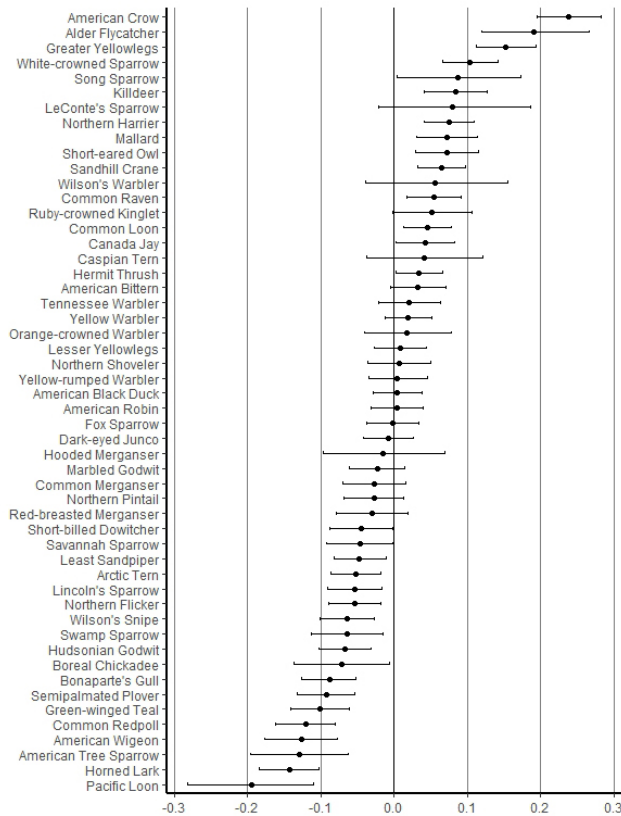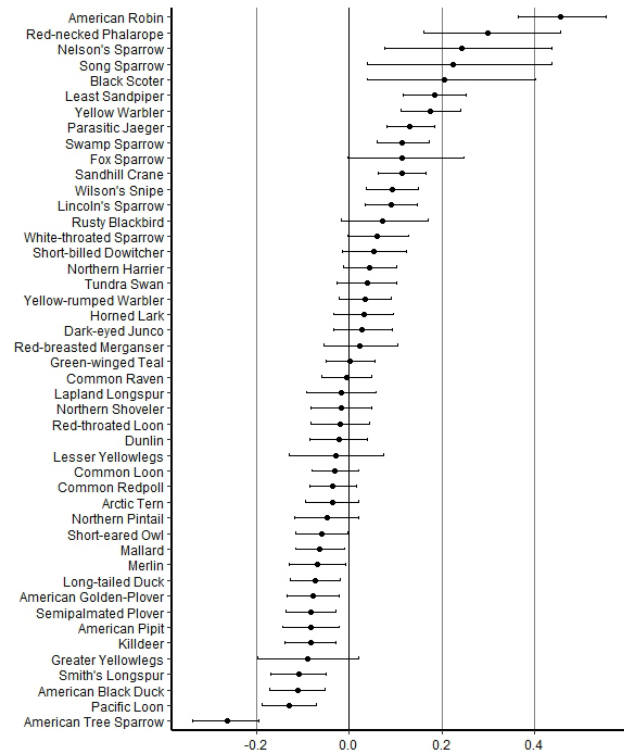


Tree Sparrow, Pacific Loon (*Gavia pacifica*), Sandhill Crane (*Antigone canadensis*), Semipalmated Plover, and Song Sparrow) and five had opposite trend directions (Killdeer (*C. vociferous*), Least Sandpiper, Mallard, Short Eared Owl (*Asio flammeus*) and Swamp Sparrow (*M. georgiana*)). There were also positive but insignificant correlations between Akimiski Island and the provincial Breeding Bird Survey trends (0.19, 95%CL = -0.128 to 0.474, P = 0.24) and the national Breeding Bird Survey trends (0.17, 95% CL = -0.114 to 0.428, P = 0.24) (Figure 4). Similarly, there were positive but insignificant correlations between Burntpoint Creek and the provincial Breeding Bird Survey trends (0.16, 95% CL = -0.249 to 0.524 P = 0.44) and the national Breeding Bird Survey trends (0.004, 95% CL = -0.308 to 0.314, P = 0.98).

## DISCUSSION

The trend that we observed for most species matched that reported for at least one of the larger geographic scales (i.e., region, provincial or national) suggesting that the processes causing most trends that we detected at a study site scale might be part of trends occurring at the larger scales or the cause may occur outside the breeding seasons (i.e., during migration or on wintering areas for

**Fig. 3**. A scatter plot of mean probability of observation (trends) and 95% credible limits bars for comparing species observed at the Akimiski Island (1993 to 2015) and Burntpoint Creek (2001 to 2018, excluding 2006) study areas based on daily bird counts. The trend line represents the Pearson correlation (slope) between the sites.
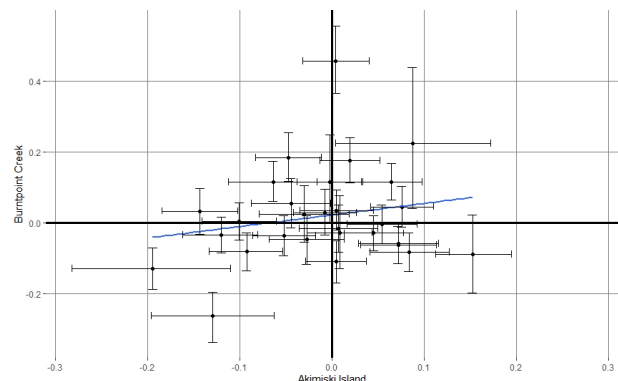
**Table 2**. Mean trends of the probability of observation (and 95% credible limits) estimated from daily bird count data for species surveyed annually at the Burntpoint Creek study site in the sub-Arctic, 2001 to 2018 (excluding 2006). Trend comparisons are with those at the 1) regional scale; the Hudson Bay Lowlands (HBL) Breeding Bird Atlas trend from estimating the probability of observation between the first (1981 to 1985) and second (2001 to 2005) Atlases in Ontario, the 2) provincial scale using trends in abundance from the North American Breeding Bird Survey (2001 to 2018) and with the 3) Breeding Bird Survey at the national scale (2001 to 2018).

| **ORDER** and common name | Scientific name | Burntpoint Creek | HBL Atlas[†] | Ontario BBS[‡] | Canada BBS[§] |
|---|---|---|---|---|---|
| **ANSERIFORMES** | | | | | |
| Tundra Swan | *Cygnus columbianus* | 0.038 (-0.025– 0.103) | Increase | | 0.695 |
| Mallard | *Anas platyrhynchos* | -0.063 (-0.117– -0.011)* | No trend | -0.94 | -0.273 |
| American Black Duck | *Anas rubripes* | -0.111 (-0.172– -0.052)* | No trend | -6.352* | -3.312 |
| Northern Shoveler | *Spatula clypeata* | -0.018 (-0.084– 0.049) | No trend | | 3.522* |
| Northern Pintail | *Anas acuta* | -0.048 (-0.118– 0.02) | No trend | -3.002 | 1.721 |
| Green-winged Teal | *Anas crecca* | 0.002 (-0.049– 0.055) | No trend | -1.625 | 0.717 |
| Black Scoter | *Melanitta americana* | 0.206 (0.038– 0.402)* | No trend | | |
| Long-tailed Duck | *Clangula hyemalis* | -0.073 (-0.127– -0.019)* | No trend | | -0.596 |
| Red-breasted Merganser | *Mergus serrator* | 0.023 (-0.055– 0.104) | No trend | -3.411 | -3.149 |
| **GAVIIFORMES** | | | | | |
| Red-throated Loon | *Gavia stellate* | -0.02 (-0.084– 0.043) | | | 0.345 |
| Pacific Loon | *Gavia pacifica* | -0.129 (-0.189– -0.072)* | No trend | | -0.07 |
| Common Loon | *Gavia immer* | -0.03 (-0.08– 0.02) | No trend | -1.556* | 0.201 |
| **ACCIPITRIFORMES** | | | | | |
| Northern Harrier | *Circus hudsonius* | 0.044 (-0.013– 0.103) | Increase | -1.873 | -1.549* |
| **STRIGIFORMES** | | | | | |
| Short-eared Owl | *Asio flammeus* | -0.06 (-0.115– -0.004)* | Increase | | -2.632* |
| **FALCONIFORMES** | | | | | |
| Merlin | *Falco columbarius* | -0.068 (-0.129– -0.008)* | Increase | -0.704 | 0.527 |
| **GRUIFORMES** | | | | | |
| Sandhill Crane | *Antigone canadensis* | 0.114 (0.063– 0.166)* | Increase | 9.22* | 6.96* |
| **CHARADRIIFORMES** | | | | | |
| Killdeer | *Charadrius vociferous* | -0.083 (-0.138– -0.028)* | Increase | -4.028* | -1.639* |
| Semipalmated Plover | *Charadrius semipalmatus* | -0.082 (-0.136– -0.03)* | No trend | | -2.292 |
| American Golden-Plover | *Pluvialis dominica* | -0.078 (-0.134– -0.022)* | No trend | | |
| Greater Yellowlegs | *Tringa melanoleuca* | -0.09 (-0.198– 0.02) | Increase | 0.469 | 1.048 |
| Lesser Yellowlegs | *Tringa flavipes* | -0.029 (-0.129– 0.075) | No trend | | -2.114 |
| Red-necked Phalarope | *Phalaropus lobatus* | 0.298 (0.162– 0.456)* | No trend | | -4.456 |
| Least Sandpiper | *Calidris minutilla* | 0.183 (0.117– 0.253)* | No trend | | -0.483 |
| Dunlin | *Calidris alpina* | -0.022 (-0.086– 0.04) | No trend | | |
| Short-billed Dowitcher | *Limnodromus griseus* | 0.053 (-0.015– 0.124) | No trend | | |
| Wilson's Snipe | *Gallinago delicata* | 0.093 (0.037– 0.149)* | Increase | -0.848 | 0.567 |
| Parasitic Jaeger | *Stercorarius parasiticus* | 0.131 (0.08– 0.184)* | Decrease | | |
| Arctic Tern | *Sterna paradisaea* | -0.036 (-0.094– 0.02) | No trend | | -2.733 |
| **PASSERIFORMES** | | | | | |
| Rusty Blackbird | *Euphagus carolinus* | 0.073 (-0.017– 0.171) | Increase | -0.692 | 0.082 |
| Common Raven | *Corvus corax* | -0.005 (-0.059– 0.049) | Increase | -0.672 | 0.526 |
| Horned Lark | *Eremophila alpestris* | 0.032 (-0.033– 0.096) | No trend | -4.978* | -3.988* |
| American Robin | *Turdus migratorius* | 0.455 (0.364– 0.555)* | Increase | -2.355* | -0.605* |
| American Pipit | *Anthus rubescens* | -0.083 (-0.144– -0.023)* | No trend | | -3.44 |
| Yellow Warbler | *Setophaga petechia* | 0.175 (0.113– 0.24)* | Increase | -2.963* | -1.6* |
| Yellow-rumped Warbler | *Setophaga coronata* | 0.034 (-0.021– 0.091) | Increase | -2.146* | -1.751* |
| American Tree Sparrow | *Spizelloides arborea* | -0.264 (-0.338– -0.196)* | No trend | | -1.987 |
| Nelson's Sparrow | *Ammospiza nelsoni* | 0.243 (0.076– 0.439)* | No trend | | -2.963* |
| Fox Sparrow | *Passerella iliaca* | 0.114 (-0.003– 0.248) | Increase | -3.587 | -0.914 |
| Lincoln's Sparrow | *Melospiza lincolnii* | 0.09 (0.035– 0.147)* | Increase | -2.091* | -0.359 |
| Swamp Sparrow | *Melospiza georgiana* | 0.115 (0.06– 0.173)* | Increase | 1.055 | 0.244 |
| Song Sparrow | *Melospiza melodia* | 0.224 (0.039– 0.437)* | Increase | -0.844* | -0.085 |
| White-throated Sparrow | *Zonotrichia albicollis* | 0.061 (-0.002– 0.128) | No trend | -3.005* | -2.165* |
| Dark-eyed Junco | *Junco hyemalis* | 0.028 (-0.035– 0.093) | Increase | -3.555* | -0.628 |
| Lapland Longspur | *Calcarius lapponicus* | -0.017 (-0.092– 0.057) | No trend | | 0.6 |
| Smith's Longspur[ǀ] | *Calcarius pictus* | -0.108 (-0.169– -0.051)* | | | |
| Common Redpoll | *Acanthis flammea* | -0.036 (-0.086– 0.015) | No trend | -2.187 | -1.62 |

[†]Regional scale Hudson Bay Lowlands trends from the Ontario Breeding Bird Atlas (Cadman et al. 2007)
[‡]Ontario provincial scale trends from the North American Breeding Bird Survey for the time period 2001 to 2018 (Smith et al. 2020)
[§]Canada scale trends from the North American Breeding Bird Survey for the time period 2001 to 2018 (Smith et al. 2020)
*Indicates a species had a trend where zero was not within the 95% credible limits
[ǀ]Data for 2007 was not included for Smith's Longspur because of a species-specific project in that year that may cause a bias in trend estimates

migratory species). The only geographic coverage overlap in data collection with our study sites was that for the regional scale; the Hudson Bay Lowlands (Cadman et al. 2007). Data collection at the provincial and national scales of the Breeding Bird Survey (Smith et al. 2020) do not overlap geographically with our study area though we still made the trend comparisons because of species range overlap. Reasons for differences in trend direction that we observed may also include environmental or ecological

**Fig. 4**. Scatter plots of the mean probability of observation (trend) and 95% credible limits bars comparing species observed at the Akimiski Island study site (1993 to 2015) with trends in abundance from the A) Ontario Breeding Bird Survey and the C) national (Canada) Breeding Bird Survey. Similar comparisons were made for the mean probability of observation (trend) and 95% credible limits bars comparing species observed at the Burntpoint Creek study site (2001 to 2018, excluding2006) with B) the Ontario Breeding Bird Survey trends and D) the Canada Breeding Survey trends for the same time period (2001 to 2018). Trend lines are Pearson correlation (slope).
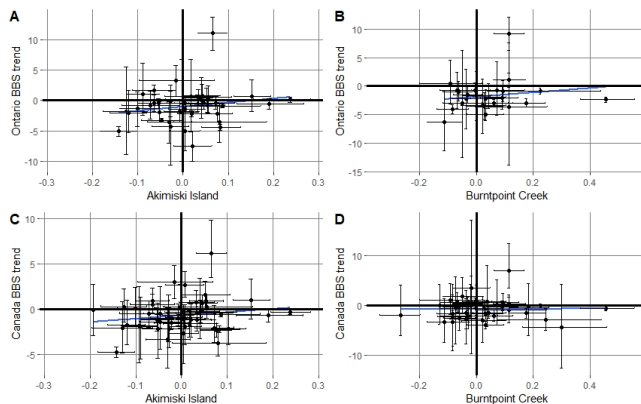


factors unique to sites near large waterfowl colonies (see below) which may cause local trends that might be undetectable at larger scales. Differences in trend direction observed between our study sites for five species found in common may be due to the negative ecological influence of the disproportionally larger densities of Lesser Snow and Canada Geese at the Akimiski Island site compared with lower densities of Canada Geese at the Burntpoint Creek site. Further, there may be an interaction with climate change processes that may be acting at different rates at the two sites. The Burntpoint Creek site is in low tundra habitat with continuous permafrost while the Akimiski Island site is in the boreal forest and is not influenced by permafrost.

We cannot offer definitive ecological explanations for trends observed because we have no observational or experimental evidence for causes as it was not the purpose of this research. However, we do offer two hypotheses (not necessarily competing) to help explain potential drivers and their mechanisms of influence on population change for the region encompassing our study areas: 1) relatively high densities of herbivores at our study sites (especially Akimiski Island) affects the habitat increasingly negatively and can indirectly affect predation rates of other bird species and their eggs/young (e.g., Abraham and Jefferies 1997, Flemming et al. 2016), and 2) climate change impacts both habitat and environmental conditions that could influence the breeding success of populations of birds in the sub-Arctic region at both sites (Wauchope 2017, Weiser et al. 2018b).

Herbivores may affect the abundance of bird populations in several ways. Herbivores may have enough impact on the vegetation that they consume that it could change important

breeding vegetation structure used for nesting cover or feeding by other species (Rockwell et al. 2003, Peterson et al. 2013, Flemming et al. 2019). If sufficiently abundant, herbivores may also attract and maintain predator populations that could also target alternate prey or their eggs (Iles et al. 2013). This is a mechanism of concern where herbivores have become over-abundant (Lamarre et al. 2017) but it is not known what minimal densities herbivores must attain for other species to be affected. There is clear evidence that Canada Geese and Lesser Snow Geese at the Akimiski Island study site have had a detrimental impact on the plant community in coastal habitats mainly through the removal of biomass and creation of de-vegetated areas (Kotanen and Abraham 2013). The presence of geese and their eggs at these study sites also attract and maintain populations of predators (e. g., Arctic fox (*Vulpes lagopus*), Common Raven, American Crow, and Herring Gull (*Larus argentatus*)) that also prey on other bird species and their young (KFA *personal communication*). Continued or increasing effects of herbivory at these study sites may have an influence on the trends observed for some species. For example, we observed positive trends for American Crow and Common Raven abundance on Akimiski Island, which could be a result of their benefiting from availability of eggs and goslings of nesting geese. These two species in turn may cause a decline in some species whose eggs they also prey upon including those we predicted might have negative trends as a result. Of the duck species nesting at Akimiski Island, where we predicted negative trends, only the Northern Pintail trend was negative (low confidence). Trends for Mallard (high confidence) and American Black Duck (low confidence) were positive. In addition, other duck species where we could estimate trends including Green-winged Teal and American Wigeon, had high confidence negative trends (Table 1). The duck species with negative trends were also in agreement with the low confidence negative trends observed at the provincial scale. We found a negative trend for all the shorebird species nesting on Akimiski Island where we made trend predictions (Semipalmated Plover, Least Sandpiper, and Short-billed Dowitcher), However, none were covered well enough by the Breeding Bird Surveys to allow comparison of trend estimates at provincial and national scales and the trends estimated at the regional scale (atlas, HBL) were all low confidence. We did find that many of the open or grassland ground-nesting shorebird and waterfowl species showed negative trends at both study sites (Tables 1 and 2). Further study is needed to advance hypotheses regarding the direct and indirect influence of abundant herbivores.

Population trends of birds observed in Arctic and sub-Arctic regions could also be influenced by a changing climate through either climate-mediated change to habitat (e.g., Tape et al. 2006), development of a mismatch between nesting phenology and resource availability (e.g., Brook et al. 2015) or from changes in growing season length or other environmental conditions (Overpeck et al. 1997, Serreze et al 2000). These are not mutually exclusive hypotheses. The result may be a species breeding range expansion or contraction (La Sorte and Thompson 2007) that may appear as a significant trend at the local scale. With a poleward shift in species breeding distribution a possibility (Brommer 2004, Virkkala and Lehikoinen 2014), trends we observed for some species could be a consequence of climate change (e.g., longer suitable temperatures or growing seasons).

Shrub expansion (Wheeler et al. 2018) in the Arctic and sub-Arctic regions is one consequence of a changing climate (Tape et al. 2006); increases in shrub cover, density, and height (Myers-Smith et al. 2011, Sturm et al. 2001) are some of the changes expected from a warming climate (Forbes et al. 2010) in this region. These changes are altering the proportions of vegetative cover and may favor the expansion of some bird species but be detrimental to others (Nguyen et al. 2003, Ballantyne and Nol 2011). Shrub encroachment is occurring at both study sites either through the spread of woody vegetation along tributaries toward Hudson Bay (at Burntpoint Creek) or from a contraction of the open graminoid habitats in the supratidal zone due to increasing woody vegetation as observed in the James Bay region including Akimiski Island (RWB, KFA *unpublished data*). The sparrow species for which we estimated an increasing trend could have benefitted from shrub expansion at both sites. At Burntpoint Creek, only Lincoln's and Song Sparrows showed an increasing trend which was consistent with the regional scale but was opposite the high confidence trend at the provincial scale. American Tree Sparrow indicated a negative trend which was consistent with that found at Akimiski Island but there were not any other high confidence trends estimated for other geographic scales. The White-throated Sparrow had a low confidence positive trend for Burntpoint Creek which was opposite the negative trends estimated at the provincial and national scales. For Akimiski Island, only the Song Sparrow indicated an increasing trend while we estimated declining trends for the Lincoln's and American Tree Sparrows. There was also a high confidence negative trend estimated for the Lincoln's Sparrow at the provincial scale. The evidence is not straight forward but our trend comparisons suggest that perhaps the local estimated positive trends for the Lincoln's, Song, and possibly also the White-throated Sparrows support the shrub encroachment hypothesis. These species also had trends opposite those estimated for at least one of the provincial or national scales. Targeted research is needed to more robustly explore potential drivers of the trends for these species at our study sites.

Using alternate sources of information on bird population trends may be even more important for species at risk or species of special concern, particularly where there is little information. Recovery strategies and legislated requirements emphasize the need for trend information for listed species to measure the success of recovery strategies. We reported a low confidence trend (positive but 95% credible limits overlapping 0) for the Rusty Blackbird (*Euphagus carolinus*) at Burntpoint Creek which is listed as a species of special concern, mainly due to population declines from the loss of wintering habitat (COSSARO 2017, COSEWIC 2017). The Rusty Blackbird population was considered stable between 2004 and 2014 in Canada (COSEWIC 2017) which overlapped the period that they were monitored at Burntpoint Creek (2001 to 2018). There was a positive highly confident trend at the regional scale but trends at the provincial and national scale were of low confidence.

The collection of bird list data can be done with minimal effort while collecting data for other projects. In comparison with other breeding bird surveys, it is not nearly as time consuming or as expensive. For regions where there is little or no other monitoring conducted for breeding birds, bird list analysis of time-series data can be a valuable method to estimate trends. These types of analyses may help fill a knowledge gap over a wide geographic area where accessibility is poor and there is scant other information.

*Responses to this article can be read online at:*
https://www.ace-eco.org/issues/responses.php/1821

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