

POPULATION TRENDS OF TUNDRA-NESTING BIRDS IN CHURCHILL,  
MANITOBA: POTENTIAL EFFECTS OF INCREASING LESSER SNOW GOOSE  
(*CHEN CAERULESCENS CAERULESCENS*) POPULATIONS

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This is to certify that I have examined this copy of a master's thesis by

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examining committee have been made.

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## **CHAPTER ONE**

### **POPULATION TRENDS OF TUNDRA-NESTING BIRDS AT CAPE CHURCHILL, MANITOBA: EFFECTS OF INCREASING SNOW GOOSE POPULATIONS**

## ABSTRACT

Recent increases in lesser snow goose (*Chen caerulescens caerulescens*) abundance have raised the question of how populations of other tundra-nesting birds are impacted by increased goose herbivory and habitat degradation. Line transect surveys were conducted on 30 transects at the Nestor 1 study site outside of Churchill, Manitoba in June 1984, 1999, and 2000 to obtain estimates of population densities of tundra-nesting birds and determine if density changes have occurred in relation to increased goose habitat damage. We also compared bird abundance at a smaller scale between degraded and non-degraded freshwater sedge meadows.

Herring gulls (*Larus argentatus*) and semipalmated sandpipers (*Calidris pusilla*) exhibited significant ( $P < 0.05$ ) increases in population densities from 1984 to 1999 and 2000. Dunlins (*Calidris alpina*) had significantly lower densities from 1984 to 1999 ( $P < 0.10$ ) and nonsignificant declines from 1984 to 2000. Horned larks (*Eremophila alpestris*), lapland longspurs (*Calcarius lapponicus*), savannah sparrows (*Passerculus sandwichensis*), and the sedge meadow shorebird guild exhibited nonsignificant increases in density. The tundra-nesting passerine guild exhibited a significant ( $P < 0.05$ ) increase in density from 1984 to 1999. Arctic terns (*Sterna paradiseaea*), pectoral sandpipers (*Calidris melantos*), and willow ptarmigan (*Lagopus lagopus*) indicated significant ( $P < 0.05$ ) declines in densities of birds seen on transects and in the numbers of transects birds were detected on. Significantly lower abundances of passerines and shorebird guild birds were detected in degraded freshwater sedge meadows versus non-degraded meadows.

Results indicate that most species did not exhibit population density declines over time on the study area, despite increased snow goose damage and density. However, at

the scale of the habitat patch, the same groups of species had lower abundances in degraded habitats versus non-degraded habitats. Results suggest that habitat degradation led to lower bird abundance, however habitat damage may have to be widespread and severe to elicit population density declines over a large area, like Nestor 1. The species that exhibited density declines over the study area were larger, ground-nesting species.

## INTRODUCTION

Monitoring is needed to assess whether population changes are of management or conservation concern (Howe et al. 1989). In some temperate and tropical regions of North America, many bird populations have been monitored over time (Sauer et al. 1997). However, little information exists on population trends of tundra-nesting birds in the Nearctic, particularly for shorebirds and passerines (Gratto-Trevor et al. 1998). Breeding population trends are not available for many reasons, such as lack of Breeding Bird Survey data in tundra ecosystems (Gratto-Trevor et al. 1998), logistic challenges (Helmers and Gratto-Trevor 1996), and assumptions that populations are isolated and therefore undisturbed (Senner and Howe 1984).

Most of the historical focus on threats to tundra-nesting birds has revolved around migration stopover and wintering habitats. Loss and degradation of these habitats is cited as the main threat to tundra-nesting bird populations (Myers 1983, Howe et al. 1989, Morrison et al. 1994, Gratto-Trevor et al. 1998), because these areas have key food and cover resources that are often used by a majority of a population (Senner and Howe 1984). Little focus has been placed on threats to these birds on the breeding grounds (Senner and Howe 1984) and few studies have monitored population changes on the breeding grounds. Predation (Custer and Pitelka 1977, Helmers and Gratto-Trevor 1996), excessive hunting and harvesting (Pattie 1990, Forbes et al. 1992), and interspecific competition (Pattie 1990) affect long-term population trends on tundra breeding grounds. Breeding habitat loss has not previously been considered a serious threat to tundra-nesting birds. However, several authors (Ankney 1996, Abraham and Jefferies 1997, Ganter et al. 1996) recently recognized loss of breeding habitat as a threat to bird

populations. Recent changes in breeding habitats are attributed to increases in white goose populations.

Over the past thirty years, many populations of white geese increased significantly in North America (Ankney 1996). Winter counts of mid-continental lesser snow goose (*Chens caerulescens caerulescens*) and Ross' goose (*Chen rossii*) populations nearly quadrupled (USFWS 1999). Mid-continental white goose breeding populations are conservatively estimated to be > 5 million birds, with projected increases of 5-10% each year if no management actions are taken (USFWS 1999). Lesser snow geese are the most abundant component of mid-continental white goose populations, with estimates of 4.5 to 6 million individuals (Abraham and Jefferies 1997, USFWS 1999).

In total, 13 breeding colonies of lesser snow geese are monitored annually from the southern tip of Hudson Bay, Canada, to Wrangel Island, Russia, with most colonies showing population increases over the past 30 years (Cooke et al. 1995, Abraham and Jefferies 1997). The La Pérouse Bay colony, situated on the western shores of Hudson Bay, has been intensively studied since 1968 (Cooke et al. 1995). This colony grew from approximately 2,000 breeding pairs in 1968 to 22,500 breeding pairs in 1990 (Cooke et al. 1995) to 40,000 breeding pairs in 1997 (R. F. Rockwell, personal communication). Lesser snow geese are philopatric (Cooke et al. 1975, Cooke and Abraham 1980), resulting in increased densities of geese at breeding colony sites over time. High population density at La Pérouse Bay has led to increased herbivory on tundra vegetation and degradation of traditional nesting (Kerbes et al. 1990, Jano et al. 1998) and staging areas (Abraham and Jefferies 1997, Jano et al. 1998), such as southern Hudson Bay. As

habitat conditions degrade within breeding colonies, snow geese move to less degraded habitats, increasing the extent of habitat damage.

Snow goose foraging behavior negatively impacts tundra grasses and sedges. Geese “grub” (dig up roots and rhizomes of tundra sedges and grasses) and pull shoots of sedges and grasses from wetlands, which disrupts the wetland substrate (Abraham and Jefferies 1997). These foraging activities lead to the creation of small, non-vegetated patches across the landscape (Kerbes et al. 1990, Ganter et al. 1996, Abraham and Jefferies 1997). Exposed areas are prone to erosion (Kerbes et al. 1990), sedimentation, and development of hypersaline conditions (Iacobelli and Jefferies 1991, Srivastava and Jefferies 1996), which make them less suitable for revegetation and recovery (Kerbes et al. 1990, Abraham and Jefferies 1997). Similar impacts have been documented for lesser snow geese in coastal salt marshes on the Texas wintering grounds (Miller et al. 1997).

The extent of current habitat damage varies across the landscape. Salt marsh plant communities have been degraded the most, as these areas are preferred by snow geese (Kerbes et al. 1990, Abraham and Jefferies 1997). Currently, salt marsh habitats on the western shore of Hudson Bay are in states of severe to complete degradation (Abraham and Jefferies 1997). Because salt marsh habitats are highly degraded, geese have started foraging in the more common freshwater sedge meadow plant communities. Overall, the majority of this common plant community in the tundra is not degraded. However, with increasing goose populations and fragmentation of goose colonies, these areas are potentially threatened with habitat damage.

Despite research on the effects of high goose populations and habitat degradation on plants and geese (Cooch et al. 1989, Cooch et al. 1991a, Cooch et al. 1991b, Williams

et al. 1993, Cooke et al. 1995), relatively little information is available on effects on other bird species. Based on annual bird sightings at La Pérouse Bay, 30 species have shown significant declines concurrent with habitat degradation by snow geese, including complete loss of some species (Abraham and Jefferies 1997, Rockwell et al. 1997). In addition, surveys have shown apparent declines in nest density for red-necked phalaropes (*Phalaropus lobatus*), semipalmated sandpipers (*Calidris pusilla*), and savannah sparrows (*Passerculus sandwichensis*) from 1984 to the 1990s (Abraham and Jefferies 1997, Rockwell et al. 1997). Declines could be a result of loss of habitat resources needed for successful breeding.

The scale of habitat damage may impact how bird populations respond. Currently, declines in bird abundance are noted in the goose breeding colony cores, which are areas of severe and widespread damage (Rockwell et al. 1997). However, there are varying levels of degradation across the larger tundra landscape. With continuing population growth and high population levels, the size of areas impacted by geese has expanded resulting in small to moderate levels of habitat damage in small patches interspersed among largely undamaged habitat. Bird populations inhabiting areas of low to moderate overall damage may show different population trends compared to those occupying severely degraded areas. It is important to understand how bird populations respond to severe degradation as well as to slight to moderate degradation levels, because this level of damage is more typical throughout the tundra landscape.

This study reports population trends of tundra-nesting birds at a site on the periphery of a lesser snow goose colony, where habitat degradation has occurred to a lesser degree than within the colony. Our main objective was to determine if tundra-

nesting bird densities have changed significantly from 1984 to 1999 and 2000 with increasing snow goose habitat damage. We quantified population density changes on a large spatial scale (study site) and also compared bird abundance at the scale of habitat patches in severely degraded and non-degraded freshwater sedge meadow habitats. Finally, we compared data on bird densities in freshwater sedge meadows to densities in severely damaged salt marsh habitats.

## **STUDY AREA**

The Cape Churchill (CC) study area is located 35 km east of Churchill, Manitoba and 15 km southeast from the La Pérouse Bay snow goose colony core in Wapusk National Park (Figure 1). Base camp (Nestor 1) is located approximately 2 km west from the coast of Hudson Bay, where Canada goose (*Branta canadensis*) productivity has been studied since 1968 (Walter 1999).

The study area lies within the Hudson Bay Lowlands physiographic region, which is characterized as a tundra biome. The area is composed of a series of coastal salt marshes, willow (*Salix spp.*) and spruce (*Picea spp.*) patches, and beach ridges. The surrounding matrix of the landscape consists of large, permanent freshwater lakes, ephemeral freshwater pools, and freshwater sedge meadows. Existing landscape features, such as permafrost, minimal relief, and permanent water bodies, are a result of Pleistocene glaciation events. Isostatic uplift, or the uplifting of land after the retreat of glaciers, is still occurring at a rate of 1-1.5 m/century (Hunter 1970).

Beach ridges on the study site are characterized by higher elevation than surrounding freshwater sedge meadow communities and dominance of lyme grass (*Elymus arenarius*) and white mountain avens (*Dryas integrifolia*). Lowland marsh areas



are dominated by sedge species (*Carex spp.*, primarily *Carex aquatilis*). Further inland and on dryer areas, small patches of willows occur, in addition to white spruce (*Picea glauca*) and larch (*Larix laricina*). Coastal areas are generally dominated by lyme grass and sedge species among rocky and gravelly substrates. Most vegetation communities at Cape Churchill are arranged parallel to the coast. For extensive details of vegetation at the Nestor 1 study area, refer to Didiuk and Rusch (1979).

The study area has patches of degraded habitat, consistent with those resulting from snow goose foraging activities. These areas are characterized by loss of vegetation or presence of a moss carpet (Jefferies 1988). Coastal salt marshes are completely degraded, with only the presence of muddy substrates and underlying rocks. Sections of freshwater sedge meadows exhibit varying habitat degradation, from severe to no degradation.

## **METHODS**

### **Surveys**

Surveys for breeding birds were conducted using line transect methodology (Burnham et al. 1980). In 1984, 30 transects, ranging from 90 to 1870 m in length, were established within the Nestor 1 study site. Transects were generally laid out within a continuous habitat type, although some transects crossed multiple habitat types. Each transect was surveyed once between mid- and late June and during the hours of 1200 to 2000. Transects were only surveyed on days with winds < 15 mph, minimal to no fog or precipitation, and clear to moderately cloudy skies. Any bird that was visible or audible from any point along the transect was recorded. Information on the sex of the bird, bearing of the bird when first detected (obtained using a compass), estimated distance

from detection point, habitat type, and behavior were recorded. Distances were estimated visually and were confirmed by a second observer. The observer walked at a slow and constant pace, counting individuals only once (Bibby et al. 1992). Proportion of each habitat type along the transect was estimated visually. Habitat categories were open water, beach ridge, freshwater sedge meadow, and salt marsh.

In 1999 and 2000, the transects were surveyed using the methods used in 1984, with a few modifications. Transects were surveyed twice in 1999, whereas transects were only surveyed once in 2000. For the second surveys in 1999, the walking route on transects was reversed, and transects were visited at different times of the day compared to the first survey. In 1999 and 2000, distances were estimated via a parallel-type range finder (Ranging 400 model).

### **Density Estimation ( $n \geq 30$ )**

Sightings from surveys were converted to perpendicular distances and entered into the DISTANCE computer program to obtain density estimates for bird species or guilds with  $n \geq 30$  sightings in 1984, 1999, and 2000 (Buckland et al. 1993). Guilds were compiled for shorebirds that nest in freshwater sedge meadows, passerines that nest or are dependent upon willow patches for reproduction, and freshwater sedge meadow-nesting passerines (Appendix A). To avoid any complications with aberrant observations far from the line, only observations up to 200 m in perpendicular distance were used (Buckland et al. 1993). For each species or guild in each year, multiple detection function models and curves were fit to the resulting detection function histogram. We tested detection functions for appropriate fit with uniform/cosine, half-normal/cosine, and

hazard/polynomial functions. Akaike's Information Criterion (AIC) was used to distinguish which model was the best fit to the detection data (Akaike 1985, Burnham and Anderson 1992, Buckland et al. 1993).

AIC often resulted in selection of different detection functions for the three years when analyzing for a species or guild. It was important to have consistency among years with detection function choice (Buckland et al. 1993). Therefore, we selected detection functions for some years in which the chosen function did not have the lowest AIC value of all tested detection function models for that year. For most analyses, the detection function used was the best fit for two of three years examined (one with lowest AIC value for those years). For the year(s) in which the chosen detection function did not have the lowest AIC value, we evaluated model fit as follows: First, to use a detection function that did not have the lowest AIC value for that year, we required the detection function be within 10.0 AIC units of the model with the lowest AIC value for that year. Next, we looked at the chi-square value of the chosen detection function. This statistic provides another means of assessing detection function model fit. However, because it is alterable and less powerful than AIC, it is not used extensively (Buckland et al. 1993). We changed the bin width of the detection function histogram to maximize the P-value of the chi-square test. If the chi-square goodness-of-fit P-value could be increased to a non-significant value, we used the detection function with the higher AIC value.

### **Bird Abundance Estimation ( $n < 30$ )**

There were many species that did not have adequate sample sizes ( $n \geq 30$ ) during at least one of the three years for line transect density estimation. For these species, we tallied the total number of individuals seen within 200 meters (perpendicular distance) of

transects on surveys. We divided this number by the total distance traversed on surveys to obtain an estimate of encounter rate across transects. The exception to this was lesser snow geese. Any goose seen within 400 m from the transect was counted, because snow geese tended only to be approachable within 300-400 meters. Direct comparisons were made between years of these data, with large changes in density (at least 2.5 times differences between years) noted. Additionally, only species for which  $\geq 30$  individuals were detected in  $\geq 1$  of the years were included in this analysis. This was done to ensure that the species of interest were abundant enough at one point during the study period to have had density estimation done using line transect methodology.

### **Freshwater Sedge Meadow Habitat Comparison**

Many transects established in 1984 did not cover any large areas of severe habitat degradation in 1999 and 2000. Therefore, impacts of severe degradation may not be detectable in surveys of the entire study site. However, there may be distinct differences in bird abundance between degraded and non-degraded freshwater sedge meadows. Therefore, we conducted surveys in habitat patches of severely degraded and non-degraded freshwater sedge meadows to evaluate bird population responses to degradation at a smaller scale than the entire Nestor 1 study area.

In June 2000, 3 transects were established in areas of severe habitat degradation within freshwater sedge meadows. These areas were identified based on lack of vegetation or large swaths of areas with moss carpets, both of which are indicative of heavy snow goose habitat damage (Jefferies 1988, Kerbes et al. 1990, Abraham and Jefferies 1997, Kotanen and Jefferies 1997). Line transect surveys were conducted twice

at these locations during the breeding season (mid- to late June) between the hours of 0900 and 1500. Survey methods were the same as described for transects conducted in 1999 and 2000.

We compared an index of bird abundance among transects in non-degraded and degraded habitat. Four transects in undamaged habitats were selected from those established in 1984 based on their lack of visible habitat damage. To standardize comparison of survey areas in which birds were counted, only birds detected at a perpendicular distance of  $\leq 100$  m from the transect were counted. The total number of birds seen per habitat type was tallied for species and guilds. Guilds used for this analysis were freshwater sedge meadow nesting passerines and freshwater sedge meadow nesting shorebirds (Appendix A). The number of sightings was divided by the total distance walked on survey routes in degraded and non-degraded areas. Direct comparisons were made between this index to bird abundance in degraded and non-degraded areas.

To evaluate differences between degraded and non-degraded habitats, we conducted vegetation measurements on all transects. As outlined by Kerbes et al. (1990), we used a 1 m x 1 m wooden frame with 25 grid intersections, evenly spaced over the frame. Vegetation measurements began 25 m up from the starting point of the transect and were conducted 50 m perpendicularly from this point. At this location, we placed the grid on the ground and tallied the number of habitat types that grid intersections touched. These habitat types included: bare ground, moss, sedge, woody plants, other plants, ephemeral water, permanent water, or miscellaneous. In addition, the numbers of goose droppings, willow stems, and grubbed turf pieces were counted throughout the grid.

After collection of data at one site, we returned to the transect and walked another 50 m. We then walked 50 m perpendicular to the transect at this location. We alternated taking vegetation measurements to the left and right sides of the transect. Vegetation measurements were taken until the end of the transect was reached.

## **Salt Marsh Surveys**

None of the original 1984 transects traversed any of the coastal salt marsh vegetative communities. To evaluate bird abundance in these communities, we established 3 transects in salt marsh habitats in 2000. Didiuk and Rusch (1979) indicated that in the late 1960s and early 1970s, coastal salt marsh communities were well vegetated and were the key areas of brood rearing for Canada geese, as well as other waterfowl and shorebird species. These areas exhibit the most visible signs of snow goose habitat damage on the study area. Over half the coastal salt marshes along Hudson Bay are completely degraded; the remaining areas are in a state of severe degradation (Abraham and Jefferies 1997). Currently, salt marsh communities around the Cape Churchill area are characterized by a complete lack of vegetation, with occasional pockets of mossy growth.

Because the salt marsh habitat on the Nestor 1 study site was limited, transects were established to maximize survey coverage of the area. We used a drawn map of vegetative communities in the most southern unit of the study site with salt marsh habitat. This area is approximately 700 x 1000 m. We then placed a 1 x 1 cm grid over the map and numbered all cells. A cell number was chosen randomly for a starting location. If the chosen cell did not contain appropriate habitat, another cell was chosen. Next, a

transect direction was chosen randomly. However, if the transect was placed in a direction that resulted in reaching the edge of the vegetative community before 600 m, a new direction was selected. Transect length was then established to be the longest length walked before reaching the community edge or 1000 m. At the end of the initial transect, we moved 100 m north of the ending point. One row of 1 x 1 cm cells was placed on the map of the area in this location and numbered. A randomly drawn cell was selected for the starting location of the next transect. Length and direction were selected as mentioned previously, and so on, until all transects were laid out within the habitat. Surveys were conducted in the same manner as described for 1999 and 2000 line transect surveys. Numbers of birds seen within 100 m of the transect were tabulated for guilds (Appendix A). These numbers were divided by the total length of survey routes to obtain an encounter rate (index to density).

## **Data Analysis**

For line transect density estimation, we obtained a point estimate and standard error for density for each year for species and guilds with sample sizes  $n \geq 30$ . Ninety-five percent confidence intervals were constructed around point estimates using DISTANCE and were compared between years. Significant density changes were noted if there was no overlap between confidence intervals.

For species with  $n < 30$ , we determined how many transects had suitable habitat in which a particular species would be expected to occur. Expected habitat for a species was determined using knowledge from previous experiences as to which vegetative communities birds were seen nesting or foraging in. A transect was recorded as having suitable habitat for a species if that transect crossed through a section of the noted habitat

type for the species in question. Then, for each year, we tallied the proportion of transects in which individuals were detected. These proportions were compared using a chi-square test, to evaluate whether significant changes had occurred in the proportion of transects where birds were detected. A one-tailed test was used because survey data [see Bird Estimation Density ( $n < 30$ )] indicated that the species with  $n < 30$  had declined over time. Additionally, we computed the number of birds observed/m walked for each transect that a bird was detected. We compared this abundance index between years using one-way ANOVA to determine whether density on transects had changed over time.

For comparison of bird abundance between degraded and non-degraded freshwater sedge meadows, variability in index values could result from three sources. First, each transect had different compositions and arrangements of habitat types, which likely influenced bird abundance among transects. Second, variation results from differences in habitat quality. Abundance likely varies according to quality of the habitat patch surveyed (i.e. degraded vs. non-degraded). Finally, there is variation in abundance of birds seen at two different survey times during the breeding season. Variation is likely to exist between abundances of birds detected on the initial and subsequent survey. All three sources of variation contribute to variation in abundance index values we calculated for transects. To determine if one or more of these sources of variation could explain a significant proportion of total variation in the data, we used a general linear model (Cook and Weisberg 1999). This model was:



**Variation in Index to Abundance = Variation from Habitat Quality + Variation  
from Survey Times + Error**

Due to small sample sizes, we were unable to incorporate potential variation from individual transects into the general linear model. The reduced model was tested for density indices of various species and guilds across degraded and non-degraded transects. Components of the linear model that were significant at  $P < 0.05$  were considered to be elements that were associated with significant changes in bird abundance.

For comparison of plant communities between degraded and non-degraded freshwater sedge meadows, two-sample, two-tailed t-tests were used. For each transect, average estimates of percent cover of sedge, woody plants, bare ground, moss, damaged habitat (bare ground and moss), and non-moss vegetative material (sedge, woody, and leafy vegetation) were obtained. Percent cover for one sample point was estimated by the number of intersection grid points that hit that habitat type divided by the total, non-water, habitat points. We obtained a mean for all sample points on a transect for each category (to obtain the average estimates of cover per transect).

## **RESULTS**

### **Density Comparisons**

In 1984, 1999, and 2000, totals of 37, 36, and 34 species were detected on transects, respectively. Dunlin (*Calidris alpina*), herring gull (*Larus argentatus*), horned lark (*Eremophila alpestris*), lapland longspur (*Calcarius lapponicus*), savannah sparrow, and semipalmated sandpiper had  $n \geq 30$  in 1984, 1999, and 2000.

Years that did not have the lowest AIC values for a chosen detection function had appropriately high chi-square goodness-of-fit P-values (Table 1) indicating that the model exhibited acceptable fit, despite the fact it did not have the lowest AIC value of all tested models.

For dunlins, a sedge meadow nesting shorebird, there were decreases in point estimates of density from 1984 to 1999 and 2000 (Table 2, Figure 2). In 1999, the decrease in density was marginally significant ( $P < 0.10$ ). However, the 1984 and 2000 counts showed no significant difference in density. Herring gulls, habitat generalists, had significant ( $P < 0.05$ ) increases in point estimates of density (Table 2, Figure 2). Sedge meadow passerines, such as lapland longspur and savannah sparrow, exhibited nonsignificant increases in point estimates of density from 1984 to 1999 and 2000 (Table 2, Figure 2). Horned larks, a passerine that mainly uses beach ridge vegetation communities, demonstrated large nonsignificant increases in point estimates of density (Table 2, Figure 2). Semipalmated sandpiper, a sedge meadow nesting shorebird, had significant ( $P < 0.05$ ) increases in point estimates of density from 1984 to 1999 and 2000 (Table 2, Figure 2).

For the freshwater sedge meadow nesting shorebird guild, densities from 1984 to 1999 and 2000 did not change significantly, although point estimates of density were higher in 1999 and 2000 than in 1984 (Table 2, Figure 2). For the guild of willow-nesting birds, no significant changes in point estimates of density occurred, although variability was high (Table 2, Figure 2). Freshwater sedge meadow passerines showed increases in point estimates of densities through time, with a significant ( $P < 0.05$ )

increase seen from 1984 to 1999. However, the increase was not significant from 1984 to 2000 (Table 2, Figure 2).

For species with  $n < 30$ , comparisons of indexes of abundance suggested changes in density. Six species showed a notable decline from count data taken in 1984 to 1999 or 2000, while lesser snow geese exhibited an increase (Table 3). Lesser snow goose numbers have greatly increased over the past 10 years on the Nestor 1 study site (D.E. Andersen, personal communication, D.H. Rusch, personal communication). From count data, there were at least 27 times more geese seen on transects in 1999/2000 than in 1984. For species where declines were indicated, some of these apparent declines were stark. Most notable was the lack of sightings of pectoral sandpipers (*Calidris melantos*) in 1999 and 2000, as well as a lack of sightings of willow ptarmigan (*Lagopus lagopus*) in 2000. There were significant declines in number of transects birds were detected on between 1984 and 1999/2000 for arctic terns (*Sterna paradisaea*) ( $\chi^2_1 = 9.17$ ,  $P = 0.0029$ ), pectoral sandpipers ( $\chi^2_1 = 12.94$ ,  $P = 0.0003$ ), and willow ptarmigan ( $\chi^2_1 = 4.60$ ,  $P = 0.0320$ ). Long-tailed ducks (*Clangula hyemalis*) ( $\chi^2_1 = 2.53$ ,  $P = 0.1118$ ), pacific loons (*Gavia pacifica*) ( $\chi^2_1 = 0.46$ ,  $P = 0.4984$ ), and red-necked phalaropes ( $\chi^2_1 = 0.98$ ,  $P = 0.3216$ ) had large declines in count data from 1984 to 1999 and 2000; however these species were not seen on significantly fewer transects between years. For most species, indices to density on transects did not significantly change from 1984 to 1999 or 2000. However, arctic terns had significantly lower indices to abundance along transects from 1984 to 1999 ( $F_{2,17} = 1.95$ ,  $P = 0.0342$ ) and 2000 ( $F_{2,17} = 1.83$ ,  $P = 0.0420$ ). Additionally,

willow ptarmigan had significantly lower indices to abundances on transects from 1984 to 1999 ( $F_{2,10} = 2.41$ ,  $P = 0.0182$ ).

### **Freshwater Sedge Meadow Comparisons**

Differences in vegetation were evident between degraded and non-degraded habitats (Table 4). Degraded transects had significantly higher proportions of bare ground ( $t_5 = -2.56$ ,  $P = 0.0253$ ), moss ( $t_5 = 2.81$ ,  $P = 0.0030$ ) and proportions of combined moss and bare ground cover ( $t_5 = -9.62$ ,  $P = 0.0001$ ) compared to non-degraded transects. Non-degraded transects had significantly higher proportions of sedge cover ( $t_5 = 4.58$ ,  $P = 0.003$ ), proportions of non-moss vegetative cover ( $t_5 = 8.96$ ,  $P = 0.0001$ ), numbers of hummocks ( $t_3 = 9.47$ ,  $P = 0.0011$ ), and woody stems ( $t_5 = 5.54$ ,  $P = 0.0013$ ).

Bird densities were calculated for four species and two guilds. In all species and guilds, large declines in abundance were seen in degraded areas compared to non-degraded habitats. Using the general linear model for the species and guilds, habitat damage was determined to be the most significant factor explaining differences in bird abundance across transects (Table 5). Lapland longspur, savannah sparrow, shorebird, and passerine abundance were all significantly ( $P < 0.05$ ) related to habitat damage in the general linear model. Habitat damage was not a significant predictor of abundance for dunlins and semipalmated sandpipers.

### **Salt Marsh Data**

In general, results from transect surveys conducted in the coastal salt marshes (Table 6) indicated bird densities were similar to those found in degraded freshwater

sedge meadows. Salt marshes had lower total densities for the sedge meadow passerine guild compared to degraded sedge meadow areas. However, some individual passerine species (e.g. lapland longspur) demonstrated higher densities in salt marshes compared to degraded sedge meadows. Shorebird densities were highest in salt marshes. This trend also was maintained with individual dunlin counts. Finally, herring gulls exhibited highest densities in salt marshes.

## **DISCUSSION**

Across the Nestor 1 study area, trends in population densities of shorebirds from 1984 (year of low goose density and damage) to 1999/2000 (years of high goose density and damage) were variable across species and spatial scales. At the scale of the study area, semipalmated sandpipers showed significant increases in density, whereas dunlins showed potential declines in population density. Semipalmated sandpipers exhibited a large increase in point estimates of density and were relatively common on the study site. This species can use freshwater habitats among a spectrum of habitat damage (J. Sammler, unpublished data). Dunlins also use damaged freshwater sedge meadows, but not as actively as semipalmated sandpipers (J. Sammler, unpublished data). Potential declines in dunlins compared to semipalmated sandpipers could result from differential ability to adapt to habitat changes.

Passerines showed a consistent trend of nonsignificant increases over the 17 year period. Horned larks exhibited the strongest indication of population increase. This is not surprising, as this species often used beach ridge plant communities for most of their breeding and foraging activities. This particular plant community has remained relatively

intact, and geese avoid this habitat due to its lack of food resources. However, lapland longspurs and savannah sparrows use freshwater sedge meadows for breeding and foraging. These birds also exhibited nonsignificant increases in population densities over time, and the habitats these species use have areas of severe habitat damage. However, freshwater sedge meadows are the dominant community across the Nestor 1 study area; much of this community has not yet been severely impacted by geese. Because tundra passerine species likely have relatively small habitat area requirements, large scale and widespread habitat disturbance in freshwater sedge meadows may be needed to elicit population density changes in this group.

A significant population increase was evident in herring gulls. Herring gulls are opportunistic and generalist predators (Pierotti and Good 1994). Obtaining food items and resources in tundra ecosystems is difficult for many species, even during the resource abundant summer seasons. For gulls, the ability to obtain appropriate food resources for themselves and their broods is energetically demanding and could limit population growth (Murphy et al. 1984, Morris 1987, Pierotti 1987). High prey abundance influences predator populations and reproduction. The increase in herring gull abundance is likely due to increased prey resources in the form of lesser snow goose goslings. Gulls regularly take goslings (Walter 1999, J. Sammler, personal observation), which provide easy and large food items to feed themselves and raise their broods. Additionally, other studies at snow goose colonies show that gulls change the composition of their diet to almost entirely goose resources when the nesting season begins (Samelius and Alisauskas 1999). Gosling densities at Nestor 1 have grown in recent years, due to the land migration of adults with broods from La Pérouse Bay. Increases in gosling numbers over

the past 17 years has likely provided gulls abundant food resources through the breeding season and favored high reproductive success and subsequent population increases.

For those species with sample sizes  $< 30$  for at least one year, sample sizes  $\geq 30$  were observed in 1984 but not in 1999 or 2000, except for lesser snow geese. It is difficult to determine whether population declines are occurring. However, for some species, population declines are likely occurring. Arctic terns showed a significant decline from 1984 to 1999/2000 in the number of transects birds were seen along as well as the encounter rate of birds on transects. These declines may be difficult to relate to snow goose damage, as some tern species have low annual nest site fidelity (Parnell et al. 1995, Thompson et al. 1997). Although data on arctic tern site fidelity are scarce, some studies indicate high degrees of site fidelity for some arctic tern colonies, particularly when annual habitat changes are not dramatic (Bergman 1980, J. Sammler, personal observation). Some studies have noted large similarities in nesting habits of arctic terns to those of common terns (*Sterna hirundo*), a species with high site fidelity for some colonies (Bergman 1980, Burger 1991). Therefore, it is likely that arctic terns exhibit fairly high annual nest site fidelity. Examination of historical tern nesting areas at Nestor 1 shows many of these areas no longer support terns (J. Sammler, personal observation). Therefore, local declines are possible. Other large ground-nesting birds have shown similar declines. Willow ptarmigan were found on significantly fewer transects between 1984 and 1999/2000. This species is a permanent resident, so it is likely that declines at Nestor 1 may be related to habitat condition. The most notable apparent decline was seen in pectoral sandpipers; birds were not observed on transects at all during 1999/2000.

This once common species is rarely observed on the study site during the migration or breeding seasons (J. Sammler, personal observation).

One potential explanation for why some species may be declining on the study site relates to the predator community and its relationship to snow geese. Snow geese are not abundant at Nestor 1 during the initial nesting period (early to mid-June). Almost all snow geese nest at La Pérouse Bay. When synchronized hatching occurs, large numbers of geese walk their goslings from the colony core to the Nestor 1 study area. Therefore, goslings, as a food source for predators, are not available until the third or fourth week of June. This abundant food resource may encourage high predator densities. However, before movement of snow goose broods, the study area predators use other species as prey. This predator community may exert an unusually high annual predator pressure on other birds before snow geese arrive at CC, particularly on ground-nesting birds. Therefore, medium to large size ground-nesting birds, such as pacific loon, arctic tern, long-tailed duck, and willow ptarmigan, may be very susceptible to predation early in the nesting season.

The primary predators likely to impact nesting success of many birds are herring gulls, parasitic jaegers (*Stercorarius parasiticus*), and arctic fox (*Alopex lagopus*). Gulls have increased significantly at Nestor 1 and are highly opportunistic, particularly in the beginning of the breeding season (J. Sammler, personal observation). Parasitic jaegers likely do not have a large impact on bird populations, because jaegers defend relatively large territories (Wiley and Lee 1999) and only 3-4 pairs are seen on the site from year to year (J. Sammler, personal observation). Arctic fox are probably important nest predators. Fox are opportunistic and are known to consume bird eggs and adults (e.g.



ptarmigan, long-tailed ducks, loons, terns, and small passerine birds) (Fay and Stephenson 1989, Prestrud 1992). With abundant populations of opportunistic predators, breeding birds may be under significant predator pressure before snow geese arrive at Nestor 1. Early nesting, medium to large-sized birds, like willow ptarmigan, arctic tern, and pectoral sandpiper, may all be major sources of food for predators during the beginning of the nesting season. With high predator activity, nest failure is likely. Because most tundra-nesting birds only attempt one clutch per breeding season, recruitment may be below levels needed to maintain stationary or increasing populations. This hypothesis has been proposed as an explanation for Canada goose declines at Nestor 1 (Walter 1999) and may contribute to nest losses of other bird species as well.

### **Freshwater Sedge Meadow Habitat Comparison**

A comparison of bird densities between degraded and non-degraded habitats showed a distinct difference in abundance of the most common species. Lapland longspur, savannah sparrow, dunlin, semipalmated sandpiper, passerines, and shorebirds were detected less frequently in degraded areas, including significantly lower abundance indices for lapland longspurs, savannah sparrows, and passerines. These results are interesting, because most of these species appeared to exhibit nonsignificant increases in density over time at the scale of the study area. However, the degraded areas either had no vegetation or moss carpet vegetation, and the entire study area is not in such a severely degraded condition. Much of the study area is comprised of areas of no to slight damage. There are likely still adequate amounts of intact and appropriate habitat for many birds to use at the scale of the study area. Therefore, the differences in density may

indicate that a certain level of degradation may have to take place for declines in densities of some species to be exhibited at a larger scale. This study suggests that even a moderate amount of intact or even slightly to moderately degraded habitat may still provide enough habitat resources for many bird species to persist and increase in abundance, particularly passerines.

### **Salt Marsh Comparison**

Salt marshes on the Nestor 1 study area are highly degraded and have shown little recovery. However, densities of birds were high for salt marshes. Several potential reasons explain why high densities of birds, particularly shorebirds, were seen in salt marshes. First, comparisons were based on direct counts along transects. This method gives an estimate of encounter rates of birds over transects but does not incorporate different sighting probabilities of birds between habitats. Salt marshes are heavily degraded and have no remaining vegetation to obscure sightings. Thus, detection probabilities were probably high. Birds in sedge meadows are likely harder to detect, resulting in lower detection probabilities.

Second, salt marshes are still used regularly by birds, particularly shorebirds. However, the majority of birds (90%) seen in salt marshes were flocks of foraging birds, probably non-breeders. Birds still forage in these degraded areas and can be quite numerous in certain locations (J. Sammler, personal observation). Therefore, the large numbers of apparent non-breeding birds in salt marshes may also lead to high density estimates. The high densities of scavenging predators, like herring gulls, is likely due to the fact that many of these birds nest on islands in the wetlands in this habitat.

Overall, this vegetative community may still be valuable to birds. This is indicated by large numbers of foraging birds recorded at salt marshes, particularly during migration (J. Sammler, personal observation). However, low densities of passerines and prevalence of primarily foraging birds in this community appears to indicate this habitat is not suitable as breeding habitat for most birds.

## **MANAGEMENT IMPLICATIONS**

With the heightened concern regarding the impact of high snow goose populations, many reasons have been proposed regarding why increasing snow goose populations need to be controlled. One of these reasons is populations of tundra-nesting birds are being negatively impacted by increasing goose populations through their impacts on tundra habitats.

Clearly, snow geese have severely degraded salt marsh communities throughout large portions of the Arctic and sub-arctic. Such severe degradation likely negatively impacts bird communities using these areas. However, the majority of the tundra landscape in our study area and elsewhere is comprised of freshwater sedge meadows. Most sedge meadows across the tundra landscape are relatively intact, despite growing use by snow geese over time in some areas. Even in areas where moderate degradation has occurred (such as Nestor 1), most bird populations do not indicate declines in density over time, even when snow goose use and habitat damage increased. Our results suggest that impacts on breeding birds using these habitats are not clearly exhibited at the scale of our study area but are evident at the habitat patch level. Thus, declines in bird populations are not apparent at larger scales under current habitat conditions.

However, despite the fact that most bird populations are not exhibiting declines, even in areas of moderate goose damage, goose damage has been shown to eventually cause declines in population density. Our study suggests that if damage is widespread and severe, population declines will occur, even in the extensive and common freshwater sedge meadow areas. Recently, snow goose colonies started to fragment out of degraded salt marshes and establish smaller clusters of breeding geese in freshwater sedge meadow communities. Geese can successfully raise broods in this habitat. As a result, the expansion of goose colonies into intact freshwater sedge meadows could result in long-term occupation and eventual degradation of such areas. With current population levels and reproductive potential of snow geese, damage of freshwater sedge meadows could be widespread and severe enough to cause declines of bird populations that use this plant community for breeding and foraging.

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Table 1. The largest  $\chi^2$  P-values for species and guilds during years when detection function selection was not based on lowest Akaike's Information Criterion values for line transect analyses at Cape Churchill, Manitoba.

<b>Species/Guild</b>	<b>Detection Function</b>	<b>Year</b>	<b>Adjusted Chi-square P-Value</b>
Dunlin	Uniform/Cosine	1984	0.10556
Herring gull	Uniform/Cosine	1999 2000	0.24772 0.58087
Horned lark	Hazard/Polynomial	2000	0.29426
Lapland longspur	Half-normal/Cosine	1984 1999	0.07830 0.55842
Semipalmated sandpiper	Hazard/Polynomial	1984	0.08470
Shorebird guild	Half-normal/Cosine	1984 1999	0.12565 0.42733
Willow-nesting passerine guild	Hazard/Polynomial	1999	0.68136

Table 2. Density estimates ( $\pm 1$  SE) in number of birds/km<sup>2</sup> for species with  $\geq 30$  individuals detected each year (1984, 1999, and 2000) based on line transect analysis at Cape Churchill, Manitoba.

Species	Detection Function	1984	1999	2000	Significance <sup>a</sup>
Dunlin	Uniform/ Cosine	22.98 $\pm$ 3.33	11.76 $\pm$ 2.43	22.90 $\pm$ 4.44	* from 1984 to 1999
Herring gull	Uniform/ Cosine	5.25 $\pm$ 0.56	21.19 $\pm$ 11.88	18.82 $\pm$ 5.13	**
Horned lark	Hazard/ Polynomial	24.88 $\pm$ 5.64	52.62 $\pm$ 14.89	49.25 $\pm$ 12.76	
Lapland longspur	Half-normal/ Cosine	50.41 $\pm$ 7.07	77.26 $\pm$ 9.92	47.83 $\pm$ 6.47	
Savannah sparrow	Half-normal/ Cosine	32.73 $\pm$ 7.04	34.64 $\pm$ 5.15	62.11 $\pm$ 9.47	
Semipalmated sandpiper	Hazard/ Polynomial	14.85 $\pm$ 3.46	60.45 $\pm$ 28.24	47.73 $\pm$ 12.94	**
Shorebird guild	Half-normal/ Cosine	48.10 $\pm$ 8.29	75.19 $\pm$ 19.74	63.52 $\pm$ 11.87	
Tundra-nesting passerines	Hazard/ Polynomial	74.33 $\pm$ 6.84	165.36 $\pm$ 31.70	112.85 $\pm$ 20.87	** from 1984 to 1999
Willow-nesting passerines	Hazard/ Polynomial	14.76 $\pm$ 6.54	14.68 $\pm$ 8.53	43.73 $\pm$ 43.76	

<sup>a</sup> Significance measured via confidence intervals. \* indicates  $P < 0.10$ , \*\* indicates  $P < 0.05$

Table 3. Encounter rate (number of birds seen within 100 m of transect divided by total km surveyed) of species with sample sizes  $n \geq 30$  for at least one of three analysis years (1984, 1999, and 2000) at Cape Churchill, Manitoba.

Species	Survey Year			Greatest Ratio Between Two Years
	1984	1999	2000	
Arctic tern	3.3253	0.3855	0.5783	8.6 times
Lesser snow goose	0.3855	20.1928	10.5542	52.4 times
Long-tailed duck	1.3012	0.5301	0.3373	3.9 times
Pacific loon	1.4940	0.8675	0.3855	3.9 times
Pectoral sandpiper	1.3494	0.0000	0.0000	---
Red-necked phalarope	1.3494	0.5301	0.1928	7.0 times
Willow ptarmigan	1.0602	0.2410	0.0000	4.4 times

Table 4. Differences in means ( $\pm 1$  SE) of vegetative characteristics of comparison transects between non-degraded and degraded freshwater sedge meadows at Cape Churchill, Manitoba (June 2000).

<b>Vegetation Characteristic</b>	<b>Non-Degraded Mean</b>	<b>Degraded Mean</b>	<b>T</b>	<b>d.f.</b>	<b>P-value</b>
% Bare Cover	14.15 $\pm$ 5.01	42.91 $\pm$ 11.38	-2.56	5	0.0253
% Moss Cover	6.72 $\pm$ 2.51	36.55 $\pm$ 12.17	2.81	5	0.0030
% Bare and Moss Cover	28.2 $\pm$ 4.4	88.13 $\pm$ 4.09	-9.62	5	0.0001
% Sedge Cover	38.81 $\pm$ 4.72	7.33 $\pm$ 4.8	4.58	5	0.0030
% Non-Moss Vegetative Cover	70.61 $\pm$ 4.73	11.87 $\pm$ 4.09	8.96	5	0.0001
No. of Woody Stems	1.82 $\pm$ 0.23	0.18 $\pm$ 0.12	5.54	5	0.0013
No. of Goose Droppings	3.5 $\pm$ 0.16	7.70 $\pm$ 1.57	-2.56	5	0.0575
No. of Grubbed Pieces	0.071 $\pm$ 0.071	3.68 $\pm$ 2.34	-1.54	2	0.1315
No. of Hummocks	5.27 $\pm$ 0.54	0.13 $\pm$ 0.069	9.47	3	0.0011

Table 5. Significance values for assessing contribution of variation due to differences in survey time and extent of habitat damage on total variation in abundance of individual bird species and species guilds at Cape Churchill, Manitoba (June 2000).

Species/Guild	Survey Variation		Damage Variation	
	F <sup>a</sup>	p-value	F <sup>a</sup>	P-value
Dunlin	0.18	0.6819	2.02	0.1827
Lapland Longspur	0.38	0.5511	28.48	0.0000
Savannah Sparrow	0.97	0.3467	8.41	0.0145
Semipalmated Sandpiper	1.41	0.2952	3.47	0.0895
Passerine Bird Guild	0.01	0.9425	37.37	0.0001
Shorebird Guild	1.72	0.2164	22.69	0.0006

<sup>a</sup> d.f. = 1,13 for all tests using the general linear model

Table 6. Comparison of mean number of birds seen/km surveyed for individual species and guilds among non-degraded freshwater sedge meadows, degraded freshwater sedge meadows, and degraded salt marshes at Cape Churchill, Manitoba (June 2000).

<b>Species/Guild</b>	<b>Non-degraded Sedge Meadows</b>	<b>Degraded Sedge Meadows</b>	<b>Degraded Salt Marsh</b>
Dunlin	3.73	0.77	7.83
Herring Gull	2.92	2.70	4.78
Lapland Longspur	4.22	0.77	1.30
Savannah Sparrow	7.47	4.25	0.00
Semipalmated Sandpiper	4.06	2.32	0.87
Sedge-meadow Shorebirds	9.42	3.09	10.87
Tundra-nesting Passerines	12.82	5.41	1.74

Figure 1. Location of the Nestor One Study Area within the Hudson Bay area, in relation to Churchill, Manitoba

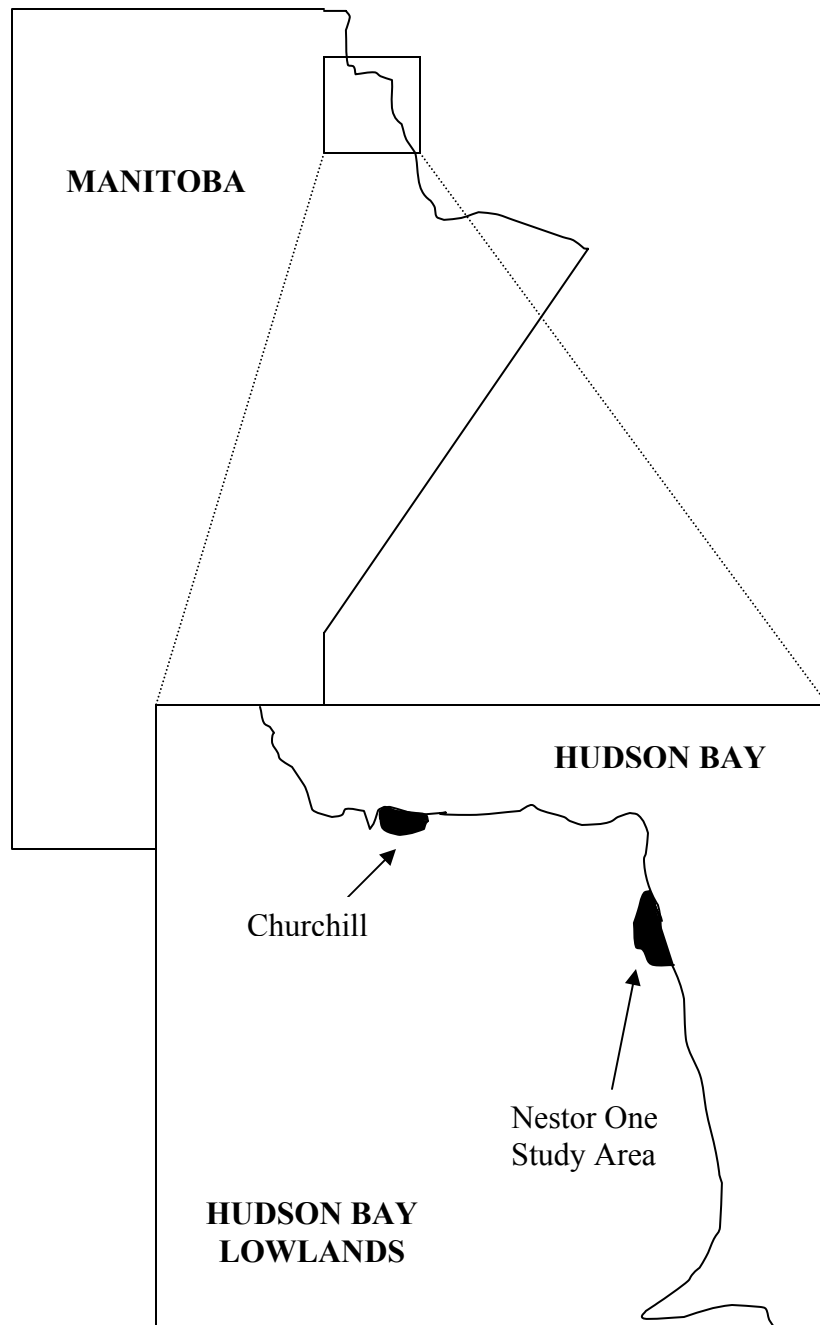
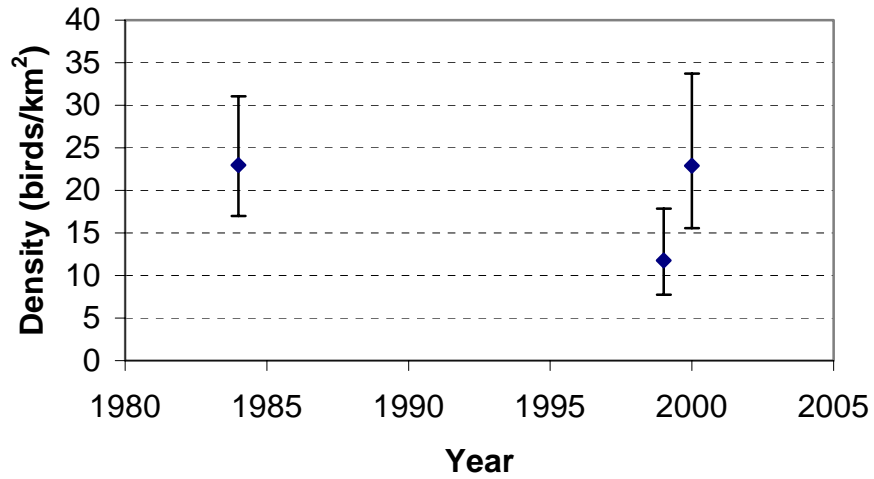




Figure 2. Line transect density estimates and corresponding 95% confidence intervals for individual species and guilds in 1984, 1999, and 2000 at Cape Churchill, Manitoba.

### DUNLIN



### HERRING GULL

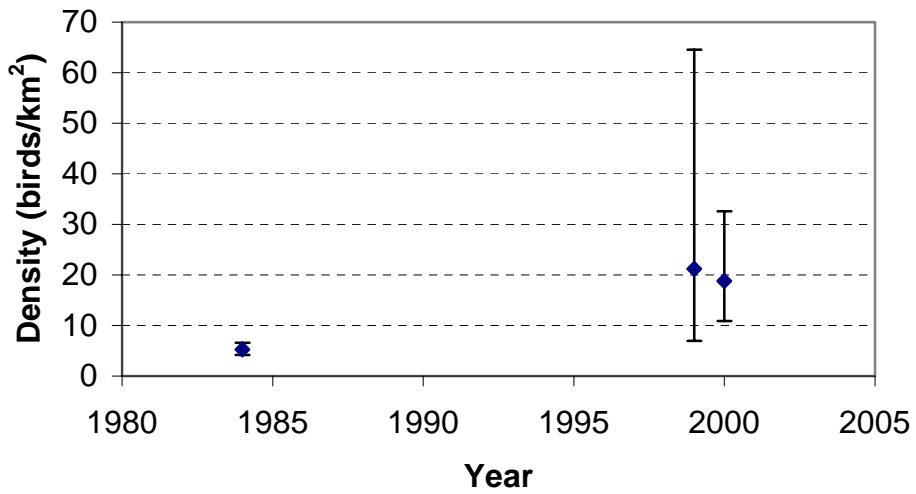
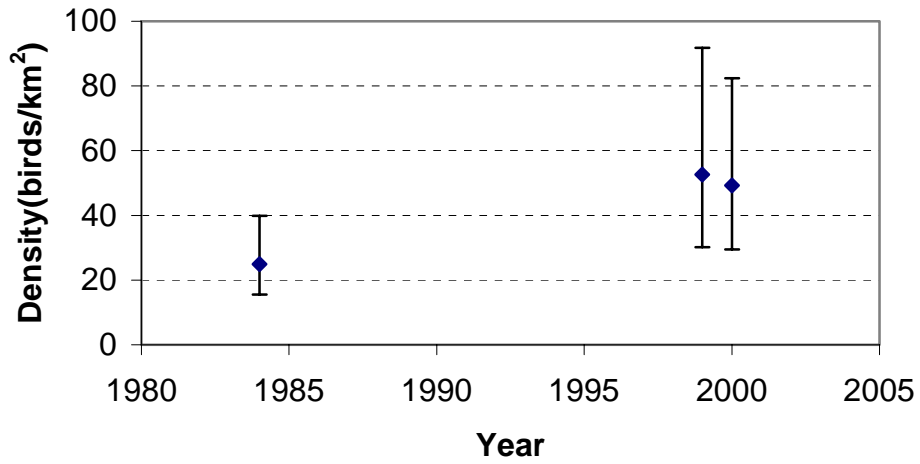


Figure 2. Continued

### HORNED LARK



### LAPLAND LONGSPUR

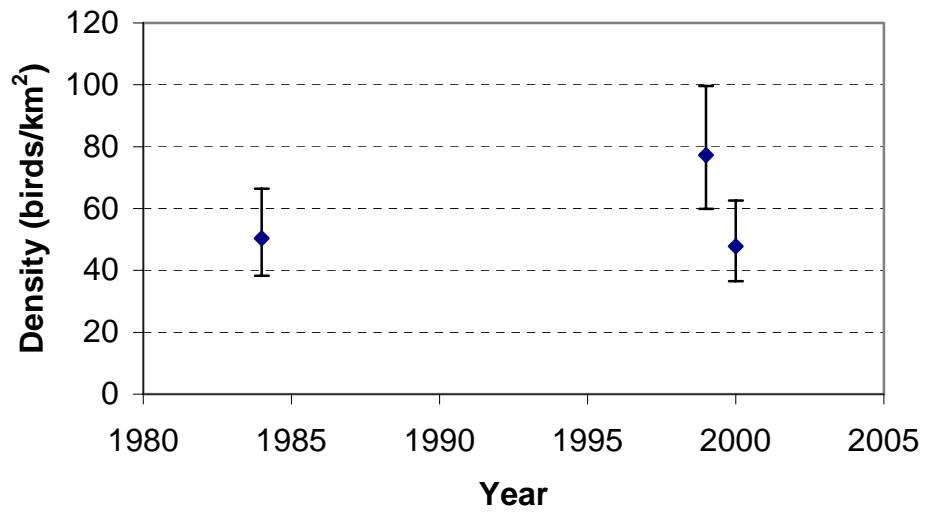
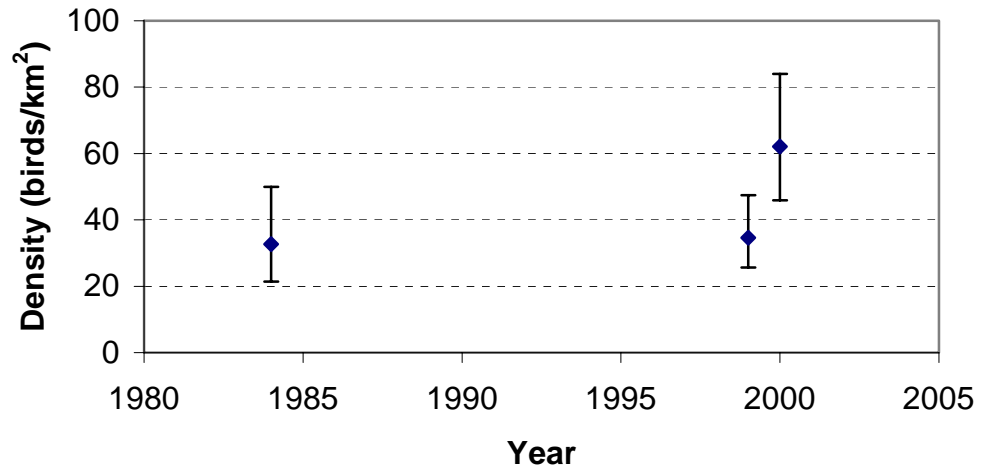


Figure 2. Continued

### SAVANNAH SPARROW



### SEMIPALMATED SANDPIPER

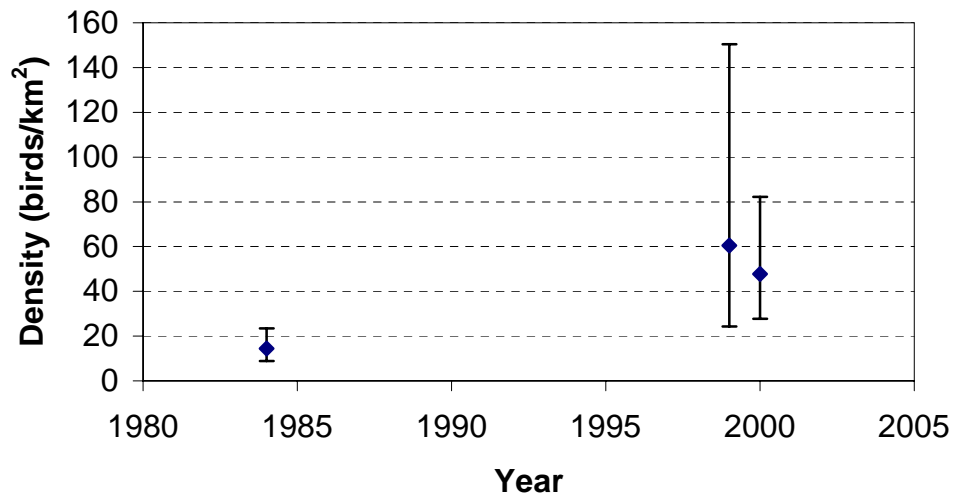
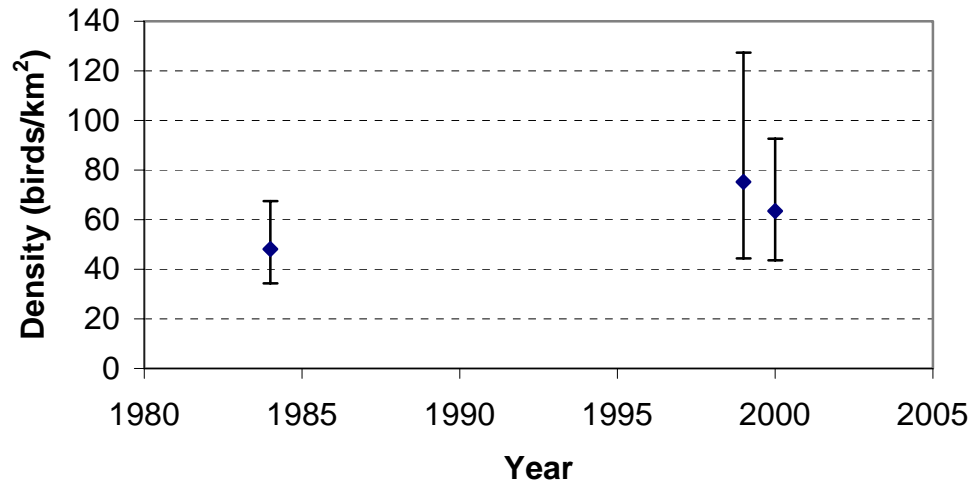


Figure 2. Continued

### SHOREBIRDS



### TUNDRA PASSERINES

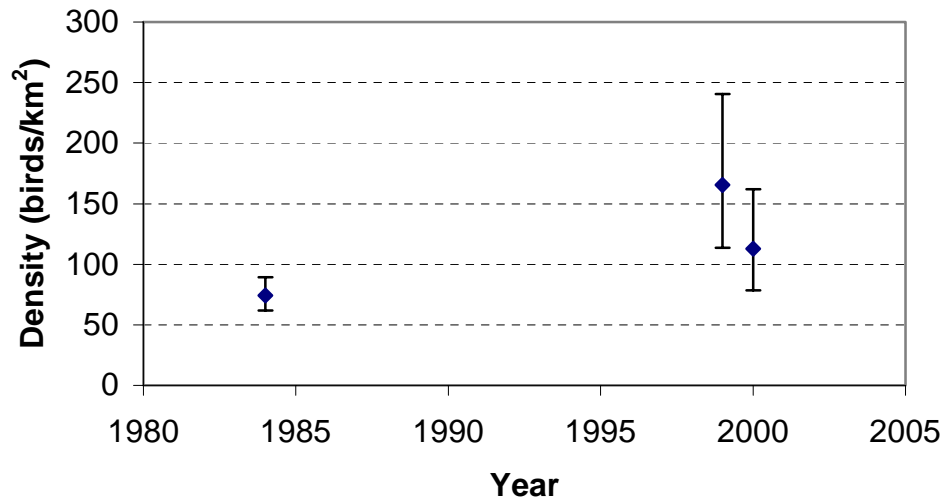
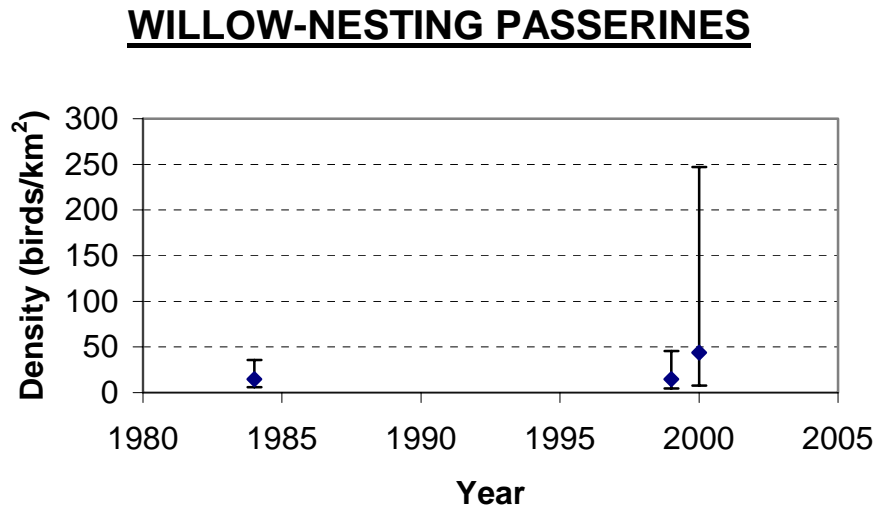


Figure 2. Continued



## **CHAPTER TWO**

**EFFECTS OF SNOW GOOSE HABITAT DEGRADATION ON FORAGING  
EFFICIENCY OF TWO SHOREBIRD SPECIES AT CAPE CHURCHILL,  
MANITOBA**

## ABSTRACT

We recorded foraging behavior of dunlins (*Calidris alpina*) and semipalmated sandpipers (*Calidris pusilla*) during June 2000 at the Nestor 1 study site outside of Churchill, Manitoba. Bird foraging behavior (number of probes, captures) was recorded and compared between freshwater sedge meadows that were degraded or non-degraded from lesser snow goose (*Chen caerulescens caerulescens*) habitat damage. Vegetative characteristics were measured at foraging sites, and foraging behaviors were regressed on vegetative characteristics to determine any relationships between foraging efficiency and extent of degradation.

No significant relationships occurred between any foraging and habitat variables for dunlins or semipalmated sandpipers. A lack of relationship between foraging behaviors and habitat characteristics could result from one of two reasons. First, due to the cold weather that extended through much of the breeding season on the study site, invertebrate growth was likely low and delayed. Therefore, damaged and non-damaged sites may have had equally low densities of invertebrates, simply due to climatic conditions. Second, it is likely that degraded habitats may still provide birds with adequate densities of food, resulting in no differences between foraging efficiency between degraded and non-degraded habitats.

## INTRODUCTION

Shorebirds have high energy requirements during the breeding season.

Shorebirds have precocial young and therefore invest large reserves of energy in egg laying (Proctor and Lynch 1993). Females are estimated to use between 82-128% of their basal metabolic rate in clutch production, which results in a large energy expenditure in eggs (Carey 1996). Males also have significant energy demands (e.g. territorial defense and breeding displays). Additionally, incubation in both sexes can also add to already large energy requirements and expenditures (Williams 1996). Therefore, to obtain energy adequate for breeding activities, shorebirds spend much of their time foraging during egg-laying and incubation stages.

Tundra-breeding shorebirds primarily forage on aquatic invertebrates within substrates of salt marshes or wet sedge meadows (Johnsgard 1981, Vacek 1999). Although strategies differ among species, shorebirds typically probe into soft substrate or feed on surface insects in these areas (Johnsgard 1981). Tundra invertebrate densities are known to change between varying habitat conditions and types (Milakovic 1999, Vacek 1999). Additionally, research has shown that foraging efficiencies of shorebirds can change with alterations in prey density (Goss-Custard 1970, 1977; Puttick 1979). Therefore, changes in habitat could affect invertebrate populations, which in turn affect shorebird foraging. Shepherd and Boates (1999) reported that semipalmated sandpipers (*Calidris pusilla*) have different foraging efficiencies in commercial baitworm harvest areas and non-harvest areas. In addition, many other studies report lower abundance of shorebirds in areas of habitat alteration, due to lower invertebrate abundance in these



areas (Colwell and Landrum 1993). Therefore, habitat alterations can greatly change foraging behavior and efficiency for shorebirds.

Recently, habitat alteration in the form of sedge meadow wetland degradation has occurred in various locations on tundra breeding grounds near lesser snow goose (*Chens caerulescens caerulescens*) breeding colonies and staging areas. Over the past thirty years, mid-continental populations of lesser snow geese have increased significantly (Ankney 1996). Increasing populations have resulted in intense foraging pressure on tundra breeding grounds, which has led to high levels of destructive herbivory to tundra vegetation (Kerbes et al. 1990, Ganter et al. 1996, Abraham and Jefferies 1997). Plant communities eventually become stripped of typical vegetative communities or grow a moss carpet that is often subsequently destroyed by geese (Jefferies 1988, Ganter et al. 1996, Abraham and Jefferies 1997). This damage alters the revegetation properties of the area, substrate salinity, and substrate hydrology (Kerbes et al. 1990, Ganter et al. 1996, Srivastava and Jefferies 1996, Abraham and Jefferies 1997).

These substrate and microsite changes have the potential to change the composition of the invertebrate community (Milakovic 1999). At the La Pérouse Bay snow goose colony outside of Churchill, Manitoba, studies demonstrate that invertebrate communities change significantly among plots of varying levels of degradation (Milakovic 1999, Vacek 1999). Therefore, changes in invertebrate communities due to habitat damage from snow geese could potentially impact other fauna dependent on invertebrates for a major portion of their diet. This includes tundra-nesting shorebirds. The goal of this study was to determine if dunlin (*Calidris alpina*) and semipalmated

sandpiper foraging efficiency was affected by potential invertebrate community changes due to snow goose damage.

## **METHODS**

### **Study Area**

The Cape Churchill study area is located 35 km east of Churchill, Manitoba and 15 km from the La Pérouse Bay snow goose colony core in Wapusk National Park (Figure 1). Base camp is located at the Nestor 1 field station, approximately 2 km from the coast of Hudson Bay, where Canada goose (*Branta canadensis*) productivity has been studied since 1968 (Walter 1999).

The study area lies within the Hudson Bay Lowlands physiographic region, which is characterized by dominance of the tundra biome. The area is composed of a series of coastal marshes, willow (*Salix spp.*) and spruce (*Picea spp.*) patches, and beach ridges. The surrounding matrix of the landscape consists of large, permanent freshwater lakes, ephemeral freshwater pools, and freshwater sedge meadows. Existing landscape features, such as permafrost, minimal relief, and permanent water bodies, are a result of Pleistocene glaciation events. Isostatic rebound, the uplifting of land resulting from glacial retreat, is still occurring at a rate of 1-1.5 m/century (Hunter 1970).

All data were collected from freshwater sedge meadow communities in varying states of damage. These communities are normally characterized by the presence of sedges (*Carex spp.*, mainly *Carex aquatilis*). Data were collected from sedge meadows that had no to little apparent damage, as well as from areas with severe damage. Areas with severe damage were characterized by a lack of vegetation or the presence of a moss carpet (Bryozoa) (Jefferies 1988). Over the whole study site, much of the freshwater

sedge meadows range from slight to moderate degradation, with pockets of severe degradation.

## **Foraging Behavior**

I followed dunlins and semipalmated sandpipers in habitats of varying degradation and recorded foraging behaviors. Initially, portions of the study site were chosen each day to search for foraging birds. Areas with little to no apparent habitat damage were searched every other day, whereas areas with severe damage were searched on intervening days. Searches and observations occurred from 18 to 25 June 2000. Behavior observations were only made on days with no precipitation and/or fog.

I used focal animal sampling (Altmann 1974) to obtain foraging behavior data. When a bird was located during searches, the observer attempted to approach the bird without disturbing it. Usually, the observer could approach the bird within 15-25 m. Once the observer was in an appropriate position, the observer watched the bird through 8 x 42 Leica binoculars for at least one minute. All behaviors were noted and recorded into an Olympus dictation recorder. Noted behaviors were characterized as probes (probe or attempt to obtain a food item), captures (obtaining a food item), movement (walking or running from one foraging site to another), preening (maintenance of feathers or bathing), territorial behavior (calling, singing, or conspecific displaying), and miscellaneous behaviors. Data on the species of bird, general locality, date, time, and presence of other birds were also taken. At the end of a foraging observation, data on the distance from the observer to the bird at the start of the observation and GPS location were recorded.

Foraging behavior tapes were transcribed to data sheets. In particular, the number of probes and captures were counted as well as the number and duration of movement,

preening, and territorial events. For both species, the amount of time spent foraging was computed for each bird. Numbers of probes and captures were divided by foraging time to obtain a standardized rate of feeding for comparison. For dunlins, we also computed the total number of foraging bouts and the amount of time spent in a bout. A foraging bout was defined as a consecutive group of probes and captures, with at least five movements involved before a one second rest. Because dunlins forage quickly and in stints of probes, we felt that the number of foraging bouts could also be a good indicator of foraging efficiency for this species. Studies have also shown that probe rates change with varying conditions and that number of probes can be correlated with number of prey captures (Goss-Custard 1984, Staine and Burger 1994, Shepherd and Boates 1999). Therefore, measuring number of probes can be a good index of an individual's ability to capture prey.

### **Habitat Assessment**

Once foraging behavior was recorded, a GPS location was taken at the site and the area was marked with colored flagging. Sites were revisited in late June and early July to obtain habitat data. Habitat was quantified with the use of a 1 x 1 m wooden frame. The frame had a grid of 25 intersection points, evenly spaced over the frame. At the foraging site, the grid was placed down, and the number of points touching a vegetative type was counted. These included: bare ground, moss, beach ridge, sedge, other plant material, ephemeral water, and permanent water. Additionally, the number of hummocks, number of goose droppings, and the presence or absence of grazing, grubbing, or sedge shoot pulling in the 1 x 1 m grid was recorded. Number of hummocks was recorded, because hummocks are conspicuously absent from degraded areas (J.

Sammler, personal observation). We estimated goose droppings as an index of goose use of the site. These data were also collected at three other surrounding sites. These sites were located 10 m at 0°, 120°, and 240° from the feeding site. In analysis, all data from the four measurement sites were pooled to characterize the foraging area. Any ephemeral and permanent water was eliminated from the analysis. Percentages of particular habitat types were computed based on total number of times an intersection hit a habitat over the total non-water intersections.

### **Data Analysis**

Foraging data were regressed on habitat data using simple linear regression (Sokal and Rohlf 1995). Residual plots were checked for normality using a Wilk-Shapiro statistic. For plots that did not show normality, a log transformation was used to correct for normality. If a transformation was performed, homoscedasticity and linearity were tested again using a score test and a F lack-of-fit test, respectively.

### **RESULTS**

No significant relationships were evident between any foraging and habitat variables for dunlins. Probes regressed on moss ( $F_{1,65} = 0.04$ ,  $P = 0.8348$ ), bare ground ( $F_{1,65} = 0.28$ ,  $P = 0.5980$ ), moss and bare ground combined ( $F_{1,65} = 0.10$ ,  $P = 0.7527$ ), non-moss vegetative cover ( $F_{1,65} = 0.12$ ,  $P = 0.7309$ ), and sedge ( $F_{1,65} = 0.37$ ,  $P = 0.5458$ ) all showed no significant relationship. For captures, nonsignificant relationships also existed for moss ( $F_{1,65} = 0.36$ ,  $P = 0.5511$ ), bare ground ( $F_{1,65} = 0.92$ ,  $P = 0.3407$ ), moss and bare ground combined ( $F_{1,65} = 0.14$ ,  $P = 0.7109$ ), non-moss vegetative cover ( $F_{1,65} = 0.11$ ,  $P = 0.7437$ ), and sedge ( $F_{1,65} = 0.07$ ,  $P = 0.7939$ ). There were no significant

relationships between probe rates and number of goose droppings ( $F_{1,65} = 0.99$ ,  $P = 0.3226$ ) or the number of hummocks ( $F_{1,65} = 0.04$ ,  $P = 0.8420$ ).

For semipalmated sandpipers, no significant relationships were evident between foraging habits and degradation variables. Probes regressed on moss ( $F_{1,68} = 0.14$ ,  $P = 0.7122$ ), bare ground ( $F_{1,68} = 0.04$ ,  $P = 0.8478$ ), moss and bare ground combined ( $F_{1,68} = 0.54$ ,  $P = 0.4641$ ), non-moss vegetative cover ( $F_{1,68} = 0.15$ ,  $P = 0.7021$ ), and sedge ( $F_{1,68} = 0.28$ ,  $P = 0.5982$ ) exhibited no significant relationships. Similarly for captures, no significant relationships existed for moss ( $F_{1,49} = 0.93$ ,  $P = 0.3394$ ), bare ground ( $F_{1,49} = 0.02$ ,  $P = 0.8851$ ), moss and bare ground combined ( $F_{1,49} = 2.22$ ,  $P = 0.1408$ ), non-moss vegetative cover ( $F_{1,49} = 3.68$ ,  $P = 0.0593$ ), and sedge ( $F_{1,49} = 0.81$ ,  $P = 0.3723$ ).

## **DISCUSSION**

We found no significant relationships with foraging behavior and efficiency of shorebirds to habitats with varying degrees of snow goose damage. There are two possible reasons for this outcome. First, no relationship could occur due to the phenology of the data collection. Data were collected in mid- to late June, which corresponds with the height of breeding season. However, in 2000, an unusual cold spell extended into the breeding season on the study area. Snow and ice did not completely melt from the area until early July. Personal observations from many individuals indicated that breeding season phenology was three weeks later than usual. The resulting cold inhibited plant growth and may have affected invertebrate larval development. Larval invertebrate development can be highly dependent upon existing climate and resources (Danks and Oliver 1972). In a side study, we took invertebrate cores and found surprisingly low

densities of invertebrates in habitats with varying damage levels (J. Sammler, unpublished data). Therefore, the extreme cold could have halted larval invertebrate growth, making all habitats essentially equivalent in terms of available invertebrate biomass. This may explain why no significant relationships existed.

Second, no significant relationships may exist, because there are no relationships between habitat damage and foraging efficiency. It is quite possible that damaged areas still harbor high to moderate densities of invertebrates for shorebirds to forage upon. Shorebirds are still seen foraging, even in the most damaged areas (Abraham and Jefferies 1997). Damage may only change the types of organisms that live in a microsite, rather than severely altering organism abundance or biomass. It has also been shown that invertebrate abundance can be extremely high in degraded habitats and often higher than at non-degraded sites (Vacek 1999).

Lack of relationship between foraging and habitat quality indicates damage to habitats does not necessarily cause significant problems for shorebirds in search for food resources. Shorebirds still feed in damaged areas, particularly during migratory stopovers (J. Sammler, personal observation). However, they are noticeably absent from these sites during breeding seasons. The reason for this absence could be attributed to loss in nesting habitat, in particular. Birds are unable to nest successfully in damaged areas, due to a complete lack of nesting substrates. Therefore, birds perhaps do not use degraded areas as readily not because of decreased food resources but due to lack of nesting habitat and low probability of encountering conspecifics.

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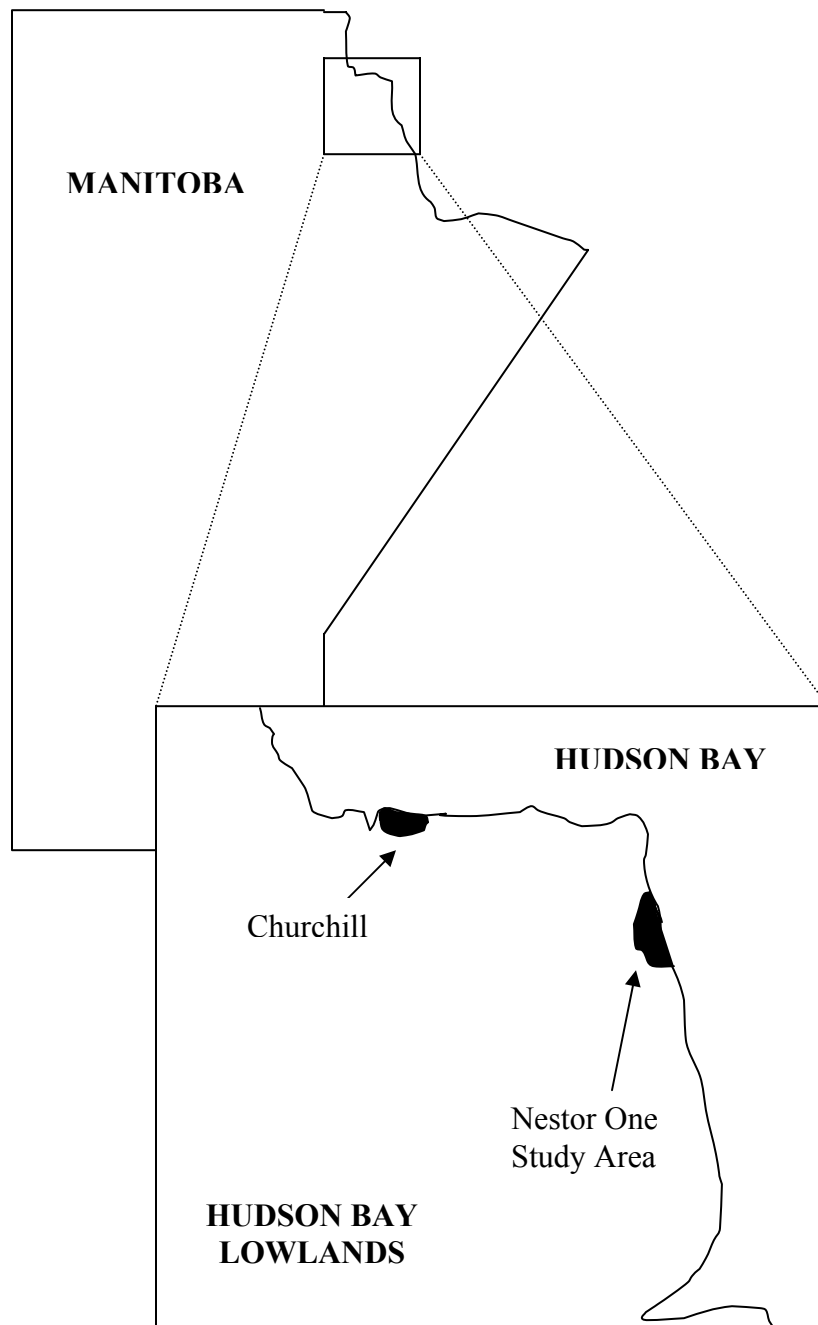
Table 1. Results of regressing dunlin foraging behavior data (dependent variable) upon habitat variables (independent variable) at the Nestor 1 study site, Churchill, Manitoba (June 2000).

<b>Dependent Variable</b>	<b>Independent Variable</b>	<b><i>F</i></b>	<b>d.f.</b>	<b>P-value</b>
Probes	Moss Cover	0.04	1,65	0.8348
	Bare Ground Cover	0.28	1,65	0.5980
	Moss and Bare Combined Cover	0.10	1,65	0.7527
	Non-Moss Vegetative Cover	0.12	1,65	0.7309
	Sedge Cover	0.37	1,65	0.5458
	Number of Goose Droppings	0.99	1,65	0.3226
	Number of Hummocks	0.04	1,65	0.8420
Captures	Moss Cover	0.36	1,65	0.5511
	Bare Ground Cover	0.92	1,65	0.3407
	Moss and Bare Combined Cover	0.14	1,65	0.7109
	Non-Moss Vegetative Cover	0.11	1,65	0.7437
	Sedge Cover	0.07	1,65	0.7939

Table 2. Results of regressing semipalmated sandpiper foraging behavior data (dependent variable) upon habitat variables (independent variable) at the Nestor 1 study site, Churchill, Manitoba (June 2000).

<b>Dependent Variable</b>	<b>Independent Variable</b>	<b><i>F</i></b>	<b>d.f.</b>	<b>P-value</b>
Probes	Moss Cover	0.14	1,68	0.7122
	Bare Ground Cover	0.04	1,68	0.8478
	Moss and Bare Combined Cover	0.54	1,68	0.4641
	Non-Moss Vegetative Cover	0.15	1,68	0.7021
	Sedge Cover	0.28	1,68	0.5982
Captures	Moss Cover	0.93	1,49	0.3394
	Bare Ground Cover	0.02	1,49	0.8851
	Moss and Bare Combined Cover	2.22	1,49	0.1409
	Non-Moss Vegetative Cover	3.68	1,49	0.0593
	Sedge Cover	0.81	1,49	0.3723

Figure 1. Location of the Nestor One Study Area within the Hudson Bay area, in relation to Churchill, Manitoba



## APPENDIX A. Bird species guilds used in analyses.

### Shorebird Guild

Semipalmated Plover (*Charadrius semipalmatus*)  
Hudsonian Godwit (*Limosa haemastica*)  
Dunlin (*Calidris alpina*)  
Semipalmated Sandpiper (*Calidris pusilla*)  
Least Sandpiper (*Calidris minutilla*)  
Pectoral Sandpiper (*Calidris melanotos*)  
Stilt Sandpiper (*Calidris himantopus*)  
Red-necked Phalarope (*Phalaropus lobatus*)

### Tundra-nesting Passerines

Horned Lark (*Eremophila alpestris*)  
Lapland Longspur (*Calcarius lapponicus*)  
Savannah Sparrow (*Passerculus sandwichensis*)  
Snow Bunting (*Plectrophenax nivalis*)

### Willow-nesting Passerines

Yellow Warbler (*Dendroica petechia*)  
American Tree Sparrow (*Spizella arborea*)  
White-crowned Sparrow (*Zonotrichia leucophrys*)  
Common Redpoll (*Carduelis flammea*)  
Hoary Redpoll (*Carduelis hornemanni*)