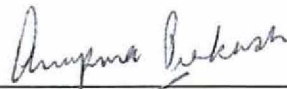


RECENT CHANGES IN PLANT AND AVIAN COMMUNITIES  
AT CREAMER'S REFUGE, ALASKA USING FIELD  
AND REMOTE SENSING OBSERVATIONS

By

Lila Maria Tauzer

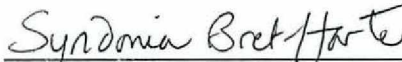
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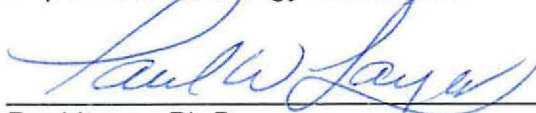


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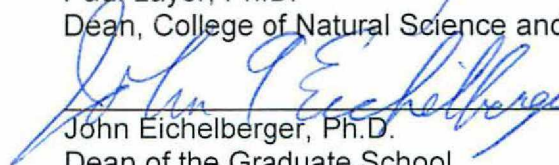


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RECENT CHANGES IN PLANT AND AVIAN COMMUNITIES  
AT CREAMER'S REFUGE, ALASKA USING FIELD  
AND REMOTE SENSING OBSERVATIONS

A  
THESIS

Presented to the Faculty  
of the University of Alaska Fairbanks

in Partial Fulfillment of the Requirements  
for the Degree of

MASTER OF SCIENCE

By  
Lila Maria Tauzer, B.S.

Fairbanks, Alaska

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## **Abstract**

Plant communities in the north are being profoundly altered by climate warming, but our understanding of the extent and outcomes of this ecosystem shift is limited. Although we assume local vegetation changes will affect avian communities, few data exist to investigate this relationship. In an interior Alaska boreal forest ecosystem, this study capitalized on available resources to assess simultaneous change in plant and avian communities over 35 years. I quantified biological change in summer avian community data (species composition, diversity, and richness) and in vegetation using archived field data, and supplemented this data with remote sensing observations for a similar time period to assess the validity of this method for documenting environmental change. Field and remote sensing data both documented successional changes resulting in denser, more coniferous-dominated habitats. Birds responded accordingly, which indicates a rapid avian response to habitat change and that they are good indicators of environmental change. Information gained provides more accurate evaluations of habitat dynamics throughout the interior boreal forest and highlights the importance of considering successional change in all long-term climate studies. It allows for better predictions of future habitat change and acts as a strong baseline for future environmental monitoring.

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## GENERAL INTRODUCTION

Vegetation changes in northern North America have been associated with recent climate warming but we have a poor understanding of the rate of vegetation change or of the cascading ecosystem effects (ACIA, 2005; IPCC, 2007). In the Arctic, where the majority of climate change research has occurred to date, studies have documented recent treeline shifts, the conversion of tundra to forest, and shifts towards taller and denser vegetation (Hinzman et al., 2005). Although less is known about the expansive boreal forest region, growing evidence shows that these forests are also undergoing changes, both directly due to anthropogenic events, and indirectly due to temperature changes (Chapin, 2006). Warming temperatures have the potential to affect boreal forests by increasing the frequency and severity of fires (Chapin et al., 2008; Kasischke et al., 2010), altering wetlands (Osterkamp et al., 2000; Jorgenson and Osterkamp, 2005), increasing plant growing season (Delbart et al., 2005), and changing plant community compositions (Barrett et al., 2011).

Succession is a natural process with real ecosystem repercussions, but the dynamics of change in the boreal forest are surprisingly complex. Although only six tree species are found in the boreal forest of interior Alaska, more than 30 forest types (Viereck et al., 1992) result from multiple successional trajectories (Kurkowski et al., 2008). Species dominance and transition rates between successional stages depend on a number of biotic and abiotic factors (Viereck, 1970; Van Cleve et al., 1996). In undisturbed areas, we generally expect successional change to be characterized by gradual densification of slow-growing (coniferous) trees, but there is also evidence that boreal forests regularly experience threshold changes, or turning points, that result in rapid transition from one state to another (Chapin et al., 2004). Perhaps because of this complexity, successional change in vegetation is often overlooked in predictive models of future change in the boreal forest.



Many questions remain about the spatial distribution of change and which habitats are most vulnerable. Global warming is predicted to be the most ecologically disruptive in the north (Post et al., 2009), but change has not been consistent across northern latitudes. In the Arctic, recent warming has been associated with an overall positive, or “greening”, trend (Stow et al., 2004), while most coniferous forests in the boreal forest have shown an opposite “browning” trend (Lloyd and Bunn, 2007; Beck and Goetz, 2011). Even within the boreal forests, habitats are thought to be responding differently to warmer temperature (Calef et al., 2005; Danby and Hik, 2007; Baird et al., 2012). It appears that landscape placement, biotic interactions, and local climate conditions are important factors that affect the rate, magnitude, and even direction of vegetation change (Stueve et al., 2011; Roland et al., In press).

Even less is known about how boreal fauna respond to local or regional changes. Vegetation characteristics directly determine animal use (Fortin and Dale, 2005); thus, the distribution and viability of many species and natural communities are predicted to be affected as habitats change due to global climatic change (Parmesan, 2006; Pimm, 2009). Birds, in particular, rely on plants for food, shelter, protection from predators, nest building supplies, breeding and/or courtship sites, and have close habitat associations (Furness and Greenwood, 1993). Most North American birds are extremely mobile throughout their annual cycle, meaning that they are capable of rapidly shifting areas of use if habitat availability changes. Indeed, successional vegetational changes in plant communities have often been shown to cause corresponding changes in bird communities (Brawn et al., 2001; Holmes and Sherry, 2001; Schieck and Song, 2006).

We assume that warming temperatures will impact boreal birds because the major habitat components for birds – vegetation structure, prey availability,

degree of wetness – are subject to alteration as climate changes. Unfortunately, we know too little to test this assumption due to an overall paucity of data. Continent-wide population declines have been observed for many boreal-breeding species, yet very few datasets are available that link population changes to observed vegetation changes on breeding grounds. In Alaska, even basic knowledge on the current distribution, abundance, and habitat associations of boreal forest birds (Handel et al., 2009) is deficient. The poor understanding of ecological change in the boreal forest region is especially problematic for managers charged with stewardship of land and species, particularly because species that occur near the edge of their geographic range are the first to be impacted by climate changes (Parmesan, 2006).

This project came about as a collaborative effort between University of Alaska Fairbanks (UAF) researchers and the staff at Alaska Bird Observatory (ABO). There is growing interest within the research community in taking an ecosystem approach to examine climate change effects and ABO was, at the time, committed to forming a long-term, citizen-science based ecological and climate monitoring site at Creamer's Field Migratory Waterfowl Refuge (Creamer's Refuge) in the heart of interior Alaska boreal forest. ABO was interested for several reasons: they wanted to expand their existing avian monitoring to more general monitoring protocols with broader implications, and they wanted to retrospectively examine recent change in order to parse out factors influencing trends in their 20+ years of existing avian demographic data. This information would be used to educate the Fairbanks community about local changes already observed and, perhaps, would allow for more accurate predictions of avian populations in the future.

With this in mind, I became aware of a previous study done at Creamer's Refuge. In 1975, a UAF graduate student collected avian abundance and plant

community data for his MS thesis entitled, “Ecological Survey of Birds, Mammals, and Vegetation of Fairbanks Wildlife Management Area” (Spindler, 1976).

Although his data were collected for a different purpose, replicating this study after 35 years presented a unique opportunity to document recent ecosystem changes in an area that is both poorly understood and thought to be experiencing rapid landscape transformations. It is informative in light of climate warming, but mostly important because there are so many questions yet unanswered regarding ecosystem change in boreal forest.

Given the impetus from ABO, I addressed two major knowledge gaps regarding ecosystem change in interior Alaska boreal forest (Chapter 1). First, I examined successional change over time at Creamer’s Refuge and answered the questions: what type of vegetation changes occurred over the last three decades at Creamer’s Refuge? Which habitats changed the most? Were the observed changes compatible with expectations of successional change in boreal forest? Second, I addressed the underlying assumption that avian communities change in concordance with habitat availability by answering: what type of changes in avian abundance and community composition occurred over the same 35 year period? How closely related are the changes observed in birds to those in vegetation? More broadly, I addressed whether “natural” succession acts as a driver for avian population change and if birds are indeed good indicators of environmental change in the boreal forest of interior Alaska.

Finally, because much of the available climate response data comes from remotely-sensed data with little validation from field or historical data, I carried out a remote-sensing analysis of vegetation change over time at Creamer’s Refuge (Chapter 2). Remote sensing analysis is attractive because large areas can be assessed in less time and cost than field work, but questions remain regarding the appropriate scale of analysis, what habitats are adequately

mapped, and if subtle successional changes can be accurately documented. Because Creamer's Refuge is much smaller than usually analyzed (area examined = 1100 ha) and easily accessible, I was able to assess the ability to isolate habitat classes using remote sensing and validate these results through ground-truthing. The primary purpose here was to evaluate the efficacy of this often-used method for documenting habitat change at a local scale, in an area where the nature of recent change is known due to site knowledge and field experience.

Understanding the impacts that vegetation change will have on ecological communities is particularly challenging because so little is known in the north, thus this study is an efficient and timely use of pre-existing data. Successional change is inevitable, and efficiency in our efforts to monitor and manage habitats is especially important at this critical time of shifting environmental conditions and plant and animal ranges; to do this, we need to capitalize on what little data we have and regularly assess the methods used to do so. Despite its small size and limited inference, I hope that knowledge gained from this research at Creamer's Refuge provides some insight into ecosystem changes in the Alaskan boreal forest.

## CHAPTER 1: IMPACT OF SUCCESSION ON THE AVIAN COMMUNITY IN AN ALASKAN BOREAL FOREST<sup>1</sup>

### 1.1 ABSTRACT

Recent changes in plant and avian community composition have been linked to accelerated warming at northern latitudes. Despite the underlying assumption that avifauna will be dramatically affected by local habitat change, few studies have taken an ecosystem approach, primarily due to a lack of comprehensive baseline data for such comparisons. In the boreal forest of interior Alaska, we capitalized on available resources to simultaneously assess change in vegetation and birds over the past 35 years. Using archived field data to compare with data collected in 2010-2011, we quantified habitat change for five 10-ha plots representing several boreal habitat types. At this same local scale, we used territory mapping to compare current avian community composition and abundances with historical data. We observed rapid successional changes and an increase in forest birds as the landscape became more forested, as well as a substantial decrease in shrub and wetland-associated bird species. Overall, avian species diversity and abundances declined since the 1970s. Townsend's Warbler (*Setophaga townsendi*) is currently found breeding there, while Gray-cheeked Thrush (*Catharus minimus*) and American Tree Sparrow (*Spizella arborea*) disappeared completely from the study area. These findings give insight into avian response patterns in the rapidly changing boreal forest, while highlighting the importance of understanding avian-habitat dynamics. Succession is one of the many drivers of avian community change and habitat impacts should be considered in all long-term monitoring plans.

<sup>1</sup>Tauzer, L. M., A. N. Powell, and S. Sharbaugh. 2013. Impact of succession on the avian community in an Alaskan boreal forest. Prepared for submission to *Avian Conservation and Ecology*.

## 1.2 INTRODUCTION

North American boreal forests are currently undergoing rapid transformations as a result of both direct anthropogenic actions (particularly oil/ hydroelectric development and logging) and indirect, climate-induced changes to the landscape (Chapin, 2006), but little is known about the cascading ecosystem effects of these changes (Cheskey et al., 2011). There is potential, however, that change in the boreal will greatly impact bird populations because this biome supports an estimated 30% of all North American breeding birds and more than 300 species (Wells, 2011). Widespread, and sometimes rapid, avian population declines have been documented in recent years (Sauer et al., 2011), although no simple explanation has been found. Causes of declines may include loss of wintering grounds, changes to or loss of forest habitats along migratory routes, and changes in habitat characteristics on the breeding grounds (NABCI, 2009).

Climate variables influence avian population trends and there is evidence of a strong response to climate warming (Leech and Crick, 2007; National Audubon Society, 2009; NABCI, 2010; Knudsen et al., 2011). Climate change has been attributed to reduced reproduction and survival (Both et al., 2006; McClure et al., 2012), as well as shifts in phenology (Gordo, 2007; Macmynowski et al., 2007) and geographic range (Thomas and Lennon, 1999; Hitch and Leberg, 2007). Models predict even more dramatic population and range changes in the near future, particularly at northern latitudes where warming trends have been most pronounced (Huntley et al., 2006; Jetz et al., 2007; Lawler et al., 2009).

Climate change is predicted to be the most ecologically disruptive in the Arctic and boreal regions (ACIA, 2005; IPCC, 2007; Post et al., 2009). These areas have already exhibited substantial warming (Barber et al., 2008; Wendler and Shulski, 2009) and vegetation shifts have been documented (Hinzman et al., 2005; Tape et al., 2006; Danby and Hik, 2007a). Areas of the boreal forest,

including western North America and northern Eurasia, warmed more rapidly than any other region since the 1970s (Chapin, 2006), and enhanced warming of permafrost in interior Alaska has resulted in shifts in hydrology and broad scale changes in forests and wetlands (Osterkamp et al., 2000; Jorgenson et al., 2001; Riordan et al., 2006). It is assumed that temperature increases at northern latitudes will lead to quicker turnover between successional stages; however, questions remain about how different habitat types will respond, and the speed of transition between stages (Chapin, 2006; Hollingsworth et al., 2010; Wolken et al., 2011). Unfortunately, in the majority of long-term avian studies designed to assess population trends, including nearly all studies of climate impacts, successional change is overlooked. The assumption that environmental conditions are consistent over time is considered a major flaw in the interpretation of trends derived from many long-term monitoring programs such as the Breeding Bird Survey and Christmas Bird Count (Keller and Scallan, 1999; Betts et al., 2007).

Another problem with assessing mechanisms of avian population trends is that historical data are often lacking or inadequate. This is especially true of the western boreal forest, where even basic knowledge of avian natural history, habitat associations, and current species distributions is unknown for many species (Machtans and Latour, 2003; Handel et al., 2009). The majority of research on boreal birds is from eastern Canada or northern Alberta and has focused on the direct impacts caused by resource extraction (e.g. Schmiegelow et al., 1997; Hobson and Bayne, 2000; Leonard et al., 2008). However, extrapolation to the northwestern boreal is unreliable because bird-habitat associations are often different in Alaska than elsewhere (Kessel, 1998). For example, Hammond's Flycatchers (*Empidonax hammondi*) are considered coniferous-forest breeders elsewhere, but in Alaska they are most often associated with mature deciduous (birch) forests.

Given the strong association between birds and habitat, and the knowledge that the boreal landscape is rapidly changing, understanding successional patterns in boreal bird communities is a critical component of understanding population changes over time. For this reason, we explored the nature of ecosystem change in several habitats found in an interior Alaska boreal forest. More specifically, we related observed changes in avian communities to local vegetational changes by replicating a study conducted in the 1970s in interior Alaska (Spindler, 1976). Other studies in the boreal have examined differences in avian community between forest stands of varying ages (Hobson and Schieck, 1999; Imbeau et al., 1999; Schieck and Song, 2006), but very few studies have investigated the direct relationship between habitat change and bird populations over time, at the same location (but see Kirk et al., 1996). To our knowledge, this is the first such study from Alaska or the northwestern interior boreal forest.

We predicted that changes in vegetation would occur as expected by a successional pathway: with forested habitats changing the least, and open shrub habitats changing the most as resident trees matured. We also predicted that change in breeding bird species composition would closely mirror the magnitude of successional vegetation change. Most boreal forest bird species appear to be relatively plastic in their habitat use (Willson and Comet, 1996; Machtans and Latour, 2003); therefore, we expected that that there would be little change in overall species composition, but that local abundances would fluctuate as birds shifted across the landscape to find preferred breeding habitat nearby. Furthermore, we predicted that our observations would not correspond well with regional trends derived from Breeding Bird Survey because of differences in scale (local vs. regional).



### 1.3 METHODS

#### Study location

Our study was conducted at Creamer's Field Migratory Waterfowl Refuge (hereafter, Creamer's Refuge), a 1057-ha tract of public land located just north of the city boundary of Fairbanks, Alaska (64°49'N, 147°52'W; Fig. 1.1). The climate of Fairbanks is continental, characterized by low precipitation and extreme seasonal variation in temperature. Mean annual temperature is -3.0° C and mean average annual precipitation is 28.0 cm, which falls mostly during July and August. July is the warmest month (average temperature: 16.4° C) and January is the coldest (average temperature: -23.3° C) (ARCR, 2012). In the last century, mean annual temperature increased by 1.4° C, length of growing season increased by 45%, and the number of very low temperature days ( $\leq -40^{\circ}$  C) decreased (Wendler and Shulski, 2009).

The study site includes land managed by Alaska Department of Fish and Game, and land owned by University of Alaska Fairbanks and Fairbanks North Star Borough. Although the area is bounded by roads and easily accessible to the public, it is largely undeveloped except for dog-mushing and ski trails used primarily in the winter. A natural fire burned a small portion of the southern side of the refuge in the 1950's (Chuck Creamer, personal comm.) and refuge managers set two small fires (< 10 ha) in 1996 and 1997 to create wildlife habitat. Usage patterns and the number of trails have remained relatively stable during the past 35 years.

The low-lying flats of Creamer's Refuge are abandoned floodplains characterized by near-continuous permafrost, extensive thermokarst, and polygonization. Four small creeks flow into the area from nearby hills and provide drainage for a few small lakes (< 2 ha). Like much of interior Alaska boreal region, the landscape is a complex mosaic of moist shrublands, open woodlands of stunted black spruce

and tamarack trees, patches of dense coniferous and deciduous forests, and many small ponds.

To develop an ecological inventory of Creamer's Refuge, Spindler (1976) collected extensive plant and avian community data on five 10-ha (316 m x 316 m) plots. These plots represented a range of successional stages (mean age of oldest trees on plots:  $20.7 \pm 8.9$  to  $108.2 \pm 22.7$  years) and the dominant habitat types at Creamer's Refuge: tussock-shrub bog ("Low Shrub"), tall alder/ willow shrub ("Tall Shrub"), young seral birch forest ("Birch"), white spruce-hardwood forest ("White Spruce"), and open woodland dominated by dwarf black spruce ("Black Spruce"). Plots covered 4.1% of Creamer's Refuge, averaged 1.1 km apart, and were selected haphazardly for habitat homogeneity, using knowledge gained from ground and aerial surveillance. Because this project is a direct comparison to this previous work, we relocated these habitat plots and sampled plants and birds following Spindler's protocol in 2010-11; we sampled only a portion (4.29 ha) of the White Spruce plot because of land ownership issues.

### **Plot and point generation**

We relocated plots as best as possible using detailed field notes and a handdrawn map from 1975, which showed distances from plot boundaries to nearby features. We scanned and imported the map into ArcMAP Desktop 9.3.1 (ESRI Software, USA, 2009) then georeferenced this map, using road and trail junctions as distinctive location tie-points. Plot and subplot boundaries were digitally generated for upload to a GPS receiver. We randomly generated vegetation sampling points (see below) using the random point generation tool.

### **Field data**

#### *Habitat*

Except where noted, we collected habitat data in July and August 2010.

For characterizing each habitat plot, we divided each into a grid of 49 subplots ( $46 \text{ m}^2$ ) and collected vegetation data at three points per subplot. In one of these points, we recorded all plant species present within a 1.14-m radius; for the other two points, we counted all woody stems  $> 1 \text{ m}$  tall within a 1.14-m radius and recorded percent cover (to closest 10%) of grass, tussock, herbs, moss, lichen and dwarf shrub ( $< 0.3 \text{ m}$ ). At 21 of the 49 subplots, we identified and measured the oldest-looking tree, and determined tree age with an increment borer. Mean percent cover for all cover class variables was determined, as were mean age and diameter of the oldest-looking trees. Based on the circular plant species plots, we determined frequency of occurrence for each plant species present. These values were used to calculate relative frequency of trees and tall shrubs (see below) and Shannon's Diversity Index ( $H'$ ) for all tree, shrub, and overall plant species (grasses and cryptograms excluded). We defined frequency as the proportion of circular plots containing a specific species relative to the total number sampled ( $n = 49/\text{habitat}$ ).

At two points/ subplot (total = 98/habitat), trees and tall shrubs were sampled using a point-centered quarter method (Cottam and Curtis, 1956). We measured distance to the nearest woody stem with diameter at breast height (DBH)  $\geq 2.5 \text{ cm}$  in each of the cardinal directions and, for two directions, recorded species and DBH. From these values, we calculated mean diameter of trees, mean and total basal area, and the density of trees and tall shrubs for each 10-ha plot. We used frequency of occurrence (mentioned above) to determine relative frequency for each species of tree and tall shrub, and calculated relative density, relative dominance, and importance values for each tree and tree-sized shrub species using the following equations:

$$1. \text{ Relative density (species } x) = \frac{\text{Number of individuals (species } x)}{\text{Total number of individuals (all species)}} \times 100$$

$$2. \text{ Relative dominance (species } x) = \frac{\text{Basal area (species } x)}{\text{Total basal area (all species)}} \times 100$$

$$3. \text{ Importance value (species } x) = \frac{\text{relative frequency} + \text{relative density} + \text{relative dominance}}{3}$$

Importance value is a measure of abundance, but it is important to realize that this value is relative; it is independent of distance or absolute density per unit area.

In addition, at each of the 49 subplots, we measured depth to permafrost during the time of maximum thaw (late September-early October 2010) with a 1.1-m probe and calculated mean active layer depth. All means are presented  $\pm$  standard deviation (SD).

### *Birds*

We conducted breeding bird surveys from 15 May –1 July in 2010 and 2011 following standard territory mapping protocol (Bibby et al., 2000; University of Alaska Fairbanks IACUC #148723-1). Territory mapping involved visiting each habitat plot 6-8 times per year (average visit time: 3.25 h). Surveys had variable start times throughout the day and were conducted only on days with good visibility, low winds, and little or no precipitation. To minimize observer bias, observers rotated plots between sampling rounds.

At each visit, we recorded species, behavior, and GPS location of all birds seen or heard on the plot. Birds flying over or with questionable locations were excluded. We recorded main behavior (singing, calling, movements, counter-singing) as well as specific behavior that indicated breeding (food-carry, nest building, nest-defense). Using the GPS locations, we created polygons defining each bird's territory in ArcMap. In order to be considered a territory holder, singing birds had to be recorded  $\geq 2$  times in a localized area, with sightings  $\geq 10$  days apart. If  $> 50\%$  of the territory was on the plot or if breeding was documented, the territory received a score of 1; otherwise, it received a score of 0.5. Because our main objective was to document change in bird community

between 1975 and 2010-11 rather than generating abundance estimates, we included non-territorial birds such as Bohemian Waxwings, Redpoll spp., Sandhill Cranes, raptors, gulls, ducks and shorebirds in the analyses (see Appendix 1 for bird species names and list). Common and Hoary Redpoll were grouped because they are difficult to distinguish in the field.

For the 1975 bird data, we digitally transcribed Spindler's (1976) original field notes by entering locations into ArcMap, and then reassessed territories using the criteria noted above. For our data, we used both 2010 and 2011 to calculate a mean number of territories for each species and, for simplicity, refer to this in the results as '2011'. If anything, combining 2010 and 2011 bird data led to elevated species richness because birds seen on the plot in either year were included in the calculation of cumulative species richness (Appendix 2). For each year and habitat plot, we calculated overall abundance (total number of breeding territories for all species of birds), breeding species richness (number of species of birds holding breeding territories), and species diversity using Shannon's Diversity Index ( $H'$ ).  $H'$  was chosen because it emphasizes species richness rather than dominance. For final analyses, we grouped birds into habitat-associated avian guilds (Sharbaugh, 2007).

### **Statistical analyses**

We used R statistical software Version 2.14 (R Development Core Team, Vienna, Austria, 2011) for all statistical analyses. The "vegan" package was used to calculate dissimilarity matrices and perform ordinations.

### *Habitat*

We used unpooled t-tests to test differences in habitat types between 1975 and 2011. This was not possible, however, when we only had access to 1975 summary results rather than raw data. Diversity values ( $H'$ ) for tree, shrub, and

overall plant species were directly compared between years and habitat types. We quantified the magnitude of habitat change for each plot by using NMDS ordination of all habitat variables to generate a dissimilarity matrix. Euclidian distance values were used as measures of relative “ecological distance”, with small numbers indicating little change between 1975 and 2011 and high numbers indicating greater amounts of change.

### *Birds*

We quantitatively compared overall number of breeding bird species, species richness, and species diversity ( $H'$ ) values between years for each habitat plot (Bibby et al., 2000). Similar to habitat, we used NMDS ordination of the number of territories of each bird species to generate a dissimilarity matrix. In this case, we used the Bray-Curtis index of dissimilarity to quantify the magnitude of change in avian communities between 1975 and 2011 for each plot, because it can deal with zeros in abundance data (Magurran, 2004).

Finally, to assess correlation between the two dissimilarity matrices (habitat and birds), we performed a Mantel’s test of association.

## **1.4 RESULTS**

### **Vegetation**

Although the direction and amount of change varied by habitat type, vegetation structure changed on all plots between 1975 and 2011 (Table 1.1). Forest increased across Creamer’s Refuge, as did canopy cover and litter, while understory shrub decreased. Mean tree age increased significantly ( $P < 0.05$ ) on all plots, and there was an increase in total tree basal area as existing trees matured; this was true even for the Black Spruce plot where mean tree size (height and DBH) decreased. Tree and tall shrub (DBH > 2.5 cm) density also increased across the Refuge.

Coniferous trees became proportionately more important compared to deciduous trees and tall shrubs (Fig. 1.2). Black spruce (*Picea mariana*) was the most abundant species at Creamer's Refuge in both years, and relative abundance and density increased over time. White spruce (*P. glauca*) density also increased on all but the Birch plot, where it was still found in small numbers in the understory. Tamarack (*Larix laricina*) density declined by 78%, especially on the Black Spruce plot where it was most commonly encountered. This species disappeared completely from Tall Shrub, but young tamaracks were still found on Black Spruce and Low Shrub.

Refuge-wide, tall tree-like shrubs decreased in relative importance, medium shrub (> 1 m tall; DBH < 2.5 cm) density decreased by 38%, and dwarf shrub (< 0.3 m tall) cover decreased by 24%. Large changes in medium shrub densities were documented between sample years, with decreases recorded for closed plots (canopy cover > 15%; Tall Shrub, Birch and White Spruce) and increases for open plots (canopy cover < 15%; Low Shrub and Black Spruce). Active layer depth did not change between 1975 and 2011.

#### *Change by habitat plot*

The young Birch plot exhibited the most overall change in vegetation (Euclidian distance = 7.331), followed by Tall Shrub (Euclidian distance = 5.685), then the forested Black and White Spruce plots (Euclidian distances = 5.411 and 4.953, respectively). Low Shrub continued to have the lowest tree and tall shrub densities of all the habitats and demonstrated the least overall vegetation change (Euclidian distance = 2.215) but, visually, it changed from a "largely treeless tussock bog" (Spindler 1976) to a sparse woodland of small birch, tamarack, and spruce trees (Appendix 4A).

Examination of old photographs did not indicate visible changes on forested plots; however, visible changes were noticeable on Birch (Appendix 4C) and Tall Shrub (Appendix 4B) plots. In 1975, the Birch plot consisted largely of dense young birch (*Betula neoalaskana*) trees regrowing after a fire in the 1950s; by 2011, it was a monotypic birch forest with a tall willow (*Salix bebbiana*) understory. Tall Shrub continued to be a wet shrubland of alder (*Alnus incana* ssp. *tenuifolia*) with a dense understory of medium shrub willow (*Salix* spp.) and birch (*B. glandulosa*), but trees were more apparent in 2011. White Spruce had less understory shrub but was characterized in both years by mature white spruce trees and high canopy cover (Appendix 4D). Black Spruce remained an open woodland of stunted trees < 10 m tall, despite a shift in tree species composition due to a decrease in tamarack trees (Appendix 4E).

## **Birds**

We conducted 89.0 (2010) and 90.3 hours (2011) of surveys compared to 83.4 hours spent by Spindler (1976). Despite a greater survey effort in 2010-11, the total number of birds detected was greater in 1975 ( $n = 1390$ ) than in 2010 ( $n = 754$ ) or 2011 ( $n = 835$ ). In 1975, 155 territories of 29 bird species were documented compared to 117.5 territories of 22 species in 2010-11. This represents a 24.1% decline in breeding species richness and 24.2% decline in overall abundance. Species richness, abundance, and diversity declined for all habitat plots except Low Shrub (Fig. 1.3A).

Thirty-three species of birds were documented breeding at Creamer's Refuge in either 1975 or 2011 (Fig. 1.4). Of these, 11 were found in 1975 but not in 2011, and four were found breeding in 2011 only. Abundance of 20 species decreased over time, but increased for 13 species. Of particular interest are Townsend's Warbler, American Tree Sparrow, Gray-cheeked Thrush, and Pine Grosbeak. Spindler (1976) recorded Townsend's Warbler as "fall visitants" only, but we



documented breeding on White Spruce in 2011 (both years). Conversely, American Tree Sparrow and Gray-cheeked Thrush made up 11.3% of the documented breeding bird territories and were found on three and five plots in 1975, respectively, but were not found at all in 2011. During repeated point count surveys across the 1100-ha Creamer's Refuge, we recorded these species only during migration (Tauzer, unpublished). Pine Grosbeak was also documented in low densities on two plots in 1975, but was not observed at Creamer's Refuge during the breeding seasons of 2011.

In both 1975 and 2011, Lincoln's Sparrow was the most abundant species documented at Creamer's Refuge (Fig. 1.4). In 1975, it was followed in abundance by Yellow Warbler, Wilson's Snipe, White-crowned Sparrow, American Tree Sparrow and Dark-eyed Junco; in 2011, it was followed by Dark-eyed Junco, Yellow-rumped Warbler, Swainson's Thrush, Hammond's Flycatcher and Orange-crowned Warbler.

#### *Change by habitat plot*

The breeding bird communities on the plots were distinct from each other (Fig. 1.3A), and the amount of change over time differed by habitat type. Avian community composition changed most in Birch (Bray-Curtis dissimilarity index (BC) = 0.734) and Tall Shrub (BC = 0.730) plots, and less on the forested plots (White Spruce, BC = 0.423; Black Spruce, BC = 0.403). The most notable change was on the Low Shrub plot, where species composition remained largely the same resulting in intermediate overall change (BC = 0.433), but the presence of additional forest-associated species caused an increase in species richness and diversity. Change in avian community between 1975 and 2011 was positively correlated to ecological change in habitat ( $r_M = 0.489$ ;  $P = 0.002$ ).

### *Change by avian-habitat guild*

Abundance of forest-associated birds increased over time while abundances of shrub and wetland birds declined (Fig. 1.4). In 1975, the majority of birds found at Creamer's Refuge were shrub-associated species but by 2011, coniferous forest-associated species dominated and deciduous species were much more common (Fig. 1.3B). Many of the common forest-associated species (e.g. Yellow-rumped Warbler, Dark-eyed Junco, Swainson's Thrush, American Robin) were now found breeding on all five habitat plots.

The greatest population increases observed were for forest-associated bird species (Dark-eyed Junco, Hammond's Flycatcher, Yellow-rumped Warbler), whereas large population decreases were documented for shrub-associated species (Yellow Warbler, American Tree Sparrow, Gray-cheeked Thrush, White-crowned Sparrow; Fig. 1.4). Many species of shrub- and wetland-associated birds were no longer breeding on the plots, while others exhibited substantial population declines. Orange-crowned Warbler was the only shrub-associated species, and Lincoln's Sparrow the only wetland bird species, that had higher abundances over time.

## **1.5 DISCUSSION**

Avian habitat on Creamer's Refuge changed significantly over 35 years; density, diversity and basal area of trees increased, as did forest-associated birds. Concomitantly, as shrub densities declined, the abundance and richness of shrub-associated birds declined. Although we documented notable declines in wetland-associated birds, we did not measure open water and thus cannot assess a relationship with habitat change. These findings provide support for rapid avian response to local environmental change and highlight the importance of habitat availability as a driver of population change.

### **Evidence of plant succession**

The habitat changes observed were generally as expected for succession in the northwestern boreal forest. In the absence of disturbance at Creamer's Refuge, the overall ecosystem shifted towards closed habitats with more tall shrubs and trees and reduced shrub density. Understory shrub density decreased on plots with high canopy cover, presumably due to light competition, but increased on plots with low cover, suggesting that light was not limiting there. Also as expected, the plots with mature forests (White Spruce, Black Spruce) exhibited less change than the early successional shrubby plots (Birch, Tall Shrub). Given that black spruce is well suited for the poorly drained permafrost-dominated soils found at Creamer's Refuge (Van Cleve et al., 1983) and that natural floodplain succession is towards sphagnum black spruce bog in interior Alaska (Viereck, 1970), it was not surprising that black spruce density and importance increased over time.

Black spruce forests are generally considered "climax" communities that remain relatively unchanged until some major disturbance, such as a fire or insect outbreak (Viereck, 1973; Chapin et al., 2008), but we documented an increase in black spruce and a considerable decrease in tamarack densities on the Black Spruce plot. This decline in tamarack trees could be explained by an outbreak of larch sawfly (*Pristiphora erichsonii*) that occurred in 1993-1999 and was associated with significant mortality on 105,200 ha of pure tamarack and mixed tamarack/black spruce stands of interior Alaska (Lamb and Winton, 2010). The frequency and severity of insect outbreaks such as this are expected to increase with climate warming (Berg et al., 2006).

In contrast to Black Spruce, which changed more than expected, the vegetation on the "young" Low Shrub plot changed less than what would be expected in a typical successional trajectory. This may be because growth and survival of trees

is limited by cold soil temperatures and wetness in the lowlands of interior Alaska (Van Cleve et al., 1996). As a result, habitats such as this tussock bog, with shallow active layers and poor drainage, tend to be maintained at a more stable condition than drier, better-drained habitats like Birch or Tall Shrub.

### **Evidence of avian succession**

At our study site, forest-dependent species such as Dark-eyed Junco, Swainson's Thrush, and Yellow-rumped Warbler have benefited from increased forest cover. However, the coupled loss of medium shrub habitats had a disproportionate negative impact on shrub-associated species such as American Tree Sparrow, White-crowned Sparrow, and Yellow Warbler. While we cannot say with certainty that habitat change is the ultimate cause of the changes observed for birds, we found evidence for this on two levels. First, the overall vegetation change observed was consistent with the change in birds by avian-habitat guild, and second, there was a positive correlation between change in habitat and birds when comparing distance matrices. The most likely explanation for the changes seen in bird community composition at Creamer's Refuge is that habitat preferred by forest birds has replaced habitat required by shrub-dependent species.

While this is the first such study in the northwestern boreal forest, several long-term studies in eastern North America have provided strong evidence of avian community succession over time, with increases in abundances of late-succession forest birds and an overall loss of "early successional" species at established sites (Askins and Philbrick, 1987; Holmes and Sherry, 2001; Brooks and Bonter, 2010). Population declines of early successional bird species [e.g. Eastern Towhee (Hagan, 1993), Least Flycatcher (Holmes and Sherry, 1988)] have been attributed to the natural increase in forest cover over a similar time period as our study. On a broader, near-continental scale, Valiela and Martinetto

(2007) found that abundances of North American forest bird species increased over time (1966–2004), while species of open and wetland habitats declined; they attributed these abundance changes to widespread regeneration and expansion of eastern North American forests in the past century.

### **Changes with climate warming**

Climate warming in the western boreal forests has been linked to large-scale landscape changes such as reduced growth of coniferous trees (Barber et al., 2000; Beck et al., 2011), increased wildfire extent (Kasischke et al., 2010), and a large scale shift towards deciduous trees (Barrett et al., 2011). Change in fire regime has implications for boreal birds because it could result in large expanses of early successional habitat becoming available for shrub-associated avian species. Early successional shrub habitats and deciduous forests have higher productivity than their coniferous counterparts (Kessel, 1998) so this type of change would likely result in increased species richness and abundances, which is the opposite of what we observed at Creamer's Refuge in the past 35 years. On the other hand, poorly-drained lowland habitats tend to be resistant to fire so that here, even with climate warming, we predict further spread of forest and a coupled decline in open shrub habitats. At Creamer's Refuge, where homogenous shrub habitats are already uncommon and fires are suppressed (due to proximity to the city of Fairbanks), there is potential that shrub habitat required by nearly 30% of the breeding bird species will decrease in coming years. This demonstrates the difficulty in making generalizations across habitats or broader ecosystems.

In Alaska, medium shrub habitats occur at both the cool wind-exposed upper elevations and at cold, wet lowland sites (Kessel, 1998). If recent warming trends continue as anticipated, these habitats could be doubly threatened by encroaching trees, first in their lowland habitats (places such as Creamer's

Refuge) and, second, at higher elevations as trees and shrubs move upslope (places such as Denali National Park, AK (Potter, 2004)). Tree encroachment or infilling has been observed in other northern habitats such as tundra-treeline (Stueve et al., 2011; Lloyd and Fastie, 2003) and wetlands of south-central Alaska (Berg et al., 2009).

The thawing of permafrost and drying of lakes due to climate change are also likely to impact boreal bird communities. Wetness was one of the most important habitat variables driving avian communities in western and northern Canadian boreal forest (Kirk et al., 1996), thus changes in surface hydrology has direct implications on boreal forest birds. In our study, both richness and abundance of wetland-associated birds declined, and we can infer from this that the amount of open water decreased on the plots. Unfortunately, our study was not sufficient to document change in moisture or open water on the 10-ha scale of plots. Remote sensing analysis of Creamer's Refuge did not demonstrate any obvious changes in water bodies during the same time period (Tauzer, unpublished).

Another well-documented prediction of climate warming is that species range shifts will occur as plants and animals move into newly suitable areas. This includes both the slow range expansion of trees and shrubs (Danby and Hik, 2007b; Matthews et al., 2011) and the rapid spread of invasive plant species (Villano and Mulder, 2008), as well as shifts in breeding distribution of birds (Huntley et al., 2006). At Creamer's Refuge, we observed the recent expansion of chokecherry (*Prunus padus*), an invasive ornamental tree, into the Birch and Tall Shrub plots. Chokecherry, also known as European bird cherry, is an important food resource for birds, especially in early spring and late winter when little food is available. It appears to be locally spreading as a result of seed dispersal by birds. Little is known about the ecosystem implications, but this tall,

quick-growing shrub has the potential to reduce light, soil moisture, and nutrient availability for other understory species.

We did not observe any bird species that had never been seen at Creamer's Refuge and there is no way to tell if population ranges have shifted over time. It is possible that altered species interactions are occurring as a result of new avian assemblages (see example by Stralberg et al., 2009), but too few data are currently available to evaluate this. On the boreal breeding grounds in Alaska, where many birds are at the northern extent of their range, warming temperatures might benefit some populations because more birds will be able to breed successfully and survive previously adverse conditions. This might explain why more species demonstrated population increases than decreases (see below).

### **Avian population implications**

To our knowledge, this is the first study in the northwestern boreal forest to examine avian successional change temporally; therefore, we have little with which to compare our population trend data. Even the North America Breeding Bird Survey (BBS), which is often used to estimate trends in bird populations, has limited inference in this region because there are few, widely-dispersed survey locations. Much of Yukon Territory and Alaska is remote and difficult to access and as a result, sample sizes are small and surveys have only been conducted consistently since 1980. Given this, and the limited spatial scope of our project, we predicted that our observations would not correspond well with regional trends derived from BBS. However, when we compared our results with BBS data for Alaska, Yukon Territory, and Bird Conservation Region 4 (Northwest Interior Forest), the majority of species (66.7%) did match regional trends. Of the species exhibiting positive or negative population trends ( $P < 0.15$ ), 21 were found breeding at Creamer's Refuge and 15 had  $\geq 1$  territory documented on our

plots (Table 1.3). Four species demonstrated BBS population declines, and 11 showed increases. Of the species documented at Creamer's Refuge, BBS population trends were upheld for all species demonstrating population declines: Lesser Yellowlegs, Blackpoll Warbler, White-crowned Sparrow and Rusty Blackbird, and for six of 11 species showing population increases: Hammond's Flycatcher, Ruby-crowned Kinglet, American Robin, Yellow-rumped Warbler, Orange-crowned Warbler, and Lincoln's Sparrow. We had conflicting results (negative where BBS indicated positive population trends) for five species: Green-winged Teal, Wilson's Snipe, Gray Jay, Northern Waterthrush, and Fox Sparrow. These species were known to breed elsewhere on the refuge, however, which highlights the importance of assessing habitat on multiple scales, especially since boreal breeding songbirds are known to use home ranges far larger than their territories and to respond to habitat at a landscape-scale (Whitaker et al., 2008; Whitaker and Warkentin, 2010).

Additional effort was made in both 1975 and 2011 to assess bird occupancy at the refuge level (Spindler, 1976; Tauzer, unpublished) and we feel that birds observed on the plots were typical of the refuge overall, but we do not know how representative this site may be of northwestern boreal forest in general. Despite this limitation, the consensus between our results and BBS data strengthens confidence in BBS trend estimates and demonstrates that small plot-based projects may be capable of detecting regional trends (Kirk et al., 1997; McNulty et al., 2008).

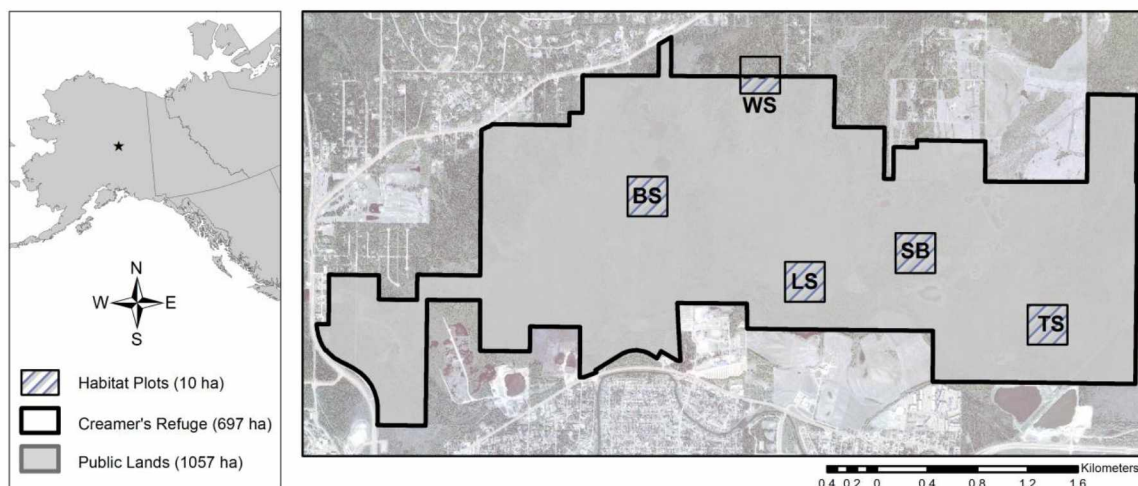
## **Conclusions**

Declines in North American bird populations are likely complex and a result of several interacting factors. Climate change and anthropogenic disturbances on northern breeding grounds will probably affect boreal birds; however, our results show that even without physical disturbances, natural succession is also an

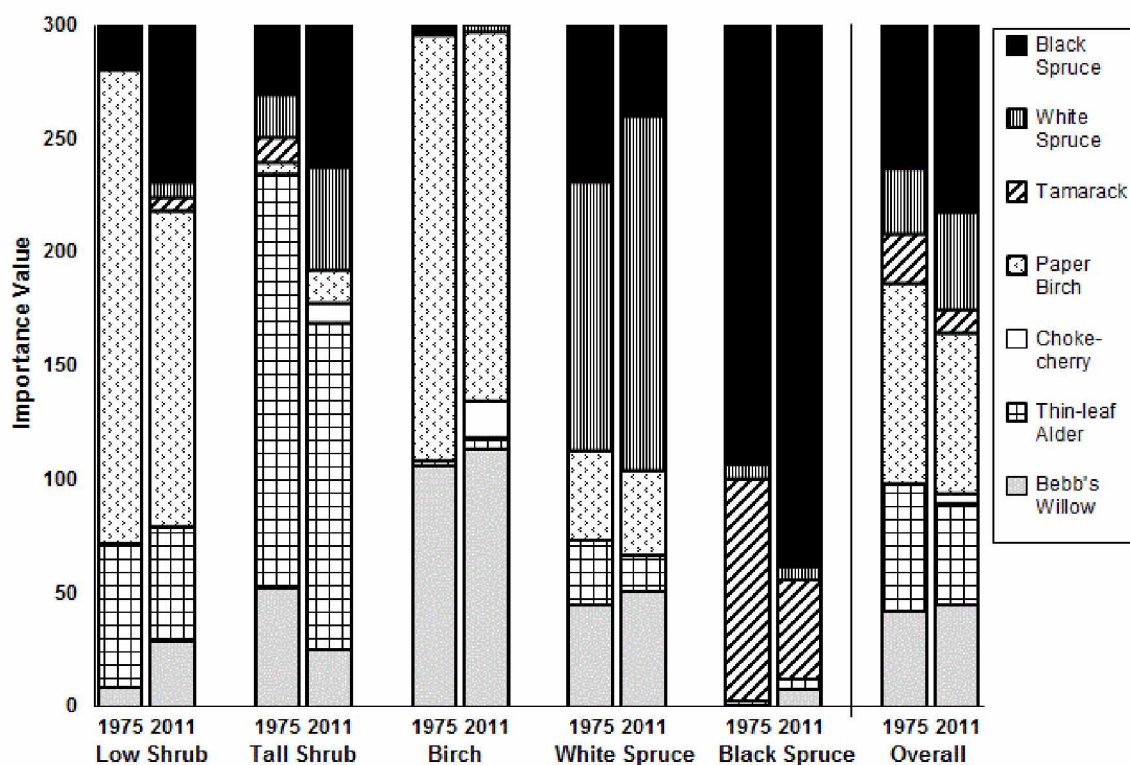


important driver of change in avian communities. We provided evidence that vegetation and avian succession are strongly linked. Given that climate warming is a pressing issue, and that rapid habitat changes are predicted in the future, it is important to consider the impact of local successional vegetation changes when assessing long-term population trends of birds. Increased knowledge of specific habitat requirements and associations would give us a better understanding of avian response to environmental change and allow us to better predict range and population changes in the boreal forest.

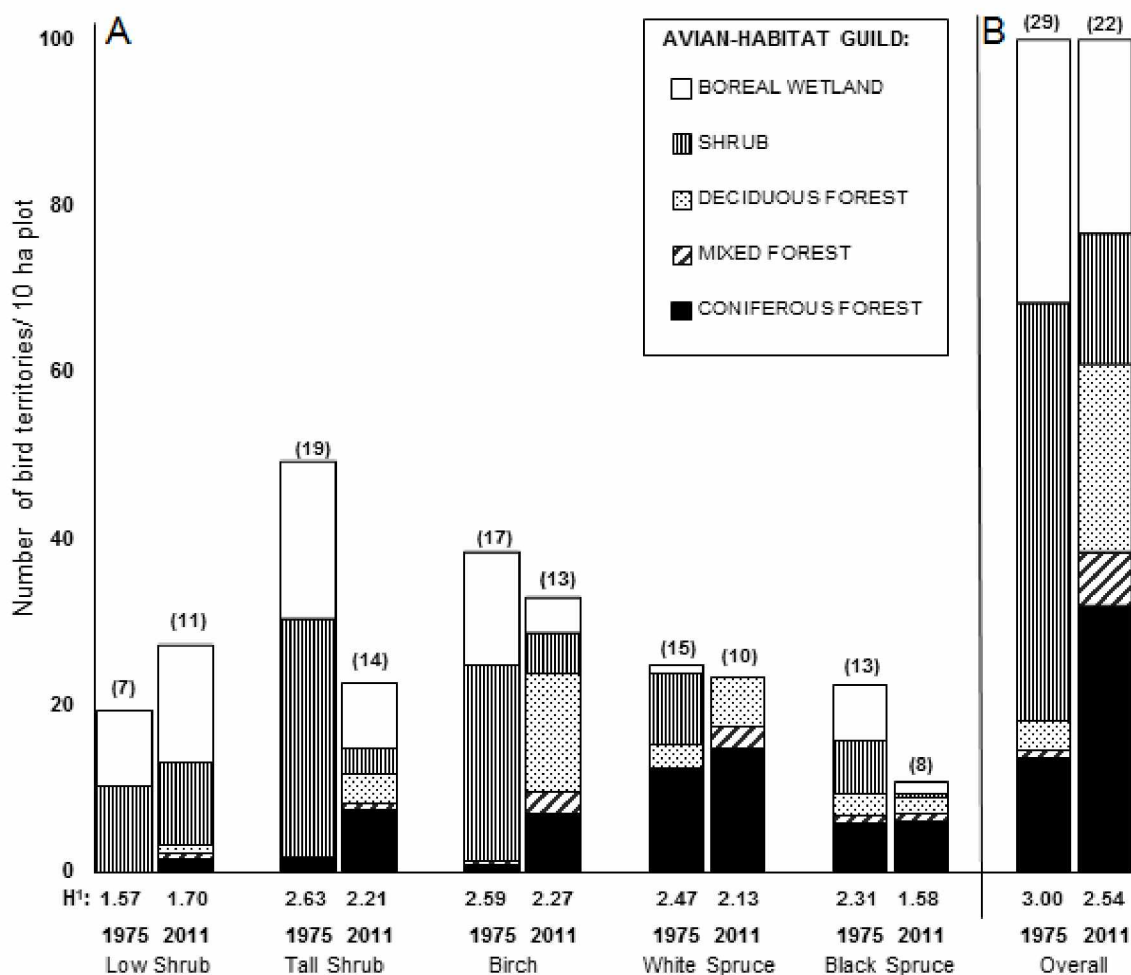
## 1.6 FIGURES



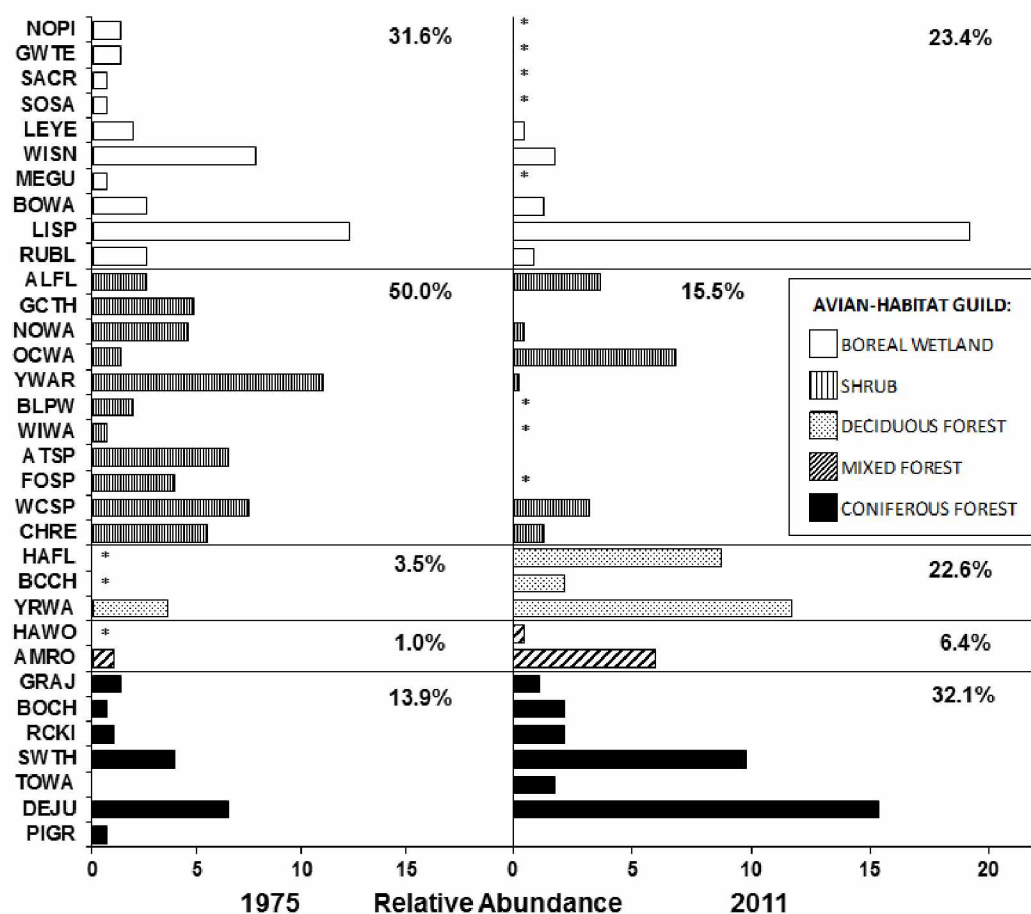
**Fig. 1.1** Study site and 10-ha habitat plots at Creamer's Refuge, Fairbanks, Alaska. Five plots were selected in 1975 to represent the dominant habitat types in this boreal ecosystem and were revisited in 2011 to assess plant and avian community change: 'BS' is open Black Spruce woodland, 'WS' is White Spruce forest, 'LS' is open Low Shrub thicket, 'SB' is young (Seral) Birch forest and 'TS' is Tall willow/ alder Shrub. In 2011, only 4.29 ha of the WS plot were surveyed for birds and vegetation because of land ownership issues.



**Fig. 1.2** Importance values for tree and tall shrub species documented on 10-ha habitat plots at Creamer's Refuge, Fairbanks, Alaska in 1975 and 2011. Trees and tall shrubs were measured using a point-centered quartered method and counted if the diameter at breast height was > 2.5 cm. Importance value (IV) is a measure of relative abundance for each species measured and is calculated by the equation:  $IV = \text{relative dominance} + \text{relative density} + \text{relative frequency}$ .



**Fig. 1.3** Abundance of avian-habitat guild-associated birds found breeding at Creamer's Refuge, Fairbanks, Alaska in 1975 and 2011 on (A) 10-ha habitat plots, and (B) overall. In A, the number of territories for each habitat guild is the total number of territories for all guild-associated species; in B, abundances in each habitat guild are proportional, based on the total number of territories for each year ( $n_{1975} = 155$ ;  $n_{2011} = 117.5$ ). The numbers shown above the bars for each plot is breeding species richness; species diversity ( $H^1$ ) is shown below the bars.



**Fig. 1.4** Relative abundance of avian species documented breeding at Creamer's Refuge, Fairbanks, Alaska in 1975 and in 2011, grouped by avian-habitat guild. For each species, relative abundance is calculated from the total number of territories documented across the Refuge (all plots summed). The numbers shown are relative abundance by habitat guild for each year. \* denotes species that were documented at Creamer's Refuge but did not hold breeding territories on the plots sampled. Species with < 0.5 territories in both years were excluded.

**Table 1.1** Vegetation data for five 10-ha habitat plots surveyed in 1975 and 2011 at Creamer's Refuge, Fairbanks, Alaska. Trees and tall shrubs were measured using a point-centered quartered method and counted if the diameter at breast height was > 2.5 cm. Dwarf shrub refers to woody plants < 0.3 m tall and medium shrub refers to those >1 m tall. Ecological distance is presented as a measure of change between years for each plot and was derived from a distance matrix of NMDS ordination using all habitat variables. Where possible, we presented mean values  $\pm$  SD. \* denotes when a species was present on plot but not documented by vegetation sampling.

	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
	1975	2011	1975	2011	1975	2011	1975	2011†	1975	2011	1975	2011
Total density of trees and tall shrubs (# stems/ ha):	28	156	663	1569	1623	893	2016	1857	815	1340	1029	1163
Total basal area of trees and tall shrubs (m <sup>2</sup> / ha):	0.07	0.31	1.19	4.39	0.62	13.93	11.36	35.26	2.18	3.92	3.09	11.56
Mean basal area of trees and tall shrubs (cm <sup>2</sup> ):	10.0	19.6 $\pm$ 22.7	14.0	28.3 $\pm$ 32.9	24.0	155.9 $\pm$ 184.0	66.0	189.8 $\pm$ 316.1	33.0	29.2 $\pm$ 35.7	29.4	84.6 $\pm$ 145.7
<b>TREES:</b>												
Density of tree species (# stems/ ha):	21	133	107	550	1159	407	1552	1481	815	1340	731	785
<i>Black spruce</i>	0.3	38.1	47.1	323.4	26.0	-	717.7	353.8	565.6	1257.6	271.3	394.6
<i>White spruce</i>	-	3.9	47.1	210.2	13.0	*	794.3	972.9	24.5	34.2	175.8	244.2
<i>Tamarack</i>	-	3.9	13.3	-	-	-	-	-	224.9	47.8	47.6	10.3
<i>Paper birch</i>	21.1	87.0	*	16.2	1119.9	407.1	40.3	154.8	-	-	236.3	133.0
<i>Aspen</i>	*	-	-	-	*	-	-	-	-	-	*	-
<i>Chokecherry</i>	-	-	*	16.2	-	*	-	-	-	-	*	3.2
Density of tree-like shrub species (# stems/ ha):	7	23	555	1003	464	486	468	376	*	*	299	378
<i>Thin-leaf alder</i>	6.6	22.5	487.3	962.1	26.0	28.3	183.5	110.6	*	*	140.7	224.7
<i>Bebb's willow</i>	*	0.8	67.6	40.4	438.2	458.0	284.3	265.3	-	*	158.0	152.9
Mean height of tallest trees (m):	6.1	6.2 $\pm$ 1.8	9.8	13.3 $\pm$ 2.6	8.7	17.9 $\pm$ 1.5	14.0	21.4 $\pm$ 3.2	10.4	9.5 $\pm$ 1.5	9.8	12.6 $\pm$ 5.4
Mean DBH of oldest looking trees (cm):	4.9 $\pm$ 2.9	9.6 $\pm$ 3.0 §	13.4 $\pm$ 4.1	22.2 $\pm$ 5.3 §	10.7 $\pm$ 3.8	29.4 $\pm$ 4.2 §	23.6 $\pm$ 9.6 *	35.0 $\pm$ 9.5	12.9 $\pm$ 3.0	13.1 $\pm$ 4.1	11.7 $\pm$ 6.7	20.6 $\pm$ 10.4 §
Mean age of oldest looking trees (years):	28.2 $\pm$ 18.4	40.5 $\pm$ 11.7 *	44.6 $\pm$ 21.4	61.6 $\pm$ 18.1 ‡	20.7 $\pm$ 8.9	53.9 $\pm$ 10.7 §	49.4 $\pm$ 9.1	72.1 $\pm$ 10.1 §	108.2 $\pm$ 22.7	135.5 $\pm$ 24.4 §	49.7 $\pm$ 37.0	72.7 $\pm$ 38.5 §
Tree species diversity (H')	0.89	0.83	0.52	1.09	0.37	0.21	1.06	1.07	0.76	0.78	1.16	1.32
<b>SHRUBS:</b>												
Density of medium shrubs (# stems/ha):	37,544	55,421	47,424	30,740	55,575	8,323	37,544	11,369	7,904	10,102	37,198	23,191
% cover dwarf shrubs:	22	24	37	12	29	6	21	22	23	37	26	20
Shrub species diversity (H')	2.38	2.37	2.59	2.59	2.04	2.16	2.52	2.31	2.45	2.52	2.84	2.80

**Table 1.1** continued.

	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
<b>OTHER:</b>												
Canopy coverage (% of sky obscured):	0.4	11	26	48	16	78	34	63	8	13	17	43
Plant species diversity (all species) (H')	2.86	2.85	3.44	3.50	3.02	3.05	3.55	3.41	3.23	3.19	3.78	3.74
% cover forbs:	9	10	11	22	8	13	11	20	10	30	10	19
% cover grass:	48	64	31	50	46	44	13	30	8	14	29	40
% tussock:	42	69	5	1	13	0	0	0	1	6	12	15
% cover moss:	10	21	9	20	8	7	33	43	42	62	20	31
% cover lichen:	4	2	3	1	2	1	6	5	9	20	5	6
% cover leaf litter:	7	4	8	19	6	58	16	30	8	1	9	22
Mean active layer depth (cm):	46.6 ± 6.8	44.9 ± 6.9	48.8 ± 8.0	48.2 ± 9.6	61.8 ± 14.8	62.0 ± 14.6	91.1 ± 21.2	81.6 ± 23.1	48.1 ± 14.9	50.2 ± 19.4	55.1 ± 18.3	54.0 ± 18.2
<b>Ecological (Euclidian) Distance:</b>	<b>2.215</b>		<b>5.685</b>		<b>7.331</b>		<b>4.953</b>		<b>5.411</b>		<b>3.198</b>	

† Only 4.29 ha were sampled on the White Spruce plot in 2011 because of land ownership issues.

\* Significant difference between 1975 and 2011 ( $P < 0.05$ ); ‡ Significant difference ( $P < 0.01$ ); § Significant difference ( $P < 0.001$ )

**Table 1.2** Avian-habitat guilds (Sharbaugh, 2007) for breeding boreal birds at Creamer's Refuge, Fairbanks, Alaska (Appendix 1). Species shown in bold had territories on at least one of five 10-ha habitat plots surveyed in 1975, 2010, and/or 2011. Population trends ( $P < 0.15$ ) derived from Breeding Bird Survey data for Yukon Territory, Alaska, or Bird Conservation Region 4 (Northwestern Interior Forest) are indicated with a (+) for positive or (-) for negative (Sauer et al., 2011).

CONIFEROUS FOREST	MIXED FOREST	DECIDUOUS FOREST	SHRUB	BOREAL WETLAND
<b>Gray Jay (+)</b>	<b>Hairy Woodpecker</b>	<b>Hammond's Flycatcher (+)</b>	<b>Alder Flycatcher</b>	<b>Northern Pintail</b>
<b>Boreal Chickadee</b>	<b>American Robin (+)</b>	<b>Black-capped Chickadee</b>	<b>Gray-cheeked Thrush</b>	<b>Green-winged Teal (+)</b>
<b>Ruby-crowned Kinglet (+)</b>	Sharp-shinned Hawk (+)	<b>Yellow-rumped Warbler (+)</b>	<b>Northern Waterthrush (+)</b>	<b>Sandhill Crane</b>
<b>Swainson's Thrush</b>	Northern Goshawk	Bufflehead	<b>Orange-crowned Warbler (+)</b>	<b>Solitary Sandpiper</b>
<b>Townsend's Warbler</b>	Great Horned Owl	American Kestrel	<b>Yellow Warbler</b>	<b>Lesser Yellowlegs (-)</b>
<b>Dark-eyed Junco</b>	Downy Woodpecker	Northern Flicker	<b>Blackpoll Warbler (-)</b>	<b>Wilson's Snipe (+)</b>
<b>Pine Grosbeak</b>			<b>Wilson's Warbler</b>	<b>Mew Gull</b>
Common Goldeneye			<b>American Tree Sparrow</b>	<b>Bohemian Waxwing</b>
Red-tailed Hawk			<b>Fox Sparrow (+)</b>	<b>Lincoln's Sparrow (+)</b>
Varied Thrush			<b>White-crowned Sparrow (-)</b>	<b>Rusty Blackbird (-)</b>
White-winged Crossbill (+)			<b>Common Redpoll</b>	American Wigeon (+)
American Three-toed Woodpecker (-)			Northern Shrike	Mallard (+)
			Black-billed Magpie (+)	Northern Shoveler
			Savannah Sparrow (-)	Blue-winged Teal
				Horned Grebe
				Bonaparte's Gull
				Olive-sided Flycatcher (-)



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## **CHAPTER 2: MAPPING HABITAT CHANGE USING REMOTE SENSING IN A SMALL, RELATIVELY UNDISTURBED BOREAL FOREST IN INTERIOR ALASKA**

### **2.1 ABSTRACT**

Growing evidence shows that ecosystems are being affected by warming temperatures, but our understanding of the magnitude and direction of vegetation change at northern latitudes is limited, primarily due to a lack of baseline data with which to make comparisons. With the goal of establishing baseline habitat maps and a comprehensive understanding of recent change to supplement long-term field-based monitoring, I examined change in vegetation and forest health at a local- and site-specific scale in an interior Alaska boreal forest. I used late-summer cloud-free Landsat data over a 25-year period (1984–2009) to create six habitat classification maps (1985, 1992, 1999, 2006, 2008, 2009) with an overall accuracy of 70.6%, and documented a successional shift towards denser, more coniferous-dominated habitats. Declines in **Normalized Difference Vegetation Index (NDVI)** and **Normalized Difference Moisture Index (NDMI)** were recorded, with coniferous forests exhibiting the most significant declines in both. **NDMI** proved useful in delineating hard-to-differentiate shrub classes and has potential in documenting direction of vegetation shift. This study demonstrates the usefulness of freely available medium-resolution satellite images for studying spatial and temporal variability of foliage moisture and vegetation, two important factors influencing plant and wildlife distributions.

### **2.2 INTRODUCTION**

Areas of the boreal forest have warmed at nearly twice the global average (ACIA, 2005; Chapin, 2006; IPCC, 2007), resulting in lake drying (Riordan et al., 2006) and enhanced seasonal thawing of permafrost (Jorgenson and Osterkamp, 2005) in interior Alaska. Over the last century, there was a 45% increase in

annual growing season and an 11% decrease in precipitation in Fairbanks, Alaska (Wendler and Shulski, 2009). Widespread drought throughout interior boreal forests has led to a decline in the health of coniferous forests and an increased risk for disease and insect outbreaks (Barber et al., 2000; Beck et al., 2011). Predictions for the boreal forest region often involve dramatic changes in vegetation that are both broad in scale (Chapin et al., 2004; Calef et al., 2005) and non-linear (Soja et al., 2007; Mann et al., 2012). In part, this is because an increase in fire frequency (Kasischke et al., 2010) has caused large areas of spruce forest to be replaced by broadleaf deciduous trees (Johnstone et al., 2010; Barrett et al., 2011).

Vegetation changes have important implications for human society and wildlife populations because different vegetation types offer distinct ecosystem services (Chapin et al., 2010; Kofinas et al., 2010), yet many questions remain regarding rates of change. In the arctic tundra, where the majority of climate change research has occurred, there has been a recent shift to taller and denser vegetation (Suarez et al., 1999; Lloyd and Fastie, 2003; Tape et al., 2006). As temperatures increase, accelerated successional changes may cause similar changes in areas of the boreal forest, although few studies have implicitly examined this. Most boreal forest studies have focused primarily on post-fire recovery (Johnstone and Chapin, 2006; Shenoy et al., 2011) or floodplain recolonization (Viereck, 1970; Hollingsworth et al., 2010), so relatively little is known about secondary succession in undisturbed areas. In addition, boreal forest succession is complex and follows multiple pathways. More than anything else, however, our understanding of vegetation change is limited by a lack of historical data.

In many remote places, characterized by low human population density, difficult climatic conditions, and lack of roads, remotely-sensed images are the only

historical data available. Indeed, much of the evidence for recent widespread changes throughout the Arctic and boreal forest ecosystems comes from remote sensing technology (Stow et al., 2004; Laidler et al., 2008), rather than field observations. Remote sensing is a non-contact method of data collection, which involves the examination of aerial and satellite images and is particularly attractive to managers because it permits assessment of large areas and of places that are otherwise inaccessible due to time and cost restrictions. In the northern boreal forest, biophysical remote sensing is still in the exploratory stages, with attempts being made to establish methodological protocols and baseline vegetation inventories (Ustin and Xiao, 2001; Roach et al., 2012). In addition, land cover maps such as the National Landcover Database 2001 (<http://www.mrlc.gov/>) are widely used for landscape-level wildlife habitat assessments although, until recently, no indication of the appropriate scale for use or estimate on accuracy were available (Wickham et al., 2010; Selkowitz and Stehman, 2011).

Remote sensing is an effective method of monitoring changes in wildlife habitat (Franklin, 2010; McDermid et al., 2010) and has proved to be a very efficient tool for documenting shifts in land cover and land use (Lu et al., 2004; Rogan, 2004). Although most often used to detect rapid anthropogenic changes (Lepers et al., 2005) or extent of natural disturbances such as hurricanes (Wang and Xu, 2009) or fire (Fraser et al., 2000; Wang et al., 2008), multi-temporal satellite images have also been effectively used to quantify more gradual processes such as forest succession (Cohen et al., 2002; Schroeder et al., 2006) or climate-induced vegetation changes (Danby and Hik, 2007; Fraser et al., 2011; McManus et al., 2012). Many remote sensing studies capitalize on the fact that sensor technology collects data simultaneously in wavelengths beyond human visual range (in the near, short wave and thermal infrared regions) and use simple ratio indices, such

as a vegetation index as a measure of ecosystem function (Townshend and Justice, 1995; Prakash and Gupta, 1998; Bunn and Goetz, 2006).

The Normalized Difference Vegetation Index (NDVI) is the most commonly employed index used in ecosystem studies (Pettorelli et al., 2005). Healthy green vegetation reflects strongly in near-infrared (NIR) and absorbs energy in the red region of the spectrum, and NDVI uses this contrast in spectral response to assess the spectral contribution of green vegetation (Jensen, 2000). It is particularly useful for delineating primary vegetation types, such as grassland, shrubland, and coniferous or deciduous forest (Tucker, 1979), but is also effective at detecting dramatic change in chlorophyll contents of a plant's leaves ('greenness'), such as what happens during rapid tree death or forest regeneration. NDVI is less successful, however, in detecting subtle changes in individual trees affected by stress, or in detecting successional changes. Because a single NDVI value is related to both the abundance of vegetation and its greenness, and different species of plants have different reflectance values, small changes in plant composition can lead to changes in NDVI values that do not necessarily reflect a decline in forest health.

For detecting drought (Lin et al., 2011) or disturbance events where leaf water content changes but pigments do not drop rapidly, the Normalized Difference Moisture Index, or NDMI, has proved useful (Gao, 1996; Jin and Sader, 2005). NDMI, also referred to as Normalized Difference Water Index (NDWI), is a mathematical combination of short-wave infrared (SWIR) and NIR bands and is less influenced by atmosphere than NDVI. Data collected in the SWIR region is most sensitive to foliage moisture variation (Jensen, 2000) so it is good at identifying forest structural attributes and canopy coverage (Collins and Woodcock, 1996; Wilson and Sader, 2002; Jin and Sader, 2005). Change in foliage moisture can indicate phenological changes (Delbart et al., 2005) and tree

stress caused by drought, insects or diseases (Wang et al., 2007; Goodwin et al., 2008).

Given that remote sensing is often used to monitor temporal change, and that there are few field-validated examples of successional change in the rapidly changing boreal forest ecosystem, I was interested in examining recent habitat change at Creamer's Refuge, in Fairbanks, Alaska (Fig. 2.1). Typically, remote sensing analysis is used for more expansive areas than this small refuge, but this area was selected for several reasons. First, there was local interest in developing a long-term ecosystem monitoring station at this location. Second, this area is unique in that there are > 20 years of avian demographic data available. Finally, it is relatively undisturbed and easily accessible, which meant that I could field-validate the remote sensing- based habitat maps.

Given this impetus, my broad goals were twofold: 1) to establish baseline land cover maps for future monitoring of avian habitat, and 2) to assess whether remote sensing could be used at this temporal scale to document environmental change in this boreal forest. My specific objectives were to: use Landsat images to evaluate successional change in habitat over a 25-year period (1984-2009), provide an accuracy assessment for the most current (2009) classified habitat map, and explore forest health response of different habitat classes during the same time period, by examining trends in NDVI and NDMI.

My primary interest was in avian habitats. Worldwide, birds have shown a strong response to climate change (Møller et al., 2010). In the Alaska boreal forest, it is unclear whether changes observed in bird populations are related to changes in climate, habitat, or both (see Chapter 1). Because of my interest in the avian community, I selected detailed vegetation classes that were distinguishable in the field and associated with boreal birds. I chose to evaluate change in NDMI, even

though it is not commonly considered in remote sensing studies, because wetness is an important avian habitat component (Kirk et al., 1996; Kessel, 1998), and pronounced changes in wetlands have been observed in nearby areas of the boreal forest (Jorgenson et al., 2001).

Even without considering effects of climate change, succession in the boreal forest is influenced by numerous abiotic factors (e.g. aspect, hydrology, permafrost presence and disturbance history; Chapin, 2006; Kurkowski et al., 2008; Taylor and Chen, 2011); thus, I had no preconceived predictions about how this area of lowland boreal forest had changed since the 1980s. However, I anticipated a negative trend in NDVI values because several recent studies in the interior boreal forests of Alaska have documented negative, or “browning”, NDVI trends over time (Goetz et al., 2005; Verbyla, 2008; Beck et al., 2011; Baird et al., 2012). I had no expectations for NDMI because, to my knowledge, this is the first Alaska study to report trends for this index.

My research examines the types of change that are illuminated using freely available medium-resolution remote sensing data and investigates the temporal scale necessary to document environmental change in the interior Alaskan boreal forest ecosystem. By identifying habitat types that are most vulnerable to short-term change and quantifying the accuracy of remote sensing classification results, it may help managers (or anyone interested in setting up a long-term ecosystem monitoring program) to determine effective monitoring protocols. Although the spatial extent is small, similar habitats are found throughout the vast boreal forest ecosystem; thus, these data have potential to be applicable on a much larger scale.

In addition, the habitat classes that I selected are more detailed than normally attempted in land cover analysis because of the avian focus of this project.

Knowledge gained is applicable to future research because it will indicate if remote sensing can be used as a substitute for ground-based measurements of avian habitat. It may facilitate the use of remote sensing to predict ecological changes in both plant and avian communities over time.

## **2.3 METHODS**

### **Study area**

The study area is a 1057-ha tract of public land jointly owned by Alaska Department of Fish and Game, University of Alaska, and Fairbanks North Star Borough, hereafter referred to as Creamer's Refuge (Fig. 2.1). Creamer's Refuge is located just north of the Fairbanks, Alaska city boundary (64° 49'N, 147° 52'W). The continental climate of interior Alaska is arid, cold, and characterized by extreme seasonal variations in temperatures. Annually, average annual precipitation in Fairbanks is 28.0 cm and average daily temperature is -3.0°C, with January being the coldest month (mean temperature = -23.3°C) and July being the warmest (mean temperature = 16.4°C). Maximum annual snow depths, achieved by February, average 51.0 cm (ARCR, 2012).

The low-lying abandoned floodplains at Creamer's Refuge (altitude: 135-158 m) are a complex mosaic of wetland habitats. Typical of the northwestern portion of the boreal forest ecosystem, the area is characterized by dense shrub thickets and sparsely forested wetlands, interspersed with patches of black spruce muskeg and taller birch and white spruce forest. Small cave-in lakes and thaw ponds are common, there are areas of extensive thermokarst and polygonization, and four creeks flow slowly through the area resulting in patches of permanently-frozen, undifferentiated alluvial soil. The permafrost table ranges from < 0.25 m under black spruce muskeg to ~12 m under the fields (Spindler, 1976). Although easily accessible to the public, the majority of Creamer's Refuge is largely undeveloped except for winter dog-mushing and ski trails. A 1950s fire was

detected during a 1975 study (Spindler, 1976) but not apparent in 2010–11. Six small lakes (total area < 20 ha) were constructed to enhance waterfowl habitat at Creamer's Refuge between 1984 and 1987, and 2 small fires (total area ~ 6 ha) were set in 1996 and 1997 to create wildlife habitat (Creamer's Resource Inventory).

### **Remote sensing assessment of vegetation change**

I examined change at Creamer's Refuge in 2 ways: 1) by generating a series of thematic land cover maps and then evaluating change in percent cover of habitat classes over time, and 2) by assessing the change in index values (NDVI and NDMI) between 1985 and 2009. I used ENVI™ 4.5 for image processing and habitat mapping (except where noted) and ArcGIS™ 10 for final map generation.

Using remote sensing techniques for habitat classification and change detection involved: (a) searching and downloading available cloud-free summer time imagery of the study area; (b) preprocessing data to minimize external influences on the images; (c) masking water (and cloud, if necessary); (d) digital processing and enhancement; (e) subsetting the region of interest; and (f) performing supervised classification using field based knowledge (Fig. 2.2).

#### *Data search and download*

Landsat data was downloaded from the USGS GloVis site (<http://glovis.usgs.gov>); all images were preprocessed by the data provider to the T1 level and had a spatial resolution of 30m x 30m (900 m<sup>2</sup>). Because I was interested in changes in vegetation, I examined all imagery collected during the growing season for most plants (between 15 May and 30 September). From those available, I excluded scenes that had > 30% cloud cover, and chose seven images of similar high quality collected within six days of each other (Table 2.1). I



chose near-anniversary dates to minimize external effects caused by solar angle, seasonal differences in moisture, and phenological changes in vegetation.

### *Preprocessing*

All images used were from Landsat satellites, processed by the same facility, and came in the same projection (WGS-84). For each Landsat band and each year, I converted pixel values to at-surface spectral reflectance using the corrections described by Chander et al. (2009) and the automated process in Miramon™ 7. This process uses a dark-object subtraction and DEM model to do atmospheric and terrain correction prior to spectral reflectance calculation (Pons and Solé-Sugrañes, 1994).

### *Masking water and cloud*

The spectral signatures for dense coniferous forest were easily confused with water in several band combinations; however, in the NIR band, these two classes were distinguishable, with water showing very low values and coniferous forest showing a range of higher values. For each processed image, I masked out water pixels before further classification by visually determining a threshold in the NIR band that best delineated the water bodies.

All images were cloud-free except the 1984 image, which had a small amount of cloud cover at the periphery of the image. This patch of clouds showed much higher reflectance in the SWIR region than other areas on the image, so I masked out clouds out by visually thresholding and delineating high digital values on TM band 7 (the second SWIR band of Landsat). Although I processed and analyzed the 1984 image, I do not report final classification results because 14.2% was classified as cloud cover.

### *Digital image enhancement and processing*

Using the computed spectral reflectances for each year, I generated NDVI and NDMI ratio images, and performed principal component analyses and tasseled cap transformations (Crist and Ciccone, 1984). I ran five texture filters (data range, mean, entropy, variance, skewness) on the NDVI and NDMI images, using a 3x3 filter kernel. I used the following equations for the indices:

$$1. \text{ NDVI (Normalized Difference Vegetation Index) } = \frac{\text{NIR} - \text{R}}{\text{NIR} + \text{R}}$$

$$2. \text{ NDMI (Normalized Difference Moisture Index) } = \frac{\text{NIR} - \text{SWIR}}{\text{NIR} + \text{SWIR}}$$

Both NDVI and NDMI range from -1.0 to +1.0, where negative values correspond to dry, bare surfaces and positive values correspond to wet, vegetated surfaces. Typically, NDVI values for forests exceed 0.4 during the growing season, while NDMI values are slightly lower but vary in a wider range. Texture measures are commonly applied for extraction of forest characteristics at a variety of scales (Ge et al., 2006), and texture of NDVI has been used to draw out structural differences in vegetation (Wood et al., 2012). I was interested in exploring its utility here because it has also been used as an indicator of bird (Tuttle et al., 2006; St-Louis et al., 2009) and plant species richness (Gould, 2000).

I stacked all resulting layers ( $n = 30$ ) and, to identify layers that represented noise or redundancy, examined the spectral profiles for 10 pre-selected points representing the habitat classes of interest in ERDAS IMAGINE® 2011 (Fig. 2.3). If the pixel values were indistinguishable for all 10 points, I considered a layer of no use and ignored it; otherwise, I included it in the final layer stack. The final layer stack, used for classification, had 20 bands: Landsat TM bands 1-5 and 7; PC bands 1-4; Tasseled Cap bands 2-3; NDVI; NDMI; and three texture filters (data range, mean and skewness) for both NDVI and NDMI.

### *Subsetting*

I examined an area of interest the equivalent of 1096.5 ha, or 1283 pixels. Because the purpose of this project was to assess “natural” change, I excluded property adjacent to Creamer’s Refuge with obvious human influence (e.g. buildings, roads, and managed fields).

### *Generating thematic maps*

To quantify change in vegetation over time, I used the Mahalanobis distance algorithm to perform supervised classification on the 20-layer stack for each year image (Swain and Davis, 1978). As mentioned previously, I chose to delineate habitat classes (Fig. 2.4) that were compatible with ongoing avian research at Creamer’s Refuge. Bird occurrence is largely driven by vegetation height, heterogeneity and structural complexity (Kessel, 1979), so these classes emphasize structural characteristics of the landscape, rather than individual plant species.

Black Spruce is open habitat characterized by a thick sphagnum moss ground cover and widely spaced, stunted black spruce (*Picea mariana*) and tamarack (*Larix laricina*) trees. Coniferous Forest is widely spaced mature white spruce (*P. glauca*) trees, with high canopy cover (> 30%) and an open shrub understory. Mixed Forest has both deciduous and coniferous trees and tends to have high canopy cover and a variable shrub understory. Deciduous Forest is dominated by birch (*Betula neoalaskana*) trees with a high canopy cover and dry open understory. Scattered Woodland is characterized by black and/or white spruce trees of small stature and intermediate density; medium-tall shrubs often grow in the understory and canopy coverage is < 30% although the forest is denser than for Black Spruce. Tall Shrub often has spruce and birch trees at low densities but is dominated by tall alder (*Alnus incana* ssp. *tenuifolia*) or willow (*Salix bebbiana*) shrubs which can reach heights of  $\leq 8$  m. Low Shrub is a largely treeless and

often tussock dominated habitat, composed of dwarf willow (*Salix* spp.) or birch (*Betula glandulosa* or *B. nana*). Field is open grassland, most often the result of historic fire or drying ponds or lakes.

For the forest habitats (Birch, Coniferous Forest, Mixed Forest), I visually selected training areas using a RGB band composite of NDMI–Band4–Band3 because this three-band composite drew out forest characteristics (Fig. 2.5). Similarly, I used Band5–Tasseled Cap Greenness–NDVI for the shrub habitats (Low and Tall Shrub), and PC1–Tasseled Cap Greenness–NDVI for Black Spruce and Scattered Woodland. Field was easily distinguishable in all band combinations. For each year assessed, I created training areas of at least 100 pixels and three training areas per habitat class. I examined each year individually and assigned training areas within the same geographic region, assuming that edges shift slowly over time but the center of a vegetation patch remains the same.

#### *Accuracy assessment*

I randomly generated points ( $n = 70$ ) in ArcMap and visited them in summer 2010. At each point, I took photographs in the four cardinal directions and classified vegetation within a 50-m radius according to Viereck et al. (1992) and Kessel (1979), as well as within a 15-m radius for compatibility with spatial resolution of Landsat data. I took additional reference points ( $n = 40$ ) in clearly-defined habitat patches. Finally, I excluded points that had  $> 2$  distinct habitats within a 50-m radius because the characteristic patchiness of the northwestern boreal forest is known to cause problems with remote sensing classification.

I used the remaining field validation points ( $n = 102$ ; Fig. 2.1) to create an error matrix by extracting the pixel values from the habitat classification map. All classified images were generated using identical processing methods; therefore,

because no historical data were available to assess the accuracy of older images, I made the assumption that the overall accuracy, as estimated for the 2009 classification results, was a reasonable accuracy representation of all classified products.

### *Change in NDVI and NDMI*

To evaluate change in NDVI and NDMI, I used site knowledge to select 6–7 points in each habitat class. From these points, I calculated mean NDVI and NDMI values for each class in each year, and performed a linear regression to assess trends from 1984–2009.

I also created NDVI and NDMI difference images using simple subtraction, on a pixel-by-pixel basis, using the years 1985 and 2009. A symmetrical number of change bins were created by dividing the resulting positive and negative values into a pre-assigned number (10) of evenly spaced bins.

## **2.4 RESULTS**

### **Habitat classification and change detection**

The habitat classification maps demonstrated considerable inter-annual variation (Figure 2.6). The year 1992, in particular, yielded inconsistent classification results; more area was classified as Tall Shrub than expected, and less was classified as Scattered Woodland, Mixed Forest, Black Spruce, and Deciduous Forest (Fig. 2.7). The unusually low NDVI and NDMI values observed might be explained by the fact that 1992 had an extremely dry August, with only 28% of the average monthly precipitation (ARCR, 2012).

Regardless of inter-annual variation, some trends emerged from examining habitat change between 1985 and 2009 (Table 2.2). In 1985, Black Spruce was the most dominant vegetation type at Creamer's Refuge, followed by open forest

(Scattered Woodland) and shrublands (Tall and Low Shrub). Over the following 24-year period, there was a shift towards habitats with higher tree densities (Scattered Woodland and Mixed Forest; Fig. 2.7). Mixed Forest, Scattered Woodland and Coniferous Forest increased substantially (by 51%, 40% and 24% respectively), while Black Spruce and Deciduous Forest declined (by 52% and 24%). Tall Shrub remained relatively consistent over time although there was a notable peak observed in 1992 (Fig. 2.7; discussed above). Low Shrub increased until 1999, but subsequently declined. Change in Field was inconsistent between years.

The most distinctive changes observed in the imagery during this time period were human-caused; the creation of waterfowl and wildlife habitat in the 1980s and early 1990s resulted in noticeably high values for early successional vegetation stages (Field and Low Shrub), particularly in 1999. As time went on, the majority of these pixels changed to either Tall Shrub or Scattered Woodland. The other easily observed change was in Deciduous Forest, which expanded outward into the surrounding shrublands (Fig. 2.8). This change, although visible from the imagery, was obscured in the results because pixels classified as Deciduous Forest converted to Mixed Forest or Scattered Woodland in other areas of Creamer's Refuge.

### **Accuracy assessment**

The overall accuracy for the 2009 habitat classification map was 70.6% (Table 2.3). Habitats such as Field and Black Spruce mapped well (producer's and user's accuracy > 75%), while others yielded poor results. In particular, Mixed Forest mapped poorly (producer's and user's accuracy < 40%), as did Scattered Woodland and Tall Shrub, which showed considerable overlap.

### Change in NDVI and NDMI

Because 1992 had a dry August and unusually low NDVI and NDMI values, I excluded it from the NDVI and NDMI change analyses. When the entire 1057-ha Creamer's Refuge was assessed on a pixel-by-pixel basis, I documented a mean negative trend of  $-0.0033 \text{ NDVI yr}^{-1}$ , for a total decrease of 0.090 over the 24-year time series ( $r^2 = 0.685$ ;  $P = 0.042$ ). The overall decline recorded for NDMI ( $-0.0019 \text{ NDMI yr}^{-1}$ ) was not significant ( $r^2 = 0.470$ ;  $P = 0.132$ ).

Although the overall trends were negative for both NDVI and NDMI, the change observed was not uniform across the landscape. NDVI decreased over 94.8% of Creamer's Refuge between 1985 and 2009 but increased at 5.2%, mostly in the Deciduous Forest (Fig. 2.9A). In contrast, NDMI increased in 16.6% of Creamer's Refuge (Fig. 2.9B); it showed a decline for Coniferous Forest, a more extensive positive response in Deciduous Forest, and an increase in Black Spruce that was not observed on the NDVI change map. Both index change maps clearly demonstrated decreases in Fields and increases at the edge of Deciduous Forest, where birch trees matured and/or have expanded into shrubland (Fig. 2.8).

When looking at the selected habitat-specific points, all habitat classes showed decreases in NDVI (Table 2.4) and NDMI (Table 2.5) between 1984 and 2009, although trends for the Deciduous Forest and shrub (Low/ Tall) classes were not significant ( $P < 0.05$ ). The strongest negative trend in NDVI was documented for Field, while the most significant declines were documented for open spruce habitats (Black Spruce and Scattered Woodland). For NDMI, only Coniferous Forest and Scattered Woodland exhibited significant declines, and the strongest negative trends were documented for Field and Coniferous Forest. No change in NDMI was observed for Black Spruce, even though I documented a high

correlation between NDVI and NDMI values for all 50 points examined ( $r = 0.807$ ).

## 2.5 DISCUSSION

### **Successional change and habitat classification**

Despite the facts that the area of interest was smaller than usually examined and successional changes in vegetation are difficult to quantify (Cohen et al., 2002) using remote sensing, I was able to document subtle secondary successional changes in an interior Alaska boreal forest; my results showed that forest increased across the landscape with increases in tree density and a shift to more coniferous trees. Habitat delineation is not easy within boreal forest because it is a highly heterogenous landscape at all scales. Thus, I consider the overall accuracy of 71% using the image processing and selected classification regime successful. Using broader land cover classes would probably have resulted in better overall accuracy, but I would not have documented the availability of avian-specific habitat at Creamer's Refuge. In addition, I would likely have missed some of the fine-scale indications of successional change, which was one intent of this study.

In the absence of disturbance at this lowland boreal forest, I documented an overall shift towards denser forests, as well as a decline in open habitats (Black Spruce and Low Shrub) and Deciduous Forest between 1985 and 2009. Scattered Woodland, which is characterized by denser trees and a shrub understory, replaced Black Spruce and Tall Shrub habitat, while much of the Deciduous Forest was converted to Mixed Forest. Though both Scattered Woodland and Mixed Forest had poor accuracy (user's accuracy of 33% and 59%, respectively), coniferous trees dominated both these classes so positive changes observed in their direction indicates that the coniferous forest component increased over the landscape. This shift to coniferous is similar to



what has been documented at Denali National Park (Stueve et al., 2011; Roland et al., In press), although elsewhere in Alaska there has been reported loss of spruce habitat to deciduous due to increased fire frequency (Johnstone et al., 2010; Shenoy et al., 2011).

The largest and most consistent change at Creamer's Refuge over the 24-year time period was a decrease in Black Spruce habitat. This was somewhat surprising considering that Black Spruce is considered a stable, climax forest type (Viereck, 1970; Van Cleve et al., 1996). Black spruce trees tend to grow in low-lying areas where vegetation change is minimal because cold, water-logged soils and thick sphagnum moss prohibit the establishment of new trees and shrubs. In the absence of disturbance, paludification increases and soil temperatures tend to decrease over time due to increased insulation (Viereck et al., 1983). Because black spruce is one of the only species that can survive these prohibitive conditions (Hollingsworth et al., 2006), we might expect this vegetation type to increase over time rather than decrease. However, in this study, I observed a decrease in Black Spruce as it transitioned to Scattered Woodland. This could be explained partly by local topography and landscape placement, and partly by classification criteria. While Black Spruce decreased in prevalence across Creamer's Refuge over time, it became more concentrated in the lowest, coldest areas. In adjacent areas, however, where the soil might be slightly warmer and perhaps drier, other shrubs and trees could establish and grow over time, leading to a gradual increase in tree density and shrub understory. Ultimately, although black spruce might remain the dominant tree species, this type of change would result in a greater proportion of the landscape classified as Scattered Woodland. Scattered Woodland is a field-based habitat class primarily differentiated from Black Spruce by its shrub understory.

Another example of change that demonstrated differences between field-based and remote-sensing habitat classification was in Deciduous Forest. Little change over time was documented, although visible change in Deciduous Forest can be seen from examining the imagery (Fig. 2.7). This can be probably be attributed to the lack of distinction made (in habitat class assignment) between the young birch-willow forest of early years and the mature birch forest of later years. It is apparent from reading historical accounts (Spindler, 1976) that early successional willow-dominated habitats were previously more common at Creamer's Refuge. Likely, little overall change in Deciduous Forest was recorded because Deciduous Forest areas dominated by tall willow in 1984 changed to Scattered Woodland by 2009, while birch-dominated areas maintained their classification as Deciduous Forest as young birch trees matured.

The differences in scale between field and remote-sensing data could explain the poor accuracies observed for some for the heterogeneous habitat classes. In Landsat data, a single pixel value collectively describes all habitat components (e.g. proportion of deciduous to coniferous trees, wetness, etc.) within a 30-m x 30-m square area. Thus, for highly heterogenous landscapes like the boreal forest of Creamer's Refuge, most pixel values are the combined result of different spectral reflectances from multiple distinct patches, rather than a single land cover type. This means that remote sensing data will inherently be less accurate when compared to field-based data, although it is important to note that, even in the field, these classes are sometimes hard to distinguish. For example, a field-ascribed classification might be biased by exact location perspective (e.g. if the observer is next to a spruce tree or within a tall alder shrub).

Despite these limitations, remote sensing assessment gives us a landscape perspective that is rarely achievable using field data. By allowing us to view a larger spatial extent, remote sensing can be used to inventory and monitor areas

otherwise inaccessible. The lower spatial resolution of Landsat data at the pixel scale is compensated by the fact that there is a higher temporal resolution than that available from field data. This repeat coverage of the same area is especially important in northern latitudes where historical data are lacking. In this study, for example, assessing a span of years not only allowed us to record the consistent direction of change; it allowed us to document year-to-year variability and isolate unusual datasets such as 1992.

### **Remote sensing of avian habitat**

While remote sensing can advise research and add to our knowledge of avian habitat, it can not substitute completely for field measurements. Forest attributes were documented well in this study but shrub habitats, which are indisputably important for boreal breeding birds, were poorly resolved. Likely, remote sensing works best when paired with field assessments and ground-validation (Bayne et al., 2010). One of the primary problems that I encountered was that the bird-driven habitat classifications are somewhat different from habitat classes determined by spectral signatures, resulting in low accuracy for some distinctive avian habitats such as Tall Shrub and Mixed Forest. Partially, this is due to the spatial resolution (discussed above) but spectral resolution is also important because deciduous shrub are difficult to distinguish from birch trees. As a result, the signatures for Mixed (birch- spruce) Forest often overlapped with Tall (deciduous) Shrub, as well as with Scattered Woodland, which had a deciduous shrub understory.

Because habitat classes available from the classification do not necessarily reflect the ecological requirements of the organism under study (Gottschalk et al., 2005), using habitat classifications to model bird abundances might not be accurate enough to predict population change in response to warming. If we are specifically interested in monitoring or modeling populations of birds (rather than

vegetation or ecosystem change), another approach might be to exclude that secondary step of habitat classification (grouping birds into habitat associations and plants into vegetation groups) and instead make direct comparison between individual species occupancy or abundance data and satellite imagery. Other researchers have had success with this (Lavers et al., 1996; St-Louis et al., 2009), although there are questions about the ability to apply the remote sensing data to areas outside of the study area or to different year imagery (Nagendra, 2001).

### **Change in NDVI and NDMI**

Our findings provide support that NDVI has declined in recent decades in interior Alaska, and indicate that moisture balance may be shifting. Coniferous-dominated habitats, in particular, demonstrated the strongest negative trends in NDVI and NDMI over time, while Deciduous Forest and shrub classes did not demonstrate significant change. Many questions remain about what is causing these declines and how different vegetation types respond to climate warming, but the fact that all habitat types examined exhibited negative trends indicates that change is happening at a landscape scale.

Coniferous forests have demonstrated the strongest negative NDVI (Beck and Goetz, 2011) and growth trends associated with warming temperatures (Wilmking and Juday, 2005; Beck et al., 2011; Berner et al., 2011). This is especially true for white spruce in relatively hot, dry interior Alaska, which has shown temperature (Barber et al., 2000; Lloyd and Bunn, 2007) and water stress (McGuire et al., 2010). Even black spruce, which is the most widespread tree species, seems to be showing a climate response.

At Creamer's Refuge, we documented the most significant decreases in NDVI over time in the open black spruce habitats, Black Spruce and Scattered

Woodland. The assumption based on declining NDVI, that the health of lowland peatland black spruce has declined over time, is similar to what has been reported in other studies using NDVI values (Beck et al., 2011; Baird et al., 2012), but contradicts what Wilmking and Myers-Smith (2008) found using tree-ring analysis. Interestingly, the NDMI trend was not significant for Black Spruce, although it was for Scattered Woodland, indicating that the two indices may be reflective of different landscape changes. In fact, in the black-spruce dominated areas of Creamer's Refuge, the majority of pixels on the NDMI change map (but only a small portion on the NDVI map) exhibited increasing values over time (Fig. 2.9).

An alternative explanation for the observed decrease in NDVI and simultaneous increase in NDMI on Black Spruce could be that there was a shift in canopy cover between 1985 and 2009, as understory shrub increased over time. In our study area, shrub habitats had the lowest NDVI values of all habitat classes examined (besides Field), so a shift in this direction over time would reduce the NDVI, even if the health of the spruce trees had not declined. This explanation of increased shrub on Black Spruce is compatible with the results of our habitat classification, which showed a transition to more shrubby Scattered Woodland. Close visual comparison of the NDMI change and the habitat classification maps suggested this change as well, because the NDMI change map appeared to illuminate areas at Creamer's Refuge where canopy coverage increased rapidly or where deciduous trees expanded. Although our study did not adequately document this and more investigation is necessary to say anything conclusively, the NDMI change map roughly reflected the direction of habitat shift, with pixels that exhibited increases over time signifying areas of increased structural heterogeneity (often associated with deciduous canopy cover), and pixels exhibiting decreases signifying more coniferous trees present.

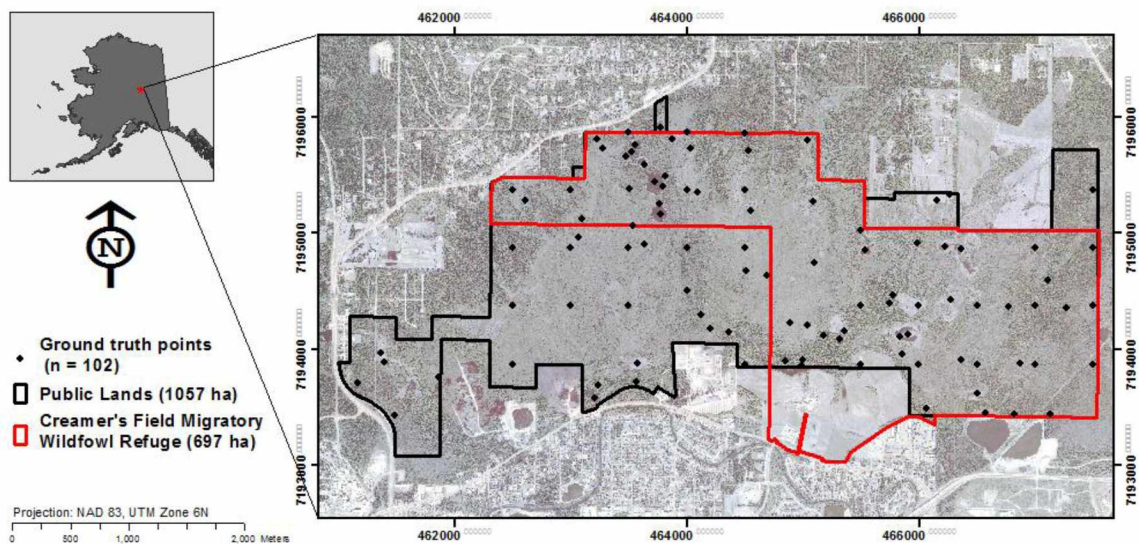
Collectively, our results imply that, at least for some areas, NDVI alone might not be a sufficient index for evaluating change in forest health over time. If, for example, we looked at NDVI trend alone, we would conclude that Black Spruce is declining in health at Creamer's Refuge; however, additional evidence (results from habitat classification and NDMI change map, as well as from documented change in vegetation (Table 1.1) and plant species composition for the 10-ha Black Spruce plot (Appendix 3)) shows that the decrease in NDVI might be a result of different landscape changes, specifically the increase in understory shrub over time. Several other researchers have questioned the usefulness of NDVI in demonstrating change in forest health for low canopy areas (Berner et al., 2011; McManus et al., 2012), because negative effects related to forest disturbance are confounded by positive change caused by green-up of understory vegetation. Because shrub and trees species are likely responding differently to the warming climate but contributing separately to the single pixel value, McManus (2012) concluded that NDVI trends were of questionable value when looking at climate change effects in sparsely-treed areas of the Arctic. This might well be the case for open areas of the boreal forest, although this study is the first to indicate this.

## **Conclusions**

We provide a local-scale assessment of environmental change in an understudied and rapidly changing region, the boreal forest of interior Alaska. Despite challenges associated with assessing a small area, remote sensing was effective at quantifying continuous forest dynamics related to natural succession. Our results highlight the importance of considering succession when investigating climate impacts because of the potential for vegetation changes to confound results. They also demonstrate the value in using a moisture index such as NDMI, particularly when examining open habitats where results from

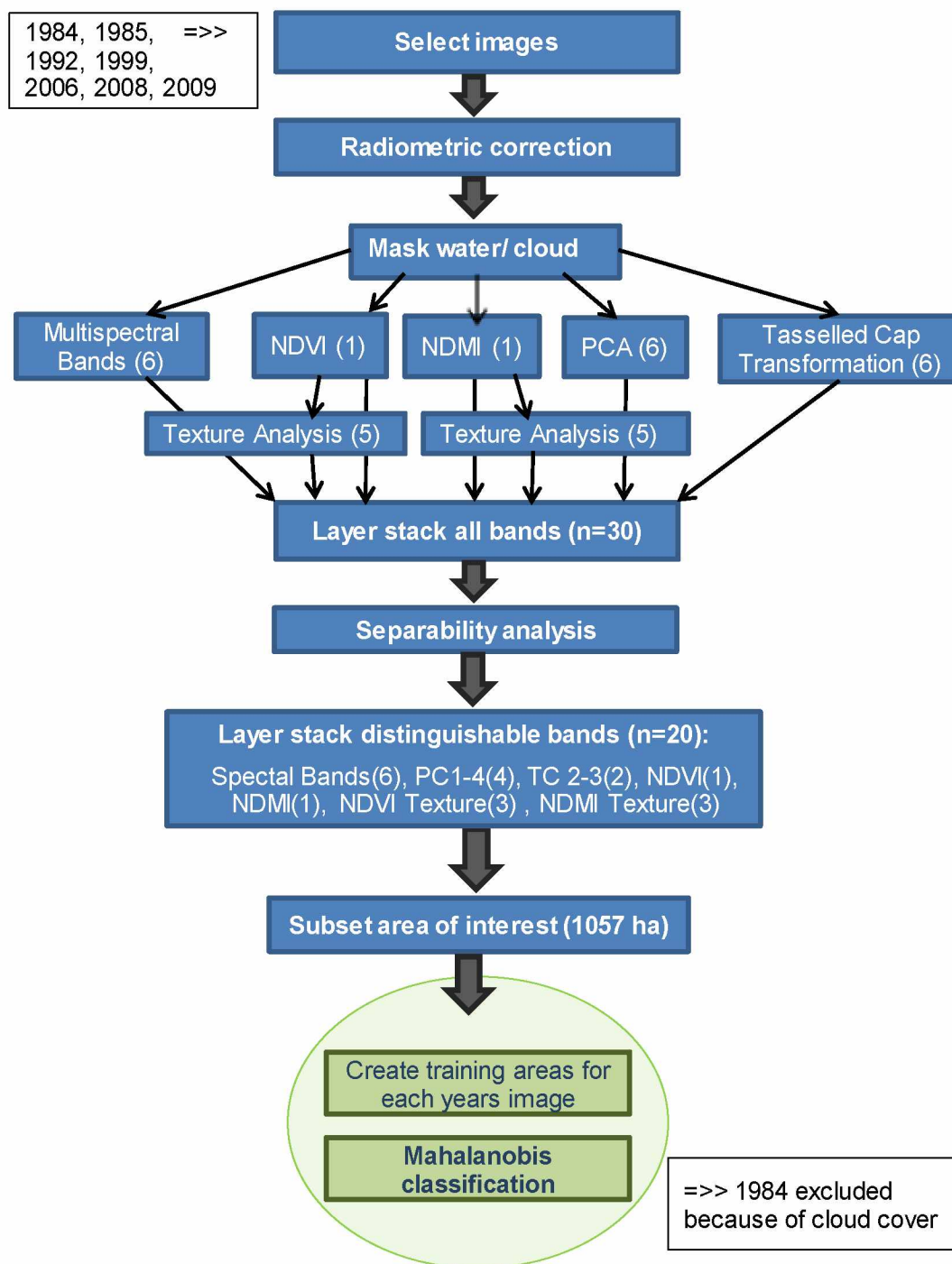
NDVI alone might be insufficient or misleading, such as in boreal wetlands or scattered treeline forests.

## 2.6 FIGURES

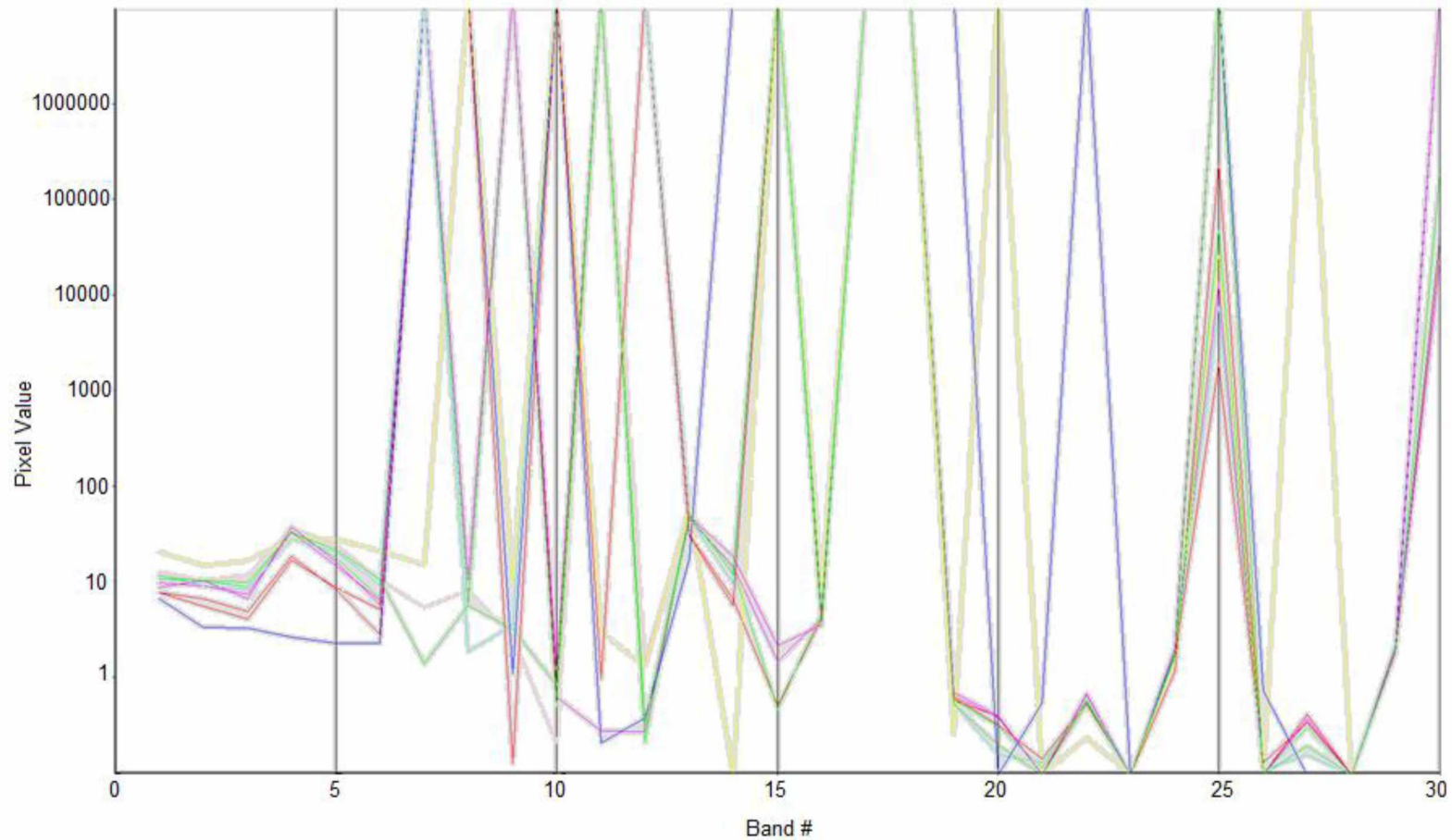


**Fig. 2.1** Study site (black outline) and field validation points (black dots) visited in 2010 at Creamer's Refuge, Fairbanks, Alaska. The study area and field validation points were selected for compatibility with avian research conducted at Creamer's Refuge in 2010-11 (Chapter 1). Field validation points were used to assess accuracy of the 2009 habitat classification generated using Landsat data.



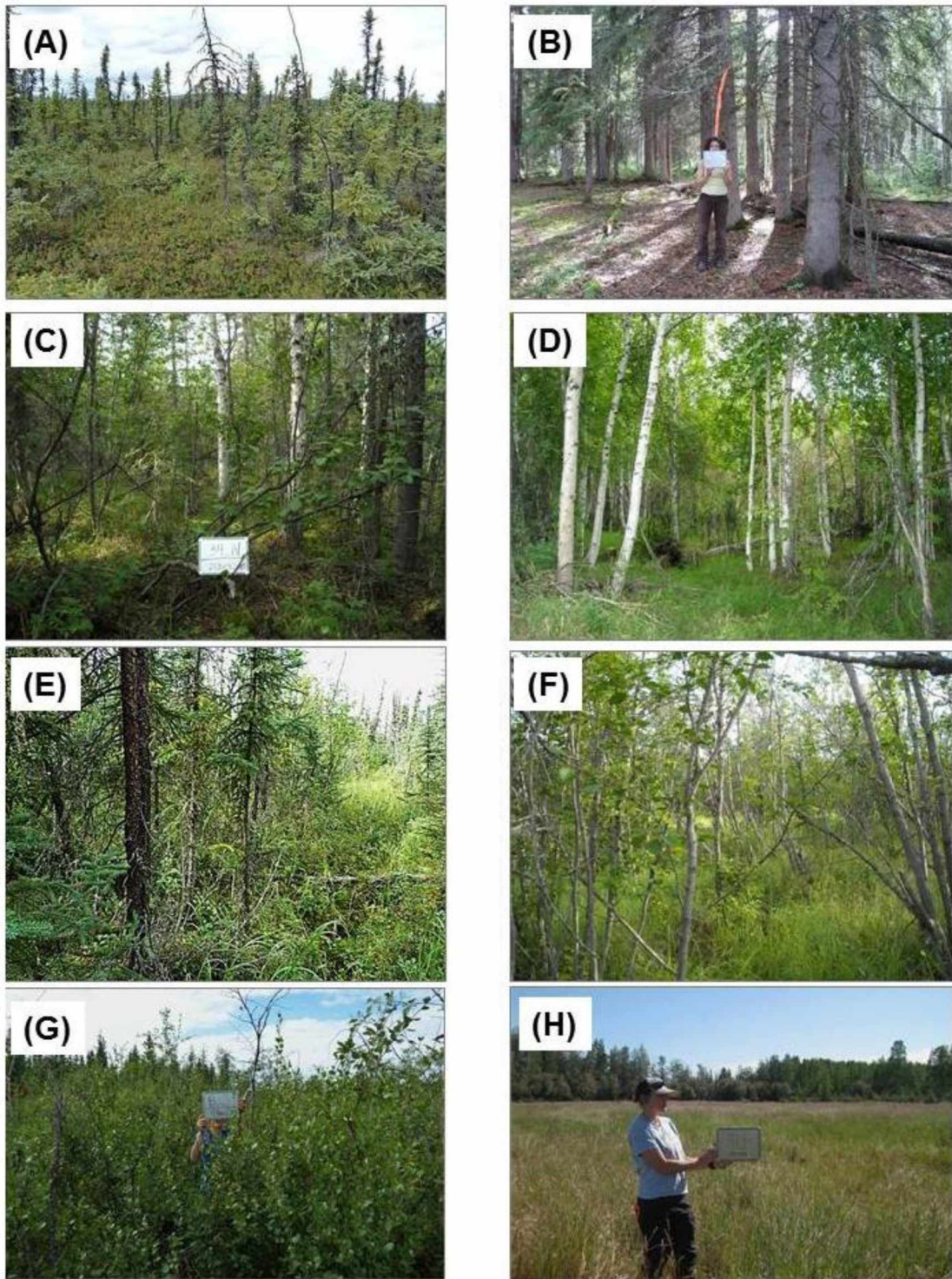


**Fig. 2.2** Flowchart of remote sensing methods used for habitat classification.

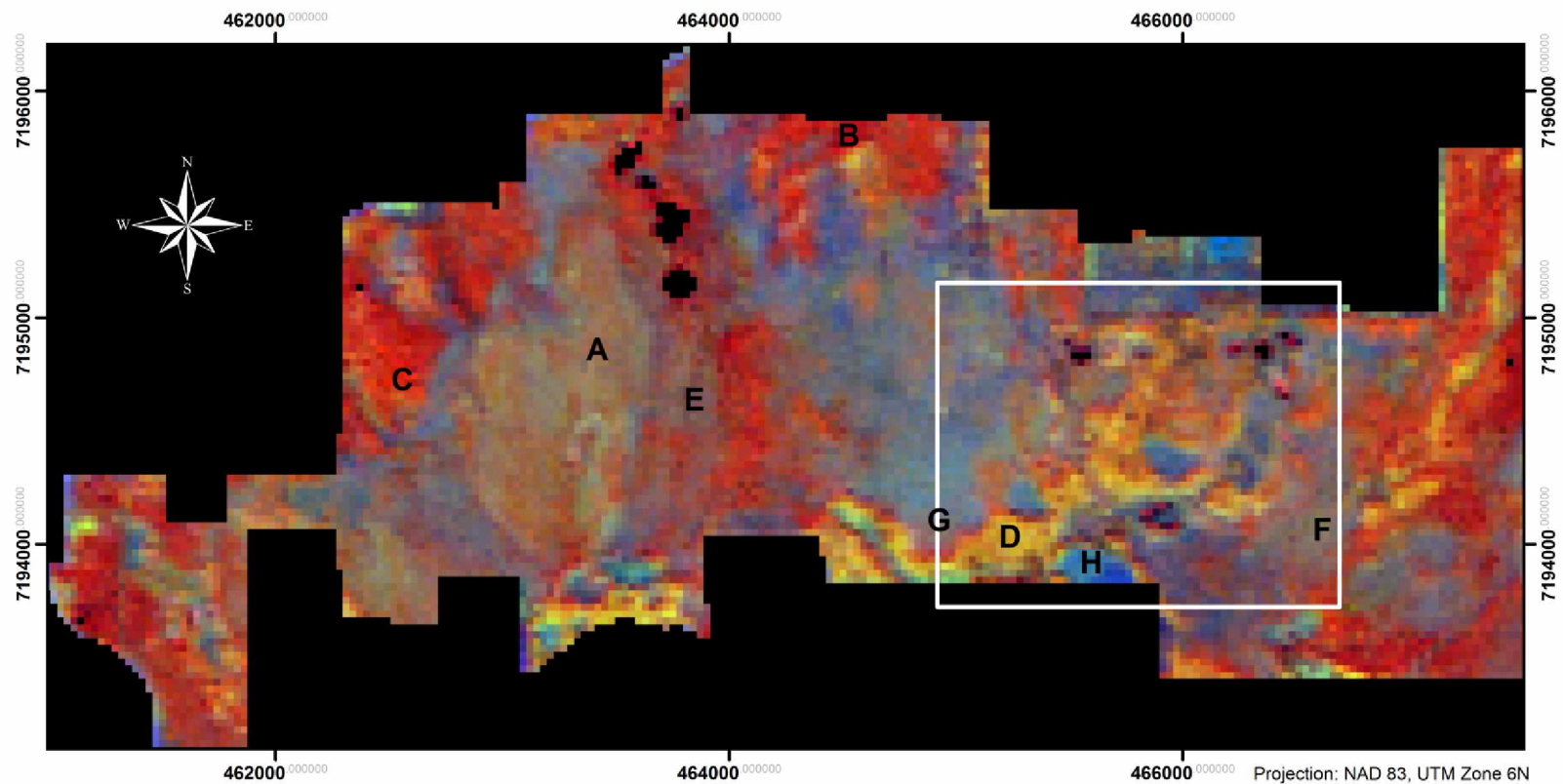


**Fig. 2.3** Spectral profiles showing pixel values in 30 different bands for 10 point locations (colored lines) at Creamer's Refuge, Alaska. I visually assessed these profiles to isolate redundant and noisy bands and, for the final 20-layer stack used for habitat classification, excluded bands such as band 16 (Tasseled Cap band 4) and band 29 (NDMI Texture Variance) that were indistinguishable for all points.



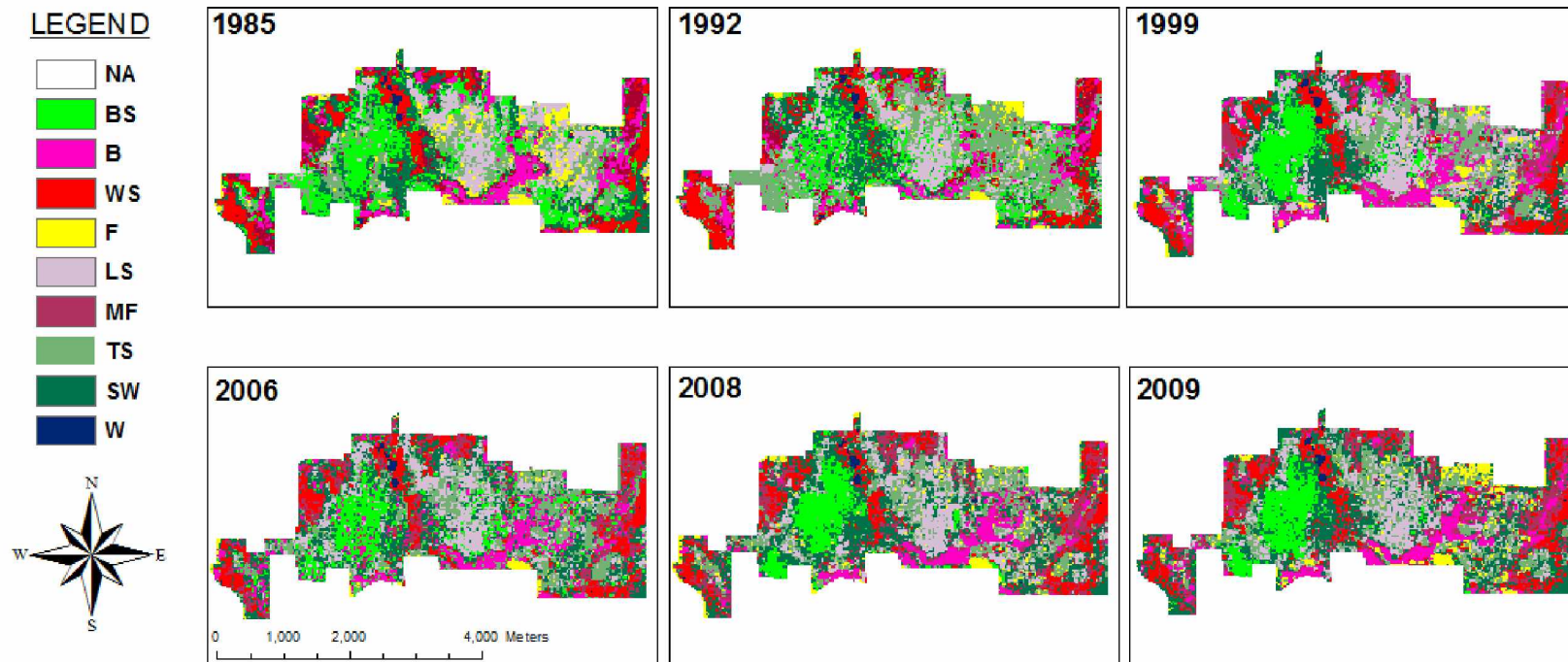


**Fig. 2.4** Vegetation communities used for avian habitat classification at Creamer's Refuge, Alaska: (a) Black Spruce; (b) Coniferous Forest; (c) Mixed Forest; (d) Deciduous (birch) Forest; (e) Scattered Woodland; (f) Tall Shrub; (g) Low Shrub; and (h) Field.

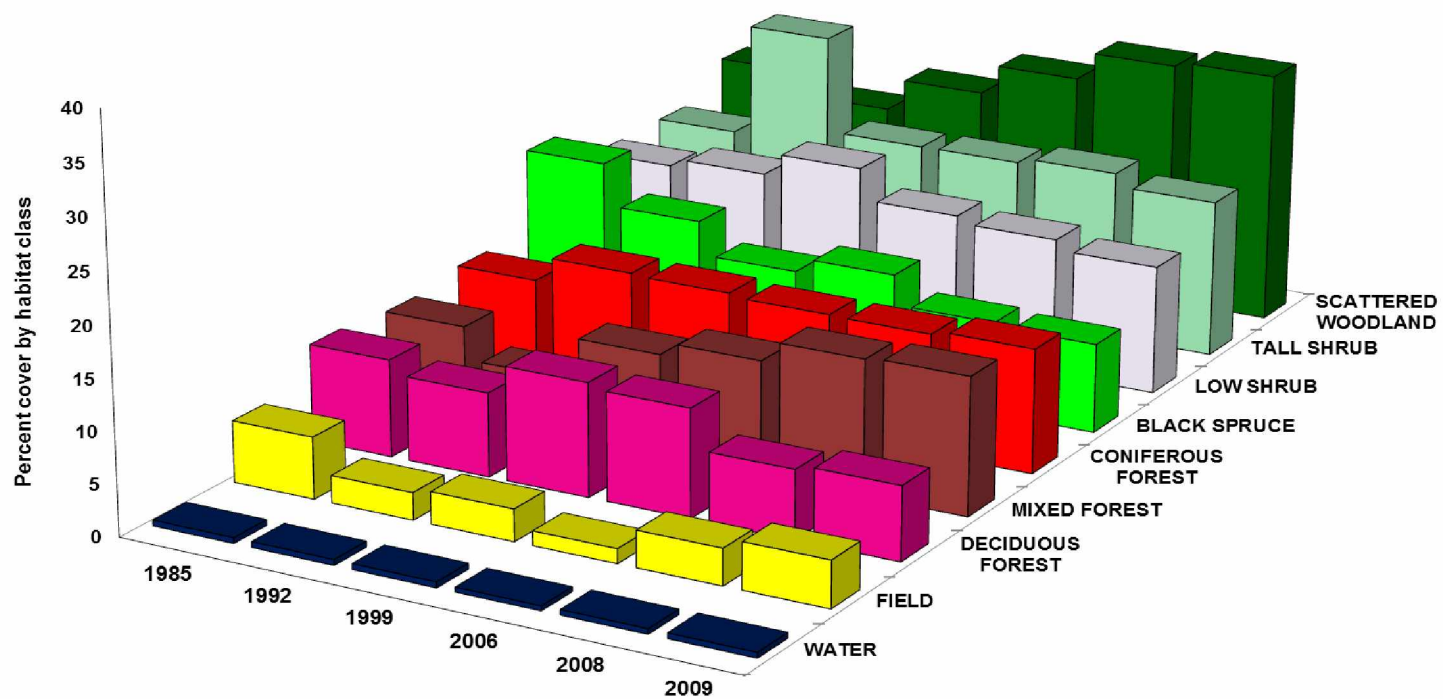


**Fig. 2.5** False color composite (NDMI-Landsat Band 4-Landsat Band 3) of 2 September 2009 Landsat image showing habitat classes at Creamer's Refuge, Alaska: (A) Black Spruce, (B) Coniferous Forest, (C) Mixed Forest, (D) Deciduous Forest, (E) Scattered Woodland, (F) Tall Shrub, (G) Low Shrub, and (H) Field. The white box shows an area of rapid change in deciduous forest (Fig. 2.8).

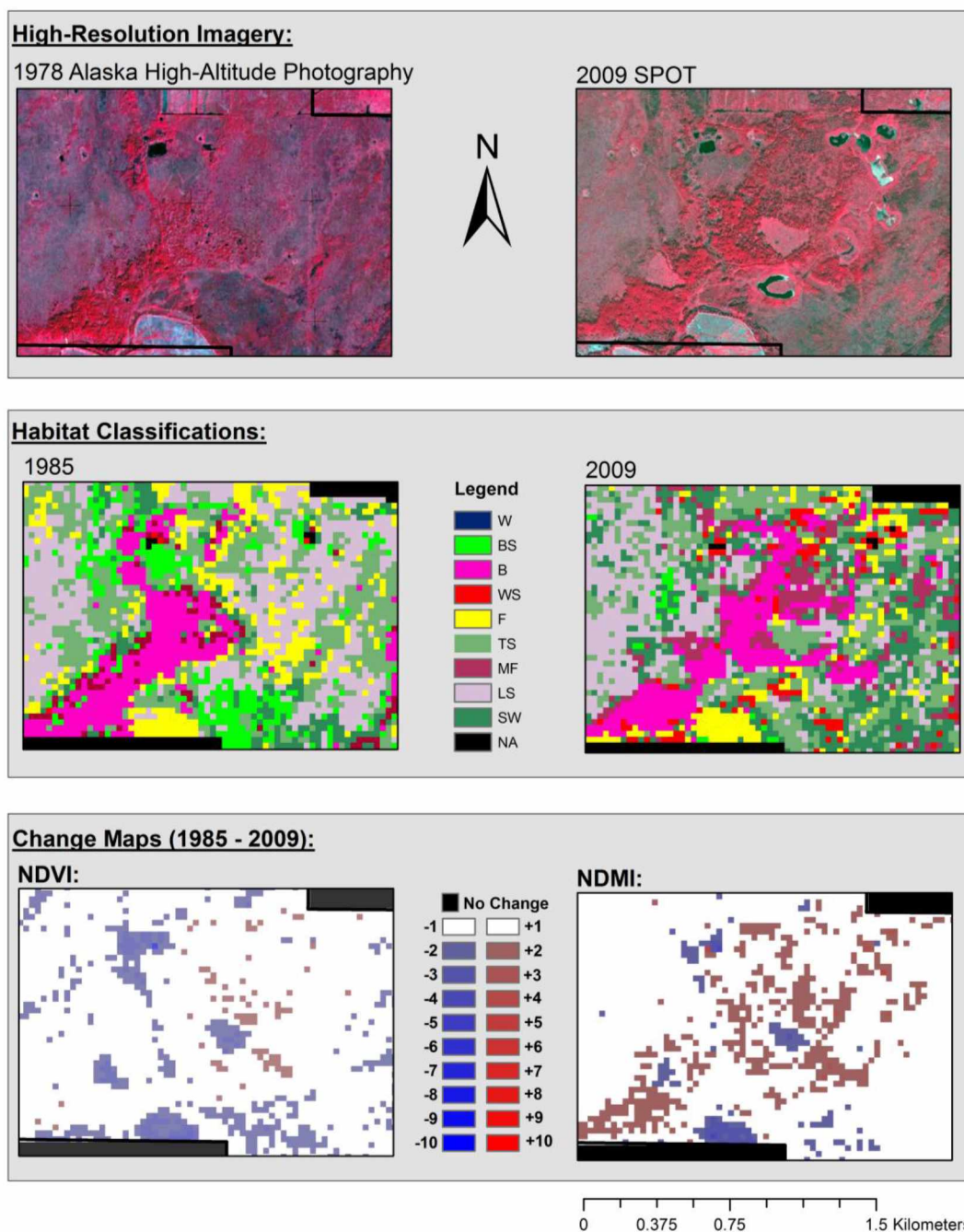




**Fig. 2.6** Habitat classification maps of Creamer's Refuge, Alaska, 1985–2009 ( $n = 6$  years). The classified area is the public lands subset (1057 ha) shown in Fig. 2.1. 'BS' is Black Spruce, 'B' is Deciduous Forest, 'WS' is Coniferous Forest, 'F' is Field, 'TS' is Tall Shrub, 'MF' is Mixed Forest, 'LS' is Low Shrub and 'SW' is Scattered Woodland, 'W' is Water.

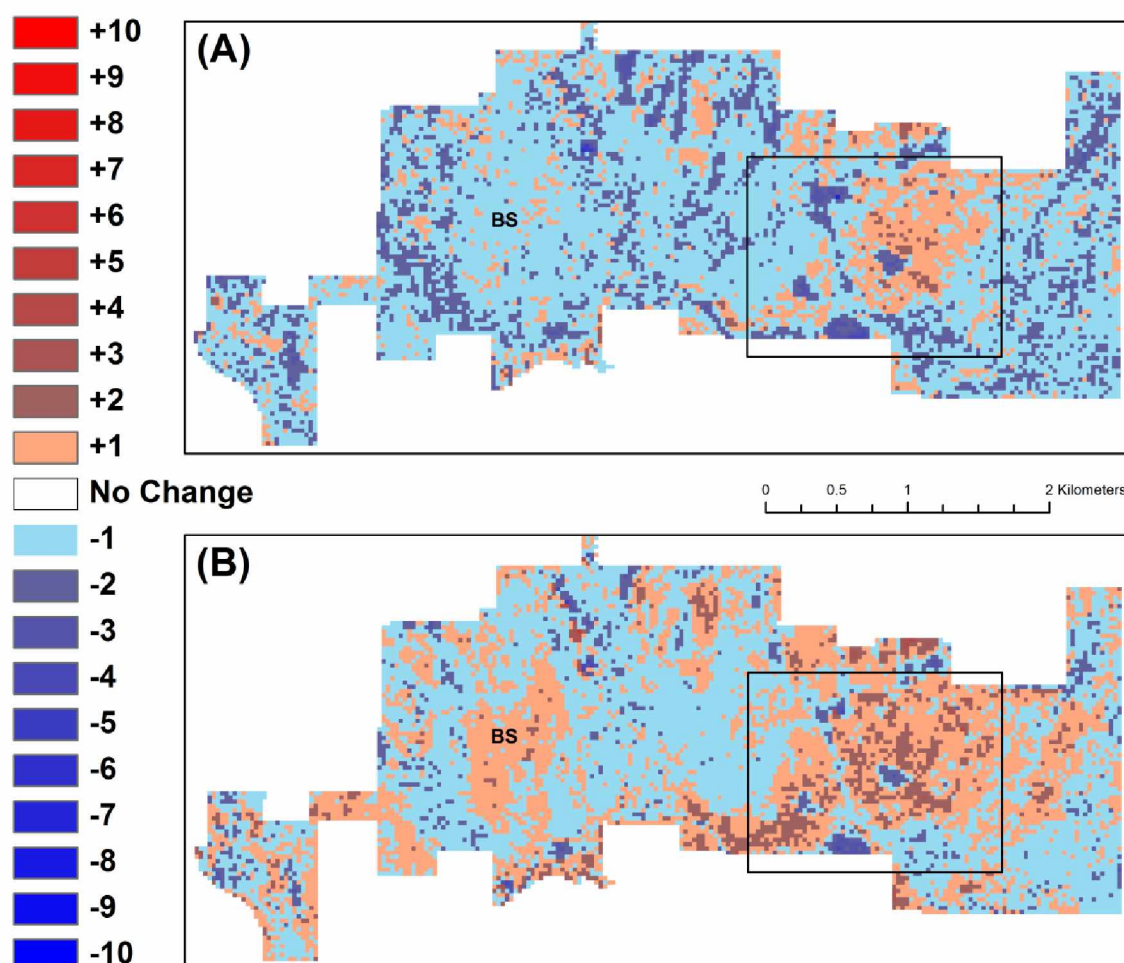


**Fig. 2.7** Land cover change by habitat class at Creamer's Refuge, Alaska, 1985–2009 ( $n = 6$  years). I excluded 1984 because 14.2% of the classified image was cloud.



**Fig. 2.8** Comparison of high-resolution imagery, habitat classifications, and NDVI and NDMI change maps (using 1985 and 2009 as input years) for a subset area of rapid change in deciduous (birch) forest at Creamer's Refuge, Alaska. For the classification maps, 'W' is Water, 'BS' is Black Spruce, 'B' is Deciduous Forest, 'WS' is Coniferous Forest, 'F' is Field, 'TS' is Tall Shrub, 'MF' is Mixed Forest, 'LS' is Low Shrub, and 'SW' is Scattered Woodland.





**Fig. 2.9** Image subtraction change maps for NDVI (A) and NDMI (B) at Creamer's Refuge, Alaska, between 1985 and 2009. Blue indicates areas of decline in the index value and orange shows areas of increase over time. 'BS' shows an area of concentrated Black Spruce habitat, and the black rectangular box show an area of rapid change in deciduous forest (Fig. 2.8).



**Table 2.1** Landsat images used for habitat classification and change detection at Creamer's Refuge, Alaska.

Year	Date	Satellite and sensor	Path/row	Comments
1984	August 28	Landsat 5-TM	70/14	some cloud; excluded from classification
1985	August 31	Landsat 5-TM	70/14	
1992	September 4	Landsat 4-TM	69/15	low precipitation; excluded from NDVI and NDMI analyses
1999	August 30	Landsat 7-ETM+	70/14	
2006	September 3	Landsat 5-TM	69/15	
2008	August 30	Landsat 5-TM	70/14	
2009	September 2	Landsat 5-TM	70/14	

**Table 2.2** Area (ha) and percent cover of nine habitat classes mapped at Creamer's Refuge, Alaska, 1985–2009 (n = 6 years). I excluded 1984 because 14.2% of the classified image was cloud cover.

	<b>1985</b>		<b>1992</b>		<b>1999</b>		<b>2006</b>		<b>2008</b>		<b>2009</b>		Relative change(%) (1985 - 2009)
	Area	%	Area	%	Area	%	Area	%	Area	%	Area	%	
Black Spruce	0	18.1	0.0	13.9	0.0	10.6	0.0	12.0	0.0	9.1	0.0	8.8	<b>-51.7</b>
Coniferous Forest	0	9.9	0.0	12.4	0.0	12.2	0.0	11.9	0.0	11.8	0.0	12.2	<b>23.5</b>
Mixed Forest	0	9.0	0.0	5.7	0.0	9.9	0.0	11.2	0.0	13.2	0.0	13.5	<b>50.5</b>
Deciduous Forest	0	9.5	0.0	8.2	0.0	11.1	0.0	10.6	0.0	6.7	0.0	7.2	<b>-24.2</b>
Scattered Woodland	0	17.8	0.0	15.3	0.0	18.6	0.0	21.7	0.0	24.5	0.0	25.0	<b>40.3</b>
Tall Shrub	0	14.6	0.0	26.0	0.0	16.3	0.0	16.3	0.0	16.9	0.0	15.6	<b>6.5</b>
Low Shrub	0	14.5	0.0	15.3	0.0	17.5	0.0	14.4	0.0	13.7	0.0	12.7	<b>-12.5</b>
Field	0	6.0	0.0	2.7	0.0	3.1	0.0	1.4	0.0	3.6	0.0	4.6	<b>-24.3</b>
Water	0	0.5	0.0	0.5	0.0	0.6	0.0	0.5	0.0	0.5	0.0	0.5	<b>-4.5</b>

**Table 2.3** Classification accuracy for 2009 habitat classification map of Creamer's Refuge, Alaska.

Habitat Class <sup>1</sup>	Black Spruce	Closed Conifer	Mixed Forest	Deciduous Forest	Scattered Woodland	Tall Shrub	Low Shrub	Field	Water	User's Accuracy
Black Spruce (6)	6	-	-	-	-	-	-	-	-	100.0
Coniferous Forest (14)	-	10	1	-	2	-	-	-	1	71.4
Mixed Forest (9)	1	-	3	2	3	-	-	-	-	33.3
Deciduous Forest (12)	-	-	1	10	-	1	-	-	-	83.3
Scattered Woodland (17)	1	1	3	-	10	-	-	-	2	58.8
Tall Shrub (12)	-	-	-	1	1	7	2	-	1	58.3
Low Shrub (13)	-	-	-	-	1	2	10	-	-	76.9
Field (12)	-	-	-	-	1	1	-	10	-	83.3
Water (7)	-	1	-	-	-	-	-	-	6	85.7
Producer's Accuracy	75.0	83.3	37.5	76.9	55.6	63.6	83.3	100.0	60.0	<b>70.6</b>

<sup>1</sup> The number shown in parentheses represents the number of points sampled in each habitat class (n = 102).

**Table 2.4** Linear regressions of mean NDVI from 1984–2009 ( $n = 6$  years) for eight habitat classes. Classes are listed by ascending mean NDVI values, from highest (Deciduous Forest) to lowest (Field). I excluded 1992 because of unusually low August precipitation.

Habitat Class <sup>1</sup>	$R^2$	Slope	$P$ -value
Deciduous Forest (7)	0.54	-0.0030	0.096
Mixed Forest (6)	0.68	-0.0044	0.043
Black Spruce (6)	0.96	-0.0028	0.001
Scattered Woodland (6)	0.94	-0.0042	0.001
Coniferous Forest (7)	0.67	-0.0034	0.047
Low Shrub (6)	0.63	-0.0049	0.059
Tall Shrub (6)	0.65	-0.0047	0.053
Field (6)	0.86	-0.0073	0.008
All points (50)	0.84	-0.0042	0.010

<sup>1</sup> The number shown in parentheses is the number of individual points in each habitat class used to calculate the mean NDVI for each year.

**Table 2.5** Linear regressions of mean NDMI from 1984–2009 ( $n = 6$  years) for eight habitat classes. Classes are listed by ascending mean NDMI values, from highest (Mixed Forest) to lowest (Field). 1992 was excluded because of unusually low August precipitation.

Habitat Class <sup>1</sup>	R <sup>2</sup>	Slope	P-value
Mixed Forest (6)	0.23	-0.0011	0.338
Deciduous Forest (7)	0.01	-0.0004	0.849
Coniferous Forest (7)	0.94	-0.0043	0.001
Scattered Woodland (6)	0.84	-0.0032	0.010
Black Spruce (6)	0.01	-0.0003	0.837
Low Shrub (6)	0.49	-0.0027	0.120
Tall Shrub (6)	0.50	-0.0036	0.116
Field (6)	0.61	-0.0060	0.067
All points (50)	0.60	-0.0026	0.071

<sup>1</sup> The number shown in parentheses is the number of individual points in each habitat class used to calculate the mean NDMI for each year.

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## OVERALL CONCLUSIONS

Over a 35-year time period, plant and avian succession occurred on all the habitat plots I examined. I observed a close link between local vegetation and avian community changes, suggesting that birds are able to rapidly respond to environmental change and shift usage based on habitat availability. In addition, I retrospectively detected forest succession using both field data (Chapter 1) and remote sensing (Chapter 2) despite low expectations for the suitability of using remotely-sensed data at this limited spatial extent.

While my research did not separate climate-induced change from other causes such as succession, I identified habitat types most vulnerable to short-term change and described the community shifts associated with this change. My data provide an excellent baseline for addressing future questions about climate and successional change in the boreal forest ecosystem of interior Alaska. For example, comparisons of the rates of transition between successional stages can be made to other areas of the boreal forest to see if the transition time has been affected by warming during the past 35 years. In addition, where the majority of boreal successional studies use a space-for-time substitution, here we have a true timeline of site-specific changes. Future studies at Creamer's Refuge would further illuminate the true nature of ecosystem changes in the interior Alaska boreal forest.

This site-specific research not only expands our current knowledge of avian communities and avian-habitat associations in interior Alaska; it shows that local habitat change can impact avian populations or, conversely, that population changes can reflect habitat availability. The close relationship observed between change in vegetation and in birds provides evidence that birds are indeed appropriate indicators of environmental change and allows for better predictions to be made regarding future populations of avian species in the boreal forest.

Regardless of climate effects, succession is a natural process; therefore, it is not surprising that, in the absence of disturbance at Creamer's Refuge, I observed a gradual shift over time towards coniferous-dominated habitats and denser forests. Tree and shrub density increased on open habitats such as Low Shrub and Black Spruce, resulted in a net loss of young shrubby habitats as trees matured and spread. Similar to what was observed in Denali National Park (Stueve et al., 2011; Roland et al., In press), deciduous forest did not increase across the landscape (Table 2.4 and Appendix 3); although individual birch trees grew and the structure of the deciduous forest changed over time (Table 1.1), it appears that the environmental conditions at Creamer's Refuge favor coniferous habitats. Even though this type of vegetation shift is gradual, secondary succession has direct ecological implications because early-successional habitats tend to have high species richness and diversity (Imbeau et al., 1999; Haney et al., 2008; Brooks and Bonter, 2010). As the shift continues at Creamer's Refuge towards more homogenous forested habitats, reduced habitat complexity will likely lead to a further loss of edge or shrub-associated avian species.

Despite low expectations for this size area, I documented an increase in coniferous forest using both field and remote sensing data, which indicates that the spatial and temporal scale was sufficient for documenting forest changes at Creamer's Refuge. Coniferous forests showed the most significant declines in NDVI and NDMI over time, providing additional support that the health of coniferous forests in interior Alaskan is declining. Because of the ability of remote sensing to assess structural changes in vegetation, even on a relative small, local scale, it has great potential in avian habitat monitoring and modeling. Such detectable changes in forest cover can drive changes in avian community composition and population because avian-habitat guilds are driven primarily by structural differences between habitats. This said, I would not have been able to



predict the change observed in avian populations at Creamer's Refuge over time by looking solely at remote sensing data. While remote sensing documented the increase in coniferous habitats, the loss of shrub habitats was not apparent and wetland habitat was not measured at all. NDMI helped marginally with this, but Landsat data was not good at distinguishing between some important avian habitats, e.g. between Field and Low Shrub, or Low and Tall Shrub, or between Mixed Forest and Tall Shrub.

In addition, though the observed overall direction of change was similar to what was documented by plot data, there was low consistency between years at the pixel level, and my results are less conclusive at the 10-ha plot scale than at the refuge scale (Appendix 5). This is most likely a result of inherent inaccuracies in remote sensing data due to variability among years, and a mixed pixel effect caused by extreme heterogeneity of the boreal forest landscape. At Creamer's Refuge, as in most boreal forest, there are often too many small vegetation patches within a 30-m<sup>2</sup> pixel of Landsat data to accurately map some of the important avian habitats. This is especially true of open habitats, where a single pixel value is a combined result of reflectances: canopy trees, patches of understory shrub, soil moisture, and small ponds all contribute to the output value. Even small patches of open water can have a strong influence on the value of a pixel.

Similarly, while Landsat data may be useful at a refuge- or landscape scale because coarse structure is an important driver of avian occupancy, it is probably not good at predicting bird abundances at the plot scale because the 30-m<sup>2</sup> pixel can not detect fine habitat details. For many bird species, shrub height and complexity are important in nest site and territory selection, but the spatial resolution is insufficient for determining these details. As well, individual species of plant might be important to birds because they provide different food resources

(Rodewald and Abrams, 2002; Strode, 2009; Wood et al., 2012), but are often difficult (if not impossible) to differentiate solely on spectral reflectance values. As well, the decrease in wetland-associated birds that I documented indicated that something happened over time at landscape scale, but wetness changes were not monitored by either habitat classifications, or by my plot measurements.

The above limitations regarding capabilities of Landsat data set are important to consider when designing long-term monitoring projects or thinking about using remote-sensing data. For future projects, we could use higher resolution datasets to extract important bird-habitat characteristics, such as moisture and shrub height. Although not without its own problems, light detecting and ranging (Lidar) data, for example, is good for vegetation height and plant structure. Lidar has potential for finer scale, more accurate habitat modelling; among other things, it has been used effectively for bird (Goetz et al., 2010; Swatantran et al., 2012) as well as butterfly habitat (Lefsky et al., 2002).

However, Landsat data does have many benefits. One of the primary draws, and the reason it was used in this study, is that this long-running dataset provides longer temporal coverage than any other satellite (Cohen and Goward, 2004). Using this dataset allowed me to look backwards at changes that had already occurred and, as operation continues in the future, means that the same processing could be used to assess habitat in the future, or even to extrapolate out from the study area. The repeat temporal coverage of the same area allowed me to be selective about which images I used (i.e. to select images within days of each other in order to minimize external causes of variability). In addition, the large dataset allowed me to develop at a trajectory, rather than a two-year snapshot, of change. This is important because, even when all attempts were made to make images directly comparable, uncontrollable inter-annual variation still resulted in large differences between years. If, for example, I had used 1992

data as a baseline, I would have come to faulty conclusions regarding vegetation changes over time solely because that year was an outlier due to external conditions, namely low monthly precipitation for August.

Another reason that Landsat data are preferable in time-change analyses is that the repeat coverage has been consistently collected from the same (or rather, several similar) satellite, which collects data regularly, from the same angle, with set wavelength receivers, and at same time of day. I initially examined two higher resolution datasets (1978 Alaska High Altitude Photograph and a 2002-3 Quickbird), but there was tremendous variation between these disparate datasets due to differences in the methods of data collection. Even with normalization, this made change detection results questionable. The consistency of Landsat data, on the other hand, meant less processing bias because it minimizes the number of input decisions required of the user during standardization, which made it easier to directly compare datasets.

### **Management implications**

Considering that successional habitat change is shown in my study and by others to have a strong impact on local avifauna, I feel that the role of vegetation is underappreciated in avian population monitoring and predictive modeling. In addition, although the findings are not conclusive, the results of my NDMI and NDVI change maps suggest that ground-level vegetational changes in open habitats can also affect remote sensing results, and are thus an important consideration in remote sensing analyses. Without considering successional changes, we might come to misleading conclusions regarding climate impacts. Even if vegetation changes are not related to a warming climate, we can not develop accurate timescales or predictions for future populations of plants or birds if we do not consider “natural” successional changes (see examples by Iverson et al., 2008 and Matthews et al., 2011).

I recommend that all avian-specific studies be complemented with detailed, repeatable habitat assessments which will help distinguish between local- and broad- scale trends. It is important for protocols to be well documented so they are repeatable at a later date; Spindler's study is testimony that you never know when or how your data will be applied or used. For population surveys repeated over time (e.g. North American Breeding Bird Survey), habitat change should be factored into the analysis of long-term trends. At least in the boreal forests on North America, some measure of wetland change also appears to be important, although the proper scale for this is yet to be determined.

Remote sensing does not provide an easy alternative to field sampling because of inherent limitations (discussed above) and the intensive computational processing and interpretation time required, but it does have direct and real applications and offers tremendous potential for modeling future change. Remote sensing is better even than ground assessments at evaluating a landscape simultaneously at several spatial scales, so it can provide important information about how landscape-scale processes influence species distribution patterns. It can also be useful for recognizing habitat variables that are important to boreal fauna, and is particularly appropriate for avian population studies because birds synergistically perceive and select habitat features at several scales (Wiens et al., 1987; Graf et al., 2005; Deppe and Rotenberry, 2008; Kuhn et al., 2011).

If we want to use remote sensing effectively for wildlife conservation and management, we need to first know which habitat classes or variables are important to animals and second, understand how vegetation changes will affect the local fauna. Many questions remain about which habitat variables are important for birds and much more information is needed about avian-habitat associations, especially here in the northwestern boreal forest. The fact that we don't understand spatial needs or complexity of habitat use on the breeding

grounds is considered a major problem with conserving migratory landbirds (Faaborg et al., 2010). We also need a greater understanding of how these important variables can be quantified and extracted using satellite imagery. Site-specific collaborative work between biologists and remote sensors should be done to develop protocols for quantifying these variables using remote sensing methods. For example, shrub height is an important nest site characteristic for many boreal bird species but is poorly quantified using remote sensing; future effort could be made to identify efficient methods for isolating this habitat variable. Once important habitat variables are identified, remote sensing can be used to develop accurate habitat-suitability maps and model bird occupancy based on current plant distributions. For predictive modeling and future climate change scenarios, these habitat suitability maps can then be applied to vegetation models developed by climate change researchers.

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**Appendix 1** Summer bird list for Creamer's Refuge, Alaska, including names and codes of 86 bird species documented in 1975 and 2010-11 during the breeding season (15 May –1 July). \* denotes summer visitants that were seen during this period, but not thought to be breeding in the study area.

Common Name	Scientific Name	CODE	1975	2010-11
Canada Goose	<i>Branta canadensis</i>	CANG	*	*
Trumpeter Swan	<i>Cygnus buccinator</i>	TRUS	-	*
Gadwall	<i>Anas strepera</i>	GADW	*	-
American Wigeon	<i>Anas americana</i>	AMWI	X	X
Mallard	<i>Anas platyrhynchos</i>	MALL	X	X
Northern Shoveler	<i>Anas clypeata</i>	NSHO	X	X
Blue-winged Teal	<i>Anas discors</i>	BWTE	*	*
Northern Pintail	<i>Anas acuta</i>	NOPI	X	X
Green-winged Teal	<i>Anas crecca</i>	GWTE	X	X
Ring-necked Duck	<i>Aythya collaris</i>	RNDU	-	X
Lesser Scaup	<i>Aythya affinis</i>	LESC	X	X
Bufflehead	<i>Bucephala albeola</i>	BUFF	X	X
Common Goldeneye	<i>Bucephala clangula</i>	COGO	*	X
Sharp-tailed Grouse	<i>Tympanuchus phasianellus</i>	STGR	X	-
Ruffed Grouse	<i>Bonasa umbellus</i>	RUGR	X	X
Spruce Grouse	<i>Falcipennis canadensis</i>	SPGR	X	X
Horned Grebe	<i>Podiceps auritus</i>	HGR	X	X
Red-necked Grebe	<i>Podiceps grisegena</i>	RNGR	X	X
Northern Harrier	<i>Circus cyaneus</i>	NOHA	X	X
Sharp-shinned Hawk	<i>Accipiter striatus</i>	SSHA	X	X
Northern Goshawk	<i>Accipiter gentilis</i>	NOGO	X	X
Swainson's Hawk	<i>Buteo swainsoni</i>	SWHA	*	-
Red-tailed Hawk	<i>Buteo jamaicensis</i>	RTHA	X	X
Sandhill Crane	<i>Grus canadensis</i>	SACR	X	X
Solitary Sandpiper	<i>Tringa solitaria</i>	SOSA	X	X
Lesser Yellowlegs	<i>Tringa flavipes</i>	LEYE	X	X
Wilson's Snipe	<i>Gallinago delicata</i>	WISN	X	X
Red-necked Phalarope	<i>Phalaropus lobatus</i>	RNPH	X	-
Bonaparte's Gull	<i>Chroicocephalus philadelphia</i>	BOGU	-	X
Mew Gull	<i>Larus canus</i>	MEGU	X	X
Rock Pigeon	<i>Columba livia</i>	ROPI	X	X
Great Horned Owl	<i>Bubo virginianus</i>	GHOW	X	X
Northern Hawk Owl	<i>Surnia ulula</i>	NHOW	X	-
Great Gray Owl	<i>Strix nebulosa</i>	GGOW	*	-
Short-eared Owl	<i>Asio flammeus</i>	SEOW	X	-
Boreal Owl	<i>Aegolius funereus</i>	BOOW	X	-
Rufous Hummingbird	<i>Selasphorus rufus</i>	RUHU	*	-
Belted Kingfisher	<i>Megasceryle alcyon</i>	BEKI	*	*
Downy Woodpecker	<i>Picoides pubescens</i>	DOWO	X	X
Hairy Woodpecker	<i>Picoides villosus</i>	HAWO	X	X
American Three-toed Woodpecker	<i>Picoides dorsalis</i>	ATTW	X	X
Black-backed Woodpecker	<i>Picoides arcticus</i>	BBWO	*	*
Northern Flicker ('Yellow-shafted')	<i>Colaptes auratus</i>	NOFL	X	X
American Kestrel	<i>Falco sparverius</i>	AMKE	X	X
Olive-sided Flycatcher	<i>Contopus cooperi</i>	OSFL	X	-
Western Wood-Pewee	<i>Contopus sordidulus</i>	WEWP	X	*
Alder Flycatcher	<i>Empidonax alnorum</i>	ALFL	X	X
Hammond's Flycatcher	<i>Empidonax hammondi</i>	HAFL	X	X
Say's Phoebe	<i>Sayornis saya</i>	SAPH	X	-
Northern Shrike	<i>Lanius excubitor</i>	NSHR	X	*
Gray Jay	<i>Perisoreus canadensis</i>	GRAJ	X	X

## Appendix 1 continued.

Common Name (continued)	Scientific Name	CODE	1975	2010-11
Black-billed Magpie	<i>Pica hudsonia</i>	BBMA	*	X
Common Raven	<i>Corvus corax</i>	CORA	X	X
Tree Swallow	<i>Tachycineta bicolor</i>	TRES	X	X
Violet-green Swallow	<i>Tachycineta thalassina</i>	VGSW	X	X
Bank Swallow	<i>Riparia riparia</i>	BANS	*	*
Cliff Swallow	<i>Petrochelidon pyrrhonota</i>	CLSW	X	X
Black-capped Chickadee	<i>Poecile atricapillus</i>	BCCH	X	X
Boreal Chickadee	<i>Poecile hudsonicus</i>	BOCH	X	X
Ruby-crowned Kinglet	<i>Regulus calendula</i>	RCKI	X	X
Gray-Cheeked Thrush	<i>Catharus minimus</i>	GCTH	X	*
Swainson's Thrush	<i>Catharus ustulatus</i>	SWTH	X	X
Hermit Thrush	<i>Catharus guttatus</i>	HETH	X	*
American Robin	<i>Turdus migratorius</i>	AMRO	X	X
Varied Thrush	<i>Ixoreus naevius</i>	VATH	X	*
Bohemian Waxwing	<i>Bombycilla garrulus</i>	BOWA	X	X
Northern Waterthrush	<i>Parkesia noveboracensis</i>	NOWA	X	X
Orange-crowned Warbler	<i>Oreothlypis celata</i>	OCWA	X	X
Yellow Warbler	<i>Setophaga petechia</i>	YWAR	X	X
Blackpoll Warbler	<i>Setophaga striata</i>	BLPW	X	X
Yellow-rumped Warbler ('Myrtle')	<i>Setophaga coronata</i>	YRWA	X	X
Townsend's Warbler	<i>Setophaga townsendi</i>	TOWA	-	X
Wilson's Warbler	<i>Cardellina pusilla</i>	WIWA	X	X
American Tree Sparrow	<i>Spizella arborea</i>	ATSP	X	-
Chipping Sparrow	<i>Spizella passerina</i>	CHSP	*	-
Savannah Sparrow	<i>Passerculus sandwichensis</i>	SAVS	X	X
Fox Sparrow	<i>Passerella iliaca</i>	FOSP	X	X
Lincoln's Sparrow	<i>Melospiza lincolnii</i>	LISP	X	X
White-crowned Sparrow	<i>Zonotrichia leucophrys</i>	WCSP	X	X
Dark-eyed Junco	<i>Junco hyemalis</i>	DEJU	X	X
Red-winged Blackbird	<i>Agelaius phoeniceus</i>	RWBL	-	*
Rusty Blackbird	<i>Euphagus carolinus</i>	RUBL	X	X
Pine Grosbeak	<i>Pinicola enucleator</i>	PIGR	X	-
White-winged Crossbill	<i>Loxia leucoptera</i>	WWCR	X	X
Redpoll spp.	<i>Acanthis flammea/</i> <i>hornemanni</i>	CHRE	X	X
Pine Siskin	<i>Spinus pinus</i>	PISI	*	-

**Appendix 2** Number of breeding bird territories documented on 10-ha habitat plots in 1975 and 2011, at Creamer's Refuge, Fairbanks, Alaska. Birds are grouped by habitat associations (Sharbaugh 2007; Table 1.2). Numbers in bold indicate that evidence of breeding was documented for that species. \* denotes birds that were seen on plot inconsistently and not thought to be breeding there; these were included in the calculation of cumulative species richness but not species diversity. The Bray-Curtis index shows dissimilarity in avian community composition between years for each habitat plot.

	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
	1975	2011	1975	2011	1975	2011	1975	2011†	1975	2011	1975	2011
<b>CONIFEROUS FOREST (n = 9)</b>	0	1.75	2	7.5	1	7.25	12.5	15	6	6.25	21.5	37.75
American Three-toed Woodpecker	-	-	-	-	-	-	-	*	-	-	0	*
Gray Jay	-	0.25	-	<b>0.5</b>	-	*	<b>1</b>	*	1	<b>0.5</b>	<b>2</b>	<b>1.25</b>
Boreal Chickadee	-	-	-	0.5	-	-	<b>1</b>	2	-	-	<b>1</b>	2.5
Ruby-crowned Kinglet	-	-	-	0.25	-	-	<b>1</b>	<b>2</b>	<b>0.5</b>	<b>0.25</b>	<b>1.5</b>	<b>2.5</b>
Swainson's Thrush	-	*	0.5	<b>2.75</b>	1	<b>4.25</b>	4.5	<b>4.5</b>	-	*	6	<b>11.5</b>
Varied Thrush	-	-	-	-	-	-	*	*	-	-	*	*
Townsend's Warbler	-	-	-	-	-	-	-	2	-	-	0	2
Dark-eyed Junco	-	<b>1.5</b>	1	<b>3.5</b>	-	<b>3</b>	<b>4.5</b>	4.5	<b>4.5</b>	<b>5.5</b>	<b>10</b>	<b>18</b>
Pine Grosbeak	-	-	0.5	-	-	-	<b>0.5</b>	-	-	-	<b>1</b>	0
<b>MIXED FOREST (n = 5)</b>	0	0.75	0	0.75	0.5	2.5	0	2.5	1	1	1.5	7.5
Sharp-shinned Hawk	-	-	-	-	-	*	-	-	-	-	0	*
Great Horned Owl	-	-	-	-	-	-	-	*	-	-	0	*
Hairy Woodpecker	-	-	-	*	-	*	-	0.5	-	-	0	0.5
Downy Woodpecker	-	-	-	-	-	-	-	*	-	-	0	*
American Robin	-	<b>0.75</b>	-	<b>0.75</b>	0.5	<b>2.5</b>	-	2	<b>1</b>	<b>1</b>	<b>1.5</b>	<b>7</b>
<b>DECIDUOUS FOREST (n = 5)</b>	0	0.75	*	3.75	*	14.25	3	6	2.5	1.75	5.5	26.5
American Kestrel	-	-	-	-	-	-	-	-	-	-	*	0
Northern Flicker	-	-	-	-	*	*	-	-	-	-	*	*
Hammond's Flycatcher	-	*	*	-	-	<b>8.25</b>	-	2	-	-	*	<b>10.25</b>
Black-capped Chickadee	-	*	-	0.5	-	<b>1.5</b>	-	<b>0.5</b>	-	-	0	<b>2.5</b>
Yellow-Rumped Warbler	-	0.75	-	<b>3.25</b>	-	4.5	<b>3</b>	3.5	<b>2.5</b>	<b>1.75</b>	<b>5.5</b>	<b>13.75</b>
<b>SHRUB (n = 13)</b>	10.5	10	28.5	3	23.5	4.75	8.5	*	6.5	0.5	77.5	18.25
Alder Flycatcher	0.5	<b>2</b>	1.5	1	1.5	<b>1.25</b>	0.5	-	-	-	4	<b>4.25</b>
Gray-cheeked Thrush	*	-	2	-	1.5	-	1.5	-	2.5	-	7.5	0
Northern Shrike	-	*	-	-	-	-	-	-	-	-	0	*
Orange-crowned Warbler	1	<b>4</b>	*	<b>2</b>	1	<b>2</b>	*	-	-	-	2	<b>8</b>
Blackpoll Warbler	-	*	3	-	-	-	-	*	-	-	3	*
Yellow Warbler	-	0.25	<b>9</b>	-	<b>8</b>	*	-	-	-	-	<b>17</b>	0.25
Wilson's Warbler	-	-	-	-	*	-	1	*	-	-	1	*
Northern Waterthrush	-	-	<b>3</b>	-	3	0.5	<b>1</b>	-	-	-	<b>7</b>	0.5
Savannah Sparrow	*	-	-	-	-	-	-	-	-	-	*	-

## Appendix 2 continued.

<b>SHRUB continued</b>	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
American Tree Sparrow	4	-	3	-	3	-	-	-	-	-	10	0
White-crowned Sparrow	4	2.25	2	*	2	1	1	-	2.5	0.5	11.5	3.75
Fox Sparrow	-	-	2	-	1.5	-	2	-	0.5	-	6	0
Redpoll spp.	1	1.5	3	*	2	*	1.5	*	1	*	8.5	1.5
<b>BOREAL WETLAND (n = 15)</b>	9	14	19	7.75	13.5	4.25	1	0	6.5	1.5	49	27.5
Sandhill Crane	*	*	-	-	1	*	-	-	-	-	1	*
Mallard	-	*	-	-	*	*	-	-	-	-	*	*
Green-winged Teal	-	-	1	-	1	-	-	-	-	-	2	0
Blue-winged Teal	-	-	-	-	*	-	-	-	-	-	*	0
American Wigeon	-	-	-	-	*	-	-	-	-	-	*	0
Northern Shoveler	-	-	-	-	*	-	-	-	-	-	*	0
Northern Pintail	-	-	1	-	1	-	-	-	-	-	2	0
Lesser Yellowlegs	-	*	2	*	-	-	-	-	1	0.5	3	0.5
Solitary Sandpiper	-	-	1	*	*	*	-	-	*	-	1	*
Wilson's Snipe	1	0.5	7	1	3	0.5	1	-	*	-	12	2
Mew Gull	-	-	-	-	-	-	-	-	1	-	1	0
Olive-sided Flycatcher	-	-	-	-	-	-	-	-	*	-	*	0
Bohemian Waxwing	*	*	*	0.5	-	-	*	-	4	1	4	1.5
Lincoln's Sparrow	8	13.5	6	6	4.5	3	-	-	0.5	*	19	22.5
Rusty Blackbird	*	-	1	0.25	3	0.75	*	-	-	-	4	1
<b>OTHER (n = 2)</b>	0	0	0	0	*	0	0	0	0	*	0	0
Red-necked Phalarope	-	-	-	-	*	-	-	-	-	-	*	0
Common Raven	-	-	*	-	-	-	-	-	-	*	*	*
Total Number of Breeding Territories /10 ha	19.5	27.25	49.5	22.75	38.5	33	25	23.5	22.5	11	155	117.5
Breeding species richness	7	11	19	14	17	13	15	10	13	8	29	22
Cumulative species richness	12	19	24	20	24	22	22	17	16	12	41	34
Species diversity (H')	1.566	1.704	2.633	2.209	2.590	2.267	2.469	2.129	2.307	1.583	3.006	2.544
<b>Bray-Curtis index of dissimilarity (BC)</b>	<b>0.433</b>		<b>0.730</b>		<b>0.734</b>		<b>0.423</b>		<b>0.403</b>		<b>0.543</b>	

†Although only 4.29 ha were sampled on the White Spruce plot in 2011, the number of breeding territories was adjusted for comparison with other plots.

**Appendix 3** Frequency of occurrence of plant species documented on 10-ha habitat plots in 1975 and 2011, at Creamer's Refuge, Fairbanks, Alaska. Frequency of occurrence was defined as the proportion of circular ( $r = 1.13$  m) plots per habitat that contained a species (based on 49/ha/plot), and was used in calculation of plant species diversity. \* denotes species that were growing on plot but not recorded by sampling.

	Low Shrub		Tall Shrub		Birch		White Spruce†		Black Spruce		Overall	
<b>TREES:</b>	1975	2011	1975	2011	1975	2011	1975	2011	1975	2011	1975	2011
<i>Betula neoalaskana</i>	41	47	8	18	78	51	22	24	-	-	29.8	28.0
<i>Larix laricina</i>	*	*	*	-	-	-	-	-	67	76	13.4	15.1
<i>Picea glauca</i>	*	4	*	24	*	2	43	38	2	4	9.0	14.6
<i>Picea mariana</i>	12	29	29	22	*	-	31	24	71	94	28.6	33.7
<i>Populus balsamifera</i>	-	*	-	-	-	-	-	-	-	-	-	*
<i>Populus tremuloides</i>	10	-	-	-	11	-	-	-	-	-	4.2	-
<b>SHRUBS:</b>												
<i>Alnus viridis</i> ssp. <i>crispa</i>	-	2	-	-	-	-	-	-	-	-	-	0.4
<i>Alnus incana</i> ssp. <i>tenuifolia</i>	8	45	80	69	*	-	22	14	4	8	22.8	27.3
<i>Andromeda polifolia</i>	-	*	-	-	-	-	-	-	55	61	11.0	12.2
<i>Arctostaphylos rubra</i>	-	-	23	12	-	-	18	14	55	31	19.2	11.4
<i>Arctostaphylos uva-ursi</i>	-	-	-	-	4	-	6	-	-	-	2.0	-
<i>Betula glandulosa</i>	63	43	49	51	11	*	2	-	53	39	35.6	26.5
<i>Betula glandulosa</i> X <i>papyrifera</i>	4	4	45	8	-	-	-	-	4	2	10.6	2.9
<i>Betula nana</i>	86	67	4	8	-	-	-	-	14	29	20.8	20.8
<i>Chamaedaphne calyculata</i>	49	71	76	86	2	2	-	-	18	39	29.0	39.6
<i>Dasiphora fruticosa</i> ssp. <i>floribunda</i>	-	-	-	*	-	-	2	*	-	-	0.4	*
<i>Empetrum nigrum</i>	-	-	-	2	-	-	4	*	10	22	2.8	4.9
<i>Ledum palustre</i>	69	88	63	76	28	18	57	52	63	98	56.0	66.4
<i>Linnaea borealis</i>	-	-	-	-	-	-	8	19	-	-	1.6	3.8
<i>Myrica gale</i>	-	-	-	*	-	-	-	-	-	2	-	0.4
<i>Prunus padus</i>	-	-	*	10	-	14	-	-	-	-	*	4.9
<i>Ribes hudsonianum</i>	-	-	10	33	2	14	12	14	-	2	4.8	12.7
<i>Ribes triste</i>	-	-	6	4	2	-	-	-	-	-	1.6	0.8
<i>Rosa acicularis</i>	-	-	20	27	7	39	53	76	2	2	16.4	28.7
<i>Rubus idaeus</i>	-	-	-	4	-	2	-	-	-	-	-	1.2
<i>Salix alaxensis</i>	-	-	-	-	-	-	2	-	-	-	0.4	-
<i>Salix arbusculoides</i>	43	8	67	22	63	4	41	10	2	2	43.2	9.3
<i>Salix bebbiana</i>	6	49	45	29	87	57	35	43	-	16	34.6	38.8
<i>Salix fuscescens</i>	14	20	-	*	-	-	-	-	4	-	3.6	4.1
<i>Salix glauca</i>	-	2	-	2	-	-	10	-	-	-	2.0	0.8
<i>Salix myrtillofolia</i>	14	2	2	-	2	2	-	5	4	6	4.4	3.0
<i>Salix niphoclada</i>	10	-	-	-	-	-	-	-	22	2	6.4	0.4
<i>Salix pseudomonticola</i>	-	-	-	4	4	6	6	-	-	-	1.2	2.0
<i>Salix pseudomyrsinites</i>	-	-	12	-	-	-	2	-	-	4	2.8	0.8
<i>Salix pulchra</i>	82	92	49	61	13	8	35	24	14	16	38.6	40.3
<i>Shepherdia canadensis</i>	-	-	-	-	-	-	18	14	-	4	3.6	3.7
<i>Spirea stevenii</i>	-	-	-	-	2	-	-	-	-	-	0.4	-
<i>Vaccinium oxycoccos</i>	12	6	6	-	-	-	-	-	73	86	18.2	18.4
<i>Vaccinium uliginosum</i>	76	78	55	57	28	10	88	57	63	78	62.0	55.9
<i>Vaccinium vitis-idaea</i>	78	84	22	33	35	31	67	81	69	84	54.2	62.3
<i>Viburnum edule</i>	-	-	-	-	-	2	-	5	-	-	-	1.4

## Appendix 3 continued.

FORBS:	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
<i>Achillea</i> spp.	2	-	-	-	11	2	4	-	-	-	3.4	0.4
<i>Aconitum delphiniifolium</i>	-	-	2	-	-	-	20	24	-	-	4.4	4.8
<i>Amerorchis rotundifolia</i>	-	-	-	-	-	-	-	-	*	-	*	-
<i>Anemone richardsonii</i>	*	-	-	-	-	-	5	-	-	-	*	1.0
<i>Astragalus</i> spp.	-	-	-	-	-	-	2	-	-	-	0.4	-
<i>Bidens cernua</i>	-	-	-	-	-	2	-	-	-	-	-	0.4
<i>Calla palustris</i>	-	-	4	-	-	4	-	-	-	-	0.8	0.8
<i>Caltha palustris</i>	-	-	14	12	-	-	-	-	-	-	2.8	2.4
<i>Cardamine pratensis</i>	-	-	4	-	4	-	-	-	-	-	1.6	-
<i>Chamerion angustifolium</i>	16	18	18	35	57	86	51	38	4	4	29.2	36.2
<i>Chrysosplenium</i> spp.	-	-	4	-	-	-	2	-	-	-	1.2	-
<i>Cicuta virosa</i>	-	-	-	8	-	2	-	-	-	-	-	2.0
<i>Comarum palustris</i>	39	33	41	51	22	27	2	-	8	8	22.4	23.7
<i>Corallorrhiza trifida</i>	-	-	2	-	-	-	-	-	*	-	0.4	-
<i>Cornus canadensis</i>	-	-	-	-	-	4	39	76	-	-	7.8	16.1
<i>Draba</i> spp.	-	-	-	-	2	-	-	-	-	-	0.4	-
<i>Drosera rotundifolia</i>	-	-	-	-	-	-	-	-	35	61	7.0	12.2
<i>Epilobium palustre</i>	6	2	2	16	24	-	2	5	2	2	7.2	5.0
<i>Erigeron</i> spp.	-	-	-	-	7	-	-	-	-	-	1.4	-
<i>Galium trifidum</i>	-	-	22	22	17	24	4	10	-	-	8.6	11.3
<i>Gentianella amarella</i>	-	-	-	2	-	-	-	-	-	-	-	0.4
<i>Geocaulon lividum</i>	-	-	2	-	-	-	4	19	8	31	2.8	9.9
<i>Geum macrophyllum</i>	-	-	-	-	-	4	-	5	-	-	-	1.8
<i>Hippuris vulgaris</i>	-	-	-	-	4	-	-	-	-	-	0.8	-
<i>Iris setosa</i>	-	*	-	4	-	2	-	-	12	8	2.4	2.9
<i>Malaxis paludosa</i>	-	-	-	-	-	-	-	-	-	2	-	0.4
<i>Mertensia paniculata</i>	-	-	10	10	-	-	51	48	-	2	12.2	12.0
<i>Moehringia lateriflora</i>	-	-	*	6	-	82	-	10	*	-	*	19.5
<i>Moneses uniflora</i>	-	-	-	-	-	-	8	14	-	-	1.6	2.9
<i>Parnassia palustris</i>	-	-	-	2	-	-	20	10	-	2	4.0	2.3
<i>Pedicularis labradorica</i>	-	-	-	-	-	-	-	-	10	29	2.0	5.7
<i>Petasites frigidus</i>	-	8	6	4	9	12	41	48	20	18	15.2	18.1
<i>Pinguicula villosa</i>	-	-	-	-	-	-	-	-	2	-	0.4	-
<i>Platanthera obtusata</i>	-	-	-	2	-	-	2	5	6	10	1.6	3.4
<i>Polemonium acutiflorum</i>	-	-	-	-	-	-	6	5	-	-	1.2	1.0
<i>Pyrola</i> spp.	8	4	14	18	17	20	53	48	6	12	19.6	20.5
<i>Ranunculus</i> spp.	20	10	51	37	2	4	12	5	39	20	24.8	15.2
<i>Rorippa islandica</i>	-	-	-	2	-	-	-	-	-	-	-	0.4
<i>Rubus arcticus</i>	2	2	76	76	54	55	53	38	6	-	38.2	34.1
<i>Rubus chamaemorus</i>	96	100	31	35	50	12	6	5	88	98	54.2	49.9
<i>Rumex arcticus</i>	-	-	14	2	-	2	-	-	6	-	4.0	0.8
<i>Saussurea angustifolia</i>	-	-	-	2	-	-	-	-	-	-	-	0.4
<i>Scutellaria galericulata</i>	-	-	-	2	-	-	-	-	-	-	-	0.4
<i>Senecio lugens</i>	-	-	-	-	-	-	*	-	-	-	*	-
<i>Spiranthes romanzoffiana</i>	-	-	-	-	-	-	-	-	-	2	-	0.4
<i>Stellaria</i> spp.	6	-	51	16	59	12	29	-	10	-	31.0	5.7
<i>Taraxacum officinale</i>	-	-	2	-	-	2	-	-	-	-	0.4	0.4
<i>Thalictrum sparsiflorum</i>	-	-	2	10	-	-	6	5	-	-	1.6	3.0
<i>Tofieldia pusilla</i>	-	-	-	-	-	-	-	-	6	2	1.2	0.4
<i>Trientalis europaea</i> ssp. <i>arctica</i>	*	-	49	33	-	20	10	38	4	-	12.6	18.2
<i>Valeriana capitata</i>	-	2	18	20	-	-	10	24	4	4	6.4	10.1



### Appendix 3 continued.

FORBS cont.	Low Shrub		Tall Shrub		Birch		White Spruce		Black Spruce		Overall	
<i>Vicia cracca</i>	-	-	-	-	-	2	-	-	-	-	-	0.4
<i>Viola epipsila</i>	-	-	-	4	-	-	8	14	-	-	1.6	3.7
<b>HORSETAILS:</b>												
<i>Equisetum arvense</i>	-	10	41	43	11	22	24	71	6	22	16.4	33.9
<i>Equisetum fluviatile</i>	-	-	4	2	-	-	-	-	-	-	0.8	0.4
<i>Equisetum palustre</i>	-	-	-	-	4	-	39	-	20	-	12.6	-
<i>Equisetum pratense</i>	-	-	43	10	7	-	76	-	2	-	25.6	2.0
<i>Equisetum scirpoides</i>	-	-	4	4	-	-	47	5	12	10	12.6	3.8
<i>Equisetum silvaticum</i>	-	-	-	-	2	2	35	10	-	-	7.4	2.3
<i>Equisetum variegatum</i>	-	-	4	-	-	-	-	-	-	-	0.8	-

† Because 4.29 ha were sampled on the White Spruce plot in 2011, the frequency of occurrence for this plot was based on 24 circular plots rather than 49.

**Appendix 4(A)** Visual plot changes on Low Shrub plot between 1975 (left) and 2011(right), Creamer's Refuge, Fairbanks, Alaska.

Summer:



Winter:



**Appendix 4(B)** Visual plot changes on Tall Shrub plot between 1975 (left) and 2011(right), Creamer's Refuge, Fairbanks, Alaska.

Summer:



Winter:

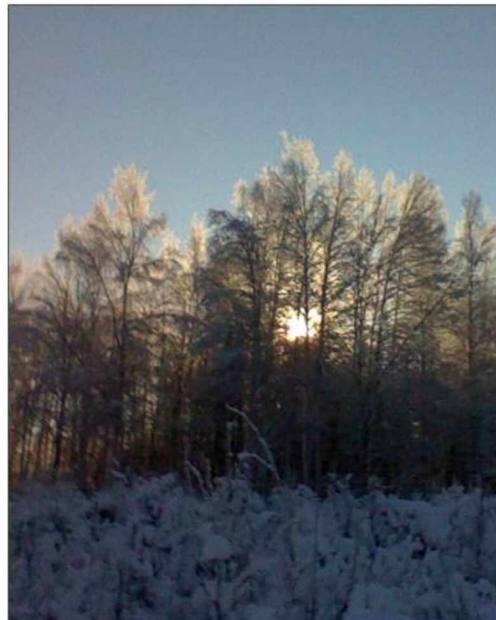


**Appendix 4(C)** Visual plot changes on Birch plot between 1975 (left) and 2011(right), Creamer's Refuge, Fairbanks, Alaska.

Summer:



Winter:



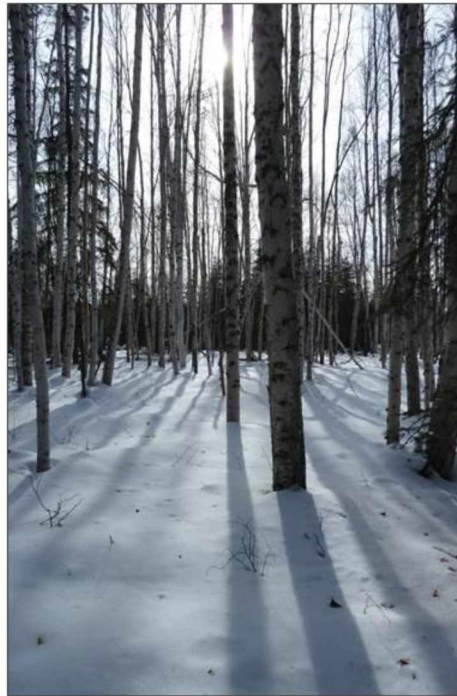


**Appendix 4(D)** Visual plot changes on White Spruce plot between 1975 (left) and 2011(right), Creamer's Refuge, Fairbanks, Alaska.

Summer:



Winter:



**Appendix 4(E)** Visual plot changes on Black Spruce plot between 1975 (left) and 2011(right), Creamer's Refuge, Fairbanks, Alaska.

Summer:

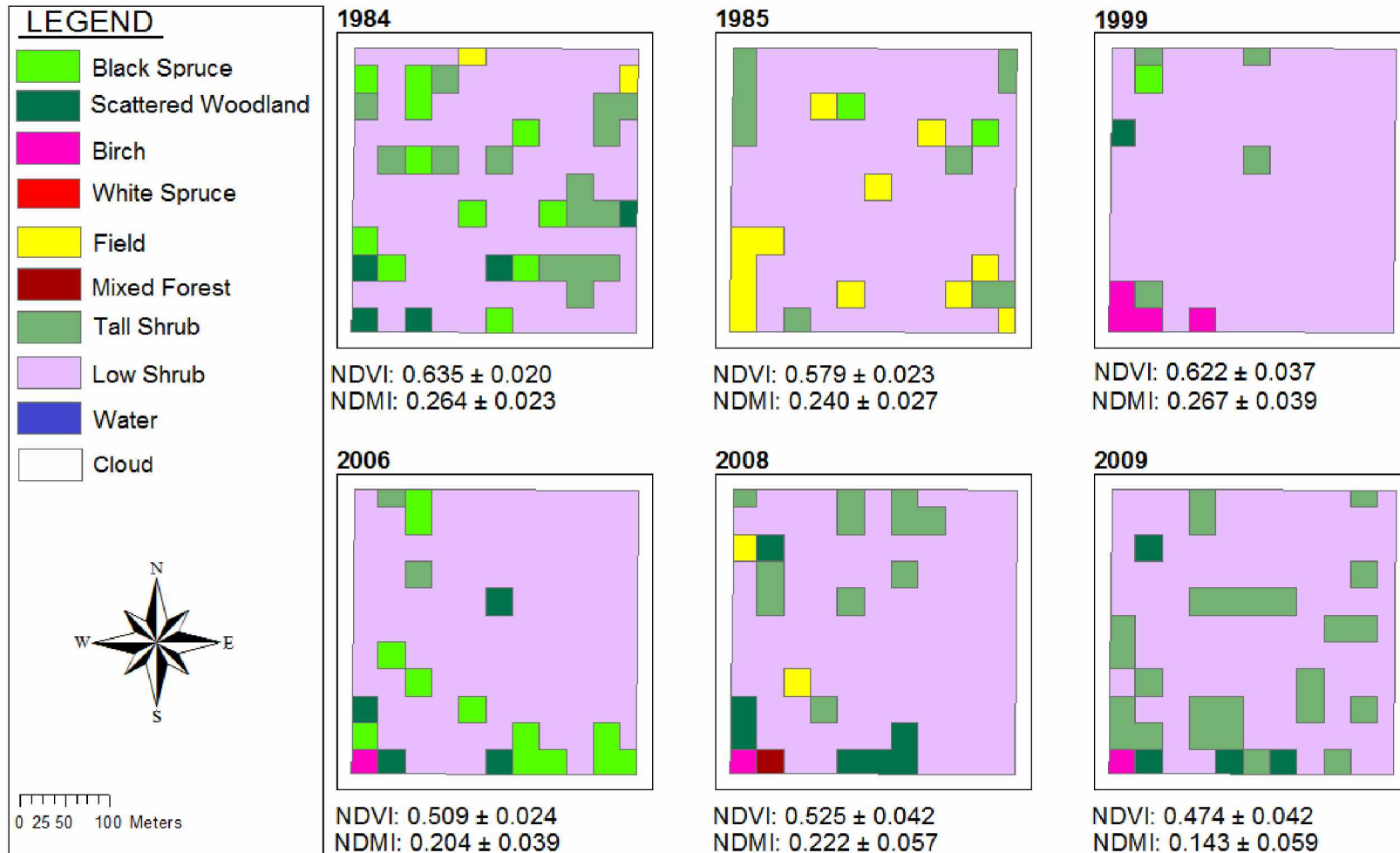


Winter:

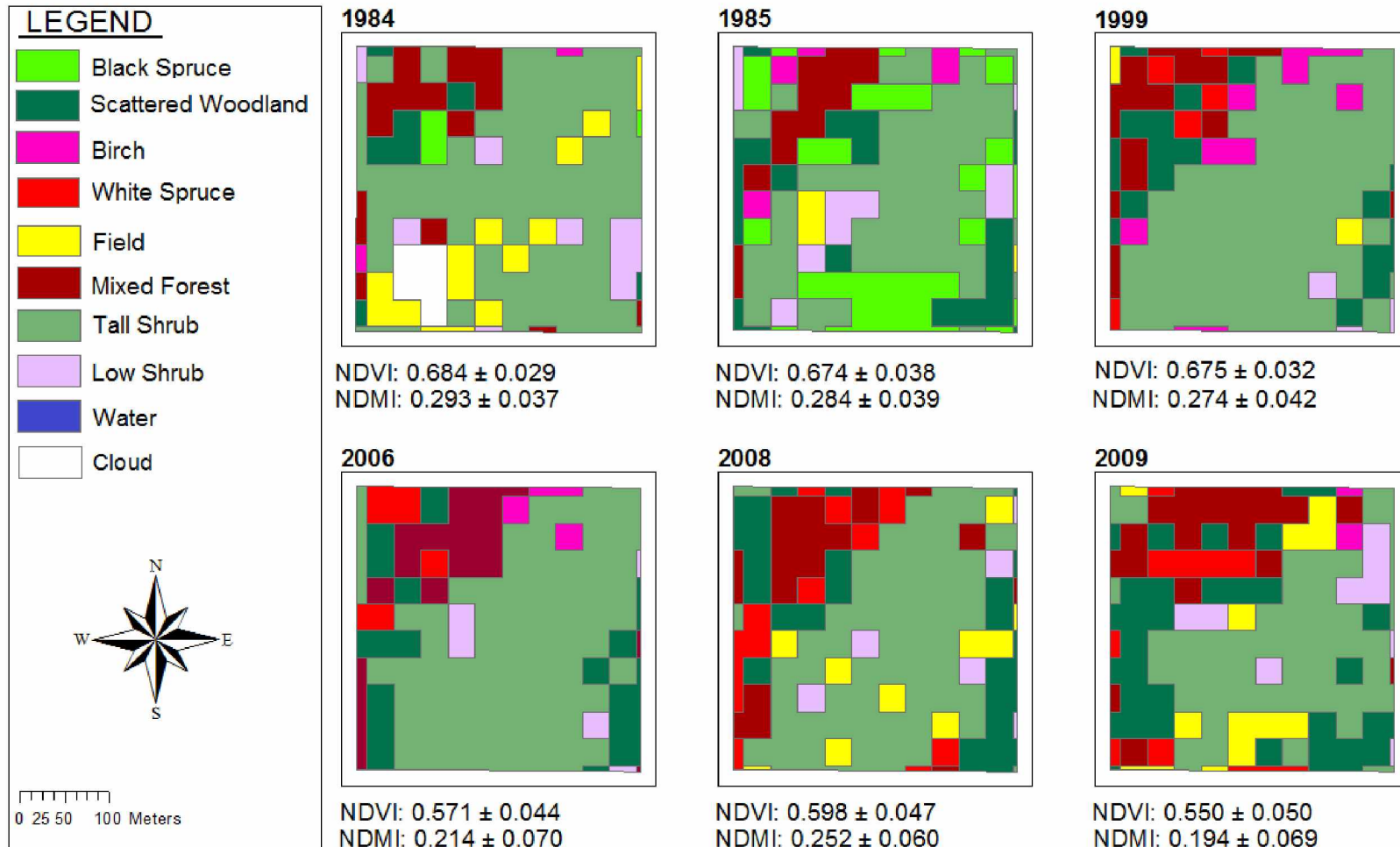




**Appendix 5(A)** Habitat classification maps of Low Shrub plot (10 ha), Creamer's Refuge, Fairbanks, Alaska, generated using Landsat data between 1984 and 2009 (n = 6 years). NDVI and NDMI values are the mean of all pixels where the majority lay within plot boundaries (n = 107)  $\pm$  SD.

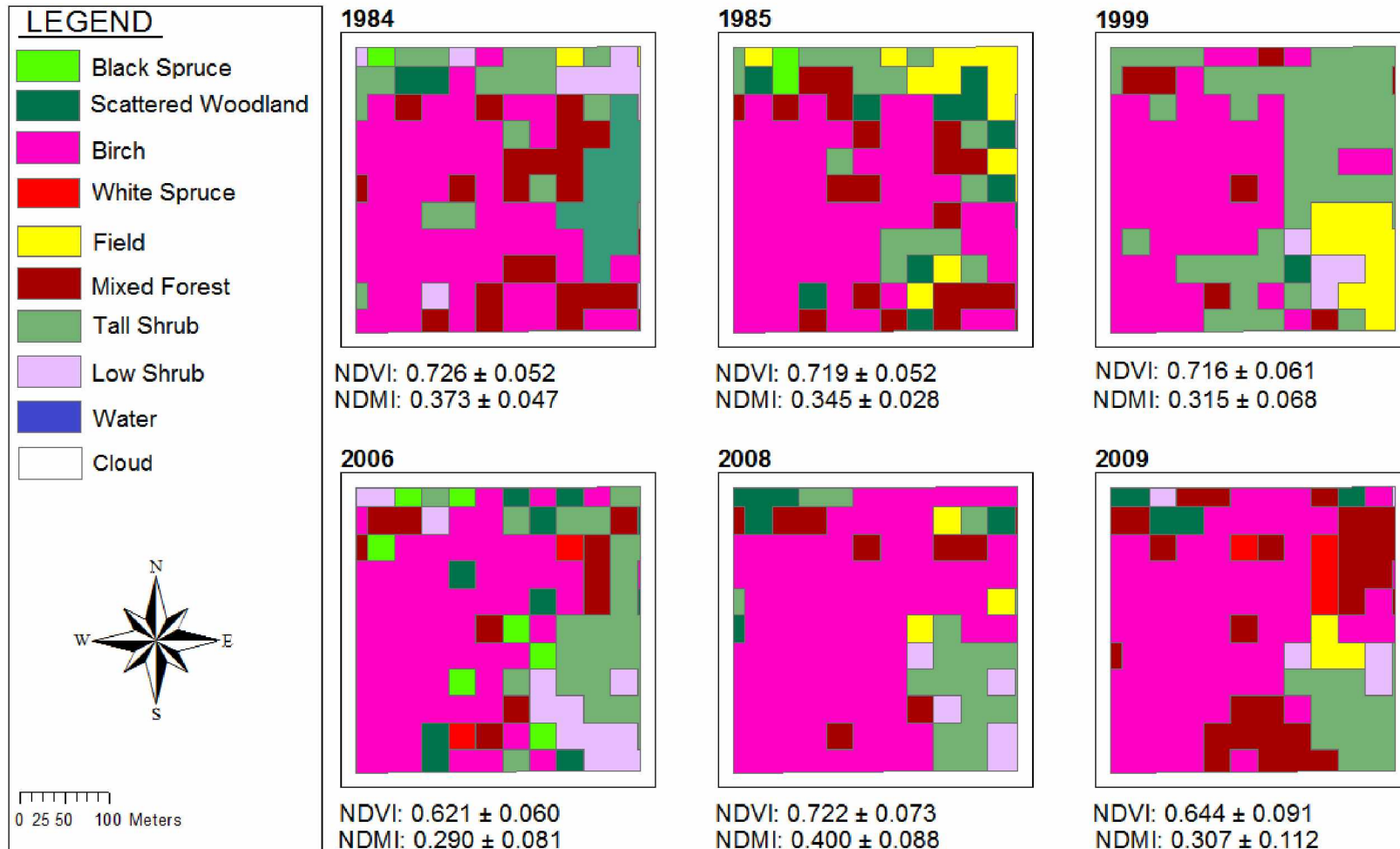


**Appendix 5(B)** Habitat classification maps of Tall Shrub plot (10 ha), Creamer's Refuge, Fairbanks, Alaska, generated using Landsat data between 1984 and 2009 (n = 6 years). NDVI and NDMI values are the mean of all pixels where the majority lay within plot boundaries (n = 110)  $\pm$  SD.

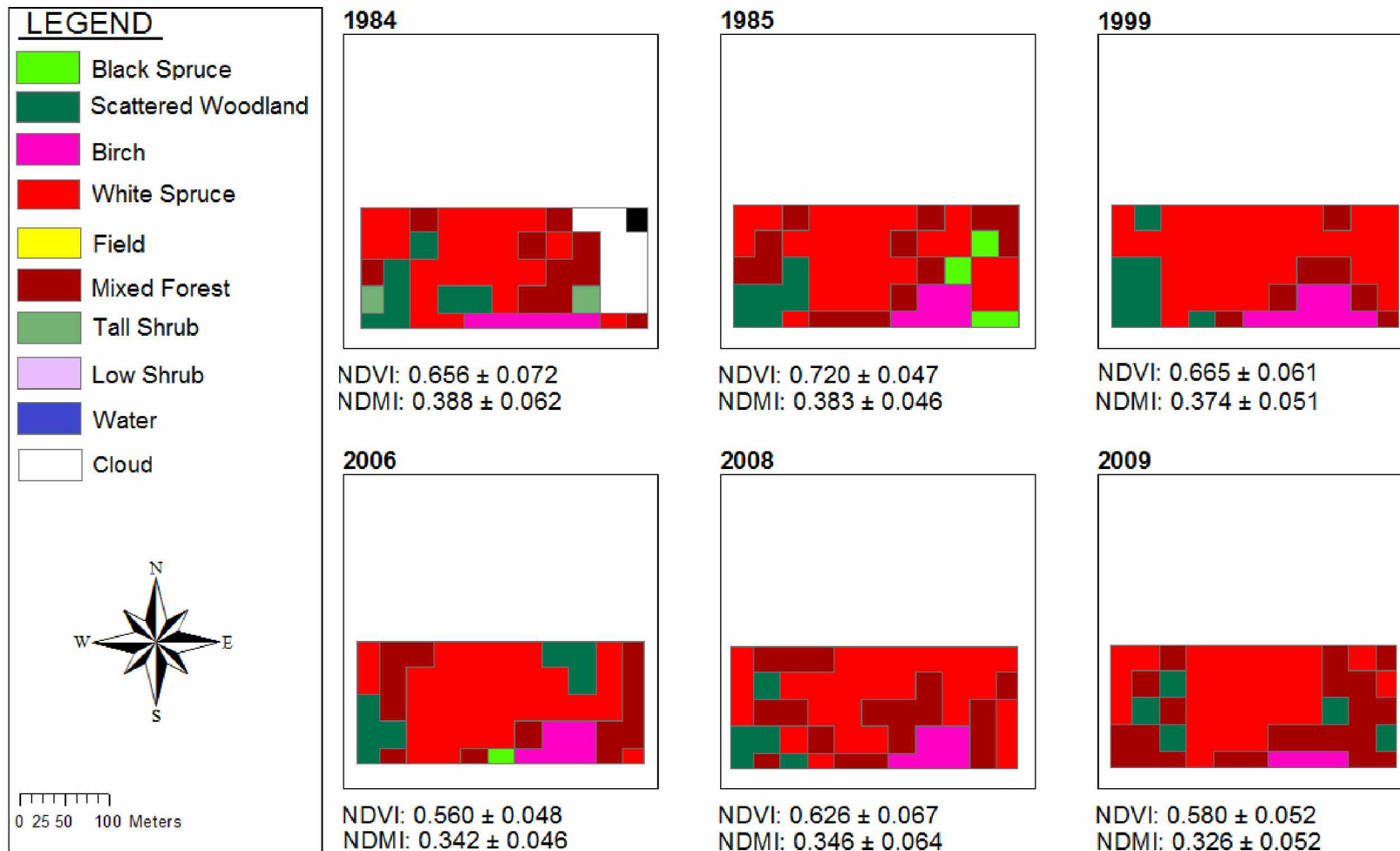




**Appendix 5(C)** Habitat classification maps of Birch plot (10 ha), Creamer's Refuge, Fairbanks, Alaska, generated using Landsat data between 1984 and 2009 (n = 6 years). NDVI and NDMI values are the mean of all pixels where the majority lay within plot boundaries (n = 100)  $\pm$  SD.



**Appendix 5(D)** Habitat classification maps of White Spruce plot (4.3 ha), Creamer's Refuge, Fairbanks, Alaska, generated using Landsat data between 1984 and 2009 (n = 6 years). NDVI and NDMI values are the mean of all pixels where the majority lay within plot boundaries (n = 50)  $\pm$  SD. Only a portion of the plot sampled in 1975 was revisited in 2011 due to property ownership issues.



**Appendix 5(E)** Habitat classification maps of Black Spruce plot (10 ha), Creamer's Refuge, Fairbanks, Alaska, generated using Landsat data between 1984 and 2009 (n = 6 years). NDVI and NDMI values are the mean of all pixels where the majority lay within plot boundaries (n = 100)  $\pm$  SD.

