WINTER RANGE STUDIES OF THE WESTERN ARCTIC CARIBOU HERD, NORTHWEST ALASKA

A

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By

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Abstract

Climate change is likely to bring a myriad of interrelated changes to the Arctic. One change is warmer and drier conditions that could increase the prevalence of wildfire in northwest Alaska. Wildfires destroy terricolous lichens that Western Arctic Herd caribou (*Rangifer tarandus*) rely on during winter; taking decades to recover. My goals were to assess the recent (1950-2007) fire regime within the herd's range, identify characteristics of habitat selected by overwintering caribou, and determine the potential impacts of climate change on the fire regime and caribou winter range. I used a combination of existing data and information collected at vegetation plots to conduct these analyses. I found that wildfires in the tundra were relatively common from 1950-2007, covering approximately 10 % of northwest Alaska. Tundra was > 4.5 times more likely to re-burn than boreal forest. This novel, yet intuitive finding could have serious implications if fire starts to become more common in the Arctic. I found that the average annual area burned more than doubled in years where mean August temperatures exceeded 11.7 ° C (53° F). Caribou use tundra and forested during winter but avoided recently (< 58 years) burned areas in both habitat types likely because they contained < ¼ of the abundance of forage lichen species than unburned habitats. I found that lichen abundance was 3 times greater in the herd's current winter range versus its historic range – supporting the theory that caribou shift ranges to compensate for deteriorating grazing conditions. Stand age was the most consistent correlate with lichen abundance. Dwarf birch (*Betula* spp.) was more abundant in recent burns which may facilitate the intensification of the future fire regime in the region. My modeling efforts revealed that wildfire is likely to become more prevalent, especially on the herd's core winter range, which could have deleterious impacts on caribou winter range and provide quality habitat for moose (*Alces alces*). My results should provide a solid foundation to develop a science-based fire management plan for the Western Arctic Herd.

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The origins of this project can be easily traced back to the intellectual curiosity of ecologists Randi Jandt (Alaska Fire Service) and Lisa Saperstein (US Fish and Wildlife Service). Their initial efforts launched my research into the effects of wildfires on caribou winter range in the boreal forests of eastern Alaska. I hope they gain some satisfaction seeing the work finally come around full circle back to the ecosystem which inspired their efforts: the tundra of northwest Alaska.

My thesis has only been completed thanks to an army of associates, colleagues, co-workers and friends that were willing to provide professional and personal advice, manuscript reviews, and statistical, programming and GIS expertise. They include, but are not limited to; Paul Atkinson, Mike Balshi, Peter Bente, Geoff Carroll, Terry Chapin, Jim Dau, Greg Dudgeon, Paul Duffy, Daniel Fortin, Tim Glaser, Dave Gustine, Tim Hammond, Claudia Ihl, Randi Jandt, Torre Jorgenson, Dave Klein, Jim Lawler, Matt Leonawicz, Tom Liebscher, Trent McDonald, Scott Miller, Randy Meyers, Tricia Miller, Peter Neitlich, Mark Olson, Kate Persons, Dustin Rice, Carl Roland, Scott Rupp, Lisa Saperstein, Josh Schmidt, Brad Shults, Anna Springsteen, Karin Sonnen, Glenn Stout, Dave Swanson, Dave Verbyla and members of the Western Arctic Herd Working Group. I thank you all.

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INTRODUCTION

Barren-ground caribou (*Rangifer tarandus granti*) are an integral part of the ecology and sociology of the circumpolar North (Gordon, 2005; Sutherland, 2005). The Western Arctic Herd (WAH), numbering over 400,000 caribou, inhabits the tundra and boreal forests of northwest Alaska (Figure I.1; Dau, 2007; Joly et al., 2007a; CHAPTER 2). Caribou herds with hundreds of thousands of animals have the potential to significantly alter their environment (Moser et al., 1979; Jefferies et al., 1992), but can be affected by changes in their environment as well. While the effects of wildfire on boreal forest caribou winter range have been a concern for decades (Klein, 1982), impacts to tundra caribou winter range have been largely ignored (but see Saperstein, 1996).

While fires are the primary mechanism for resetting vegetative succession in the boreal forest, they are less frequent and smaller in the tundra biome, especially in Canada (Wein, 1976; Payette, 1992; Higuera et al., 2009). However, two regions within the range of the WAH, the Seward Peninsula and Noatak River valleys (Figure I.1), display greater fire frequency than other tundra-dominated areas (Racine et al., 1985, 2004; Hu et al., 2010; CHAPTER 1). Certain species of terricolous lichens species found in these regions comprise a critical component of the winter diet of migratory caribou and are readily consumed by wildfires (Klein, 1982; Joly et al., 2007b). These lichens can take decades to a century or more to recover to pre-fire levels (Klein, 1982; Jandt et al., 2008). Caribou avoid burned areas in the boreal forest and tundra on a time scale coincident with lichen recovery rates (Joly et al., 2003, 2007a; CHAPTER 3).

Climate change impacts are readily apparent and have already been implicated in the decline of caribou populations across the Arctic (Hinzman et al., 2005; Vors and Boyce, 2009). However, the effects of climate change on caribou populations may be modulated by large-scale climate teleconnections, such as the Pacific Decadal Oscillation (Joly et al., 2011). Higher summer temperatures with longer growing seasons may prove to be beneficial to caribou over the summer months (Haskell and Ballard, 2004; Tews et al., 2007). These same conditions may provide vascular plant species a competitive advantage over caribou forage lichens (see review by Joly et al., 2009) and exacerbate insect harassment (Jefferies et al., 1992; Walsh et al., 1992) – both to the detriment of caribou. Climate change may also negatively impact caribou by increasing snow depths, increasing the frequency of icing events, encouraging further human disturbance and development, reducing habitat quality and quantity, and increasing wildfire on winter ranges (Jefferies et al., 1992; Putkonen and Roe, 2003; Dau, 2005; Johnson et al., 2005; Rupp et al., 2006; Joly et al., 2011). Changes to winter climate may be of greater magnitude and importance to caribou ecology than summer climate (Callaghan et al., 2004; Haskell and Ballard, 2004; Tews et al., 2007). However, increased prevalence of wildfire during summer months may deteriorate winter range by reducing forage quantity and quality during an energetically crucial period for reproductive females (White, 1983; Parker et al., 2005).

Climate change has brought (Jones et al., 2009) and is predicted to bring even warmer and drier summers to the Arctic (Callaghan et al., 2004), which are associated with increased burned area in the boreal forests (Duffy et al., 2005). It is predicted that wildfire will increase in both the boreal forest and tundra biomes under climate warming scenarios (Flannigan et al., 2000; Higuera et al., 2008). More frequent and extensive wildfires may limit the total amount of high-quality habitat that is available for overwintering caribou and nullify the argument that wildfire is not important to winter ecology of caribou because they can simply migrate to higher-quality winter ranges (Rupp et al., 2006). Moreover, this altered wildfire regime within the range of the WAH will likely change fire severity, patterns of inclusions (i.e., islands of unburned habitat within a burn perimeter) and affect species composition and distribution of both flora and fauna (Chapin et al., 1992; McGraw and Fletcher, 1992) – with the potential to further impact caribou. In short, little is known about the impacts of a changing climate and fire regime on the tundra ranges of caribou.

Goals, Objectives, Hypotheses, and Predictions

My overarching goal was to assess the potential impacts of current and plausible future wildfire regimes on the quantity and quality of winter range available to the WAH. In CHAPTER 1, I developed a quantitative baseline for the fire regime of northwest Alaska to serve as a foundation for subsequent chapters. I accomplished three goals; first, I quantified the areal extent of burned area and fire cycle (return interval) for northwest Alaska, including both tundra and boreal forest biomes. Second, I determined whether or not fires are increasing in areal extent and frequency within the range of the WAH. And lastly, I identified meteorological variables that correlated with annual area burned in northwest Alaska. I hypothesized that 1) while the proportion of area burned in the tundra of northwest Alaska would be relatively lower than in the boreal forest, it would be substantially higher than reported in the tundra biome in general; 2) the fire cycle of certain tundra-dominated regions would rival fire cycles of the boreal forest; 3) fire frequency has increased in the region; and 4) warmer and drier summers were correlated with more annual area burned in northwest Alaska.

The goal of CHAPTER 2 was to utilize existing WAH caribou location data to assess habitat selection during winter. The first specific objective of CHAPTER 2 was to document the distribution of caribou during winter while the WAH was at a population peak level. Second, I identified factors that help explain this distribution. The information developed in CHAPTER 2 provided valuable insight into the factors that shape caribou distribution while serving as a critical foundation for predicting potential changes in caribou distribution if the population continued to decline and to model how the suitability of winter range may change for caribou under different climate-change scenarios. I hypothesized that WAH caribou distribution was affected by terrain, habitat and predation pressure as well as snow conditions and disturbance. Further, the distribution of cows and bulls should be different. Cows are often found in larger groups and large patches than bulls. The small group sizes of bulls allow them to utilize smaller foraging patches.

In CHAPTER 3, I utilized vegetation plot surveys to determine caribou winter range quantity and quality in northwest Alaska. First, I identified characteristics of primary caribou winter range by comparing sites used by caribou to random locations. Next, I quantified the differences between burned and unburned habitat and among the herd's current, historic and potential future winter ranges. Finally, I identified landscape-level and site-specific factors that were associated with the abundance of forage lichens. I hypothesized that 1) caribou would seek out and locate areas of high lichen abundance; 2) that unburned habitat would provide greater lichen abundance than recently (< 58 years) burned habitat; and 3) that the herd's current winter range would provide greater lichen abundance than either the historic or potential winter ranges. The findings of CHAPTER 3 highlighted the importance of winter forage for caribou, will inform critical and costly decisions regarding fire management, and assist in the management of caribou as a subsistence and economic resource for communities in northwest Alaska and throughout the Arctic.

Warmer and drier conditions in northwest Alaska will likely lead to increased burning of lichen-rich caribou winter range. Thus, my goal for CHAPTER 4 was to identify the impacts of climate change on the fire regime, and thus caribou winter range, in tundra-dominated landscapes of northwest Alaska. I used computer simulations to project amount and quality of caribou winter range available under two different climate warming scenarios using information developed in the previous chapters and existing literature. Additionally, I projected the change in moose winter habitat because this may have additional indirect impacts on caribou winter range. I hypothesized that 1) the fire regime in northwest Alaska in the next 50-100 years will be more intense (i.e., larger areal extent of burned area) as compared to the current (1950-2007) regime; and 2) that the hypothesized increase in area burned will lead to a decrease in both quality and quantity of caribou winter range and an expansion of high-quality moose habitat as indexed by the amount of deciduous habitat in the 10-30 year age class.

In the Conclusion I outlined the most critical findings of the various chapters and placed them into context with one another and caribou ecology in general. I also provided suggestions for the direction of potential future research regarding the impacts of climate change on caribou winter range. Lastly, I developed a preliminary fire management plan based on the scientific findings of my research.

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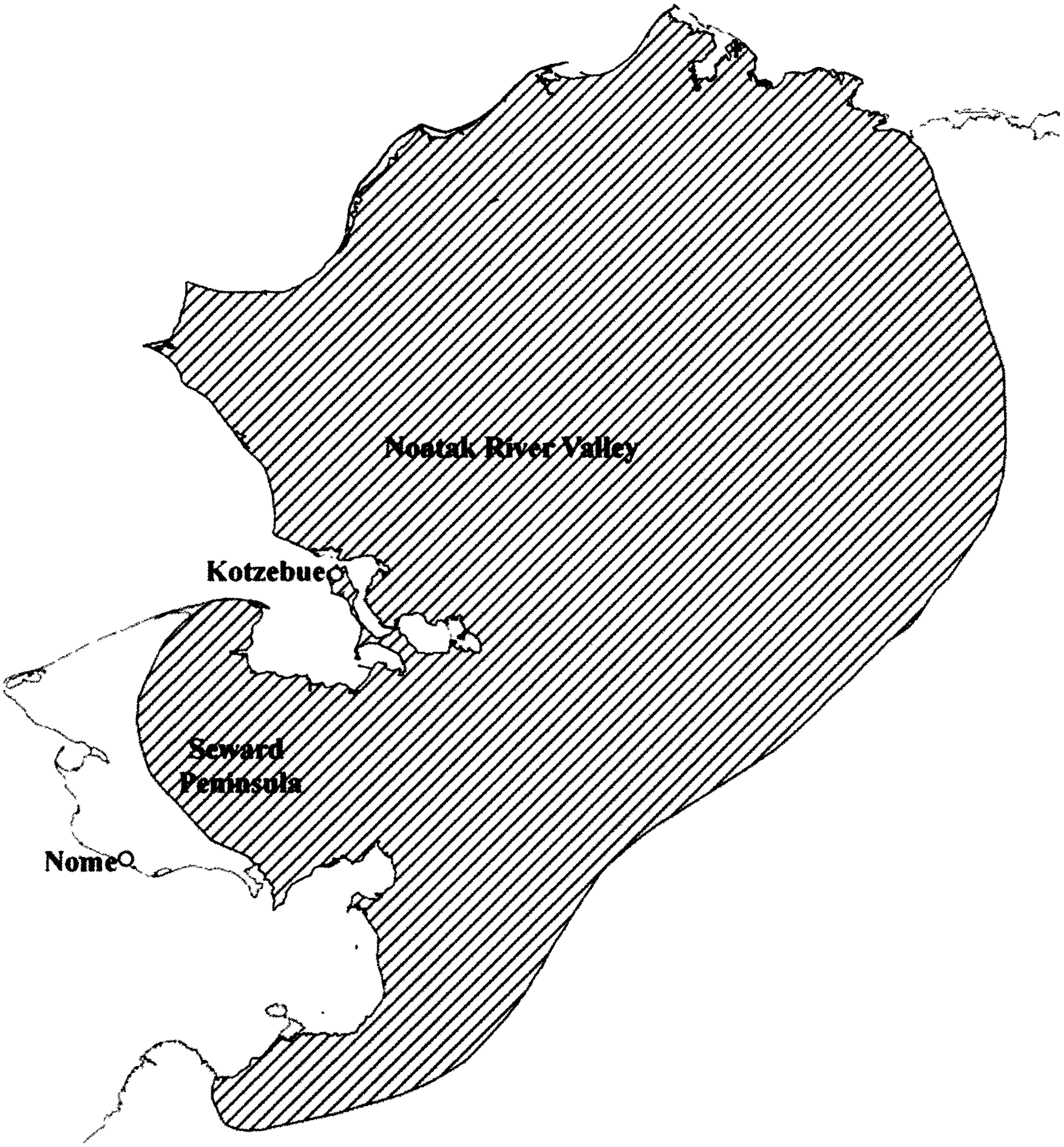

Figure I.1. Range of the Western Arctic Caribou Herd (hatched), northwest Alaska, 2011 (data courtesy of the Alaska Department of Fish and Game).

CHAPTER 1:

Fire in the Range of the Western Arctic Caribou Herd [1]

Abstract

Wildfire is the dominant ecological driver in boreal forest ecosystems. Although much less is known, it also affects tundra ecosystems. Fires effectively consume fruticose lichens, the primary winter forage for caribou, in both boreal and tundra ecosystems. We summarize 1950-2007 fire regime data for northwestern Alaska and sub-regions. We also identified meteorological factors that help explain the variability in fire extent across this landscape. We review information and inferences from recent studies on tundra fire regimes for managing caribou winter range. Climate warming may increase fire size and frequency in this region, which may substantially impact the vegetation, wildlife, and people of this region.

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Introduction

Although much attention has been focused on the role of wildfire in boreal forest ecosystems, its role in tundra ecosystems has largely been overlooked (see *Miller 1985, Racine et al. 1985*). Wildfires effectively consume ground-dwelling lichens that are the primary winter forage for Western Arctic Herd (WAH) caribou. These lichens can take decades to recover to pre-fire levels in northwestern Alaska (*Jandt et al. 2008*). Wildfires potentially limit caribou winter range due to low levels of preferred forage (*Rupp et al. 2006*). The WAH has undergone a population expansion starting in 1976 until reaching a high of 490,000 caribou in 2003 (*Dau 2005*). The herd has expanded and shifted its primary winter range to the Nulato Hills and the Seward Peninsula from its historic winter range in the Buckland Valley and Selawik National Wildlife Refuge *(Figure 1.1).* Both density-dependent (i.e., increasing herd size leading to reduced lichen biomass) and density-independent (i.e., increasing area burned by wildfires leading to less mature habitat) may factor into the changes in winter range distribution exhibited by the WAH.

Temperatures, in Alaska, including northwest Alaska, have been rising and the rate of climate warming is predicted to increase (*Stafford et al. 2000, ACIA 2005*). Summer precipitation has decreased at many locations throughout Alaska, and Barrow has seen declines in annual precipitation as well (*Stafford et al. 2000*). Warmer and drier summers are associated with greater area burned in Alaska (*Duffy et al. 2005*). In the tundra ecosystem, wildfires are also predicted to increase (*Higuera et al. 2008*). Regional temperature and precipitation are correlated with large scale climatic regimes, such as the

Pacific Decadal Oscillation (PDO) (*Hartmann and Wendler 2005*). Our goals were to 1) elucidate the fire regime in northwestern Alaska and in tundra ecosystems, 2) test whether wildfires are increasing in extent and frequency in the range of the WAH and 3) identify meteorological variables that correlate with annual area burned.

Methods

We summarized the data contained in the Alaska Fire Service's geodatabase, which catalogs the extent, number and location of large fires mapped from 1950-2007 (*Figure 1.2*; data at *http://agdc.usgs.gov/data/fire/index.html*). Kasischke et al. (*2002*) performed similar analyses for Interior Alaska, but we calculated them for northwestern Alaska and various subsets including the WAH core winter range, outer range, and potential future winter range - defined as a 30-mile (50-km) wide buffer around the outer range *(Figure 1.1.A)*. We also summarized burn area by conservation unit, ecoregion, tundra ecosystem, boreal forest ecosystem and also the area north of the 68th latitude *(Figures 1.1 and 1.2)*. Tundra ecosystems were differentiated from forested areas using the 33-yard (30-m) National Land Cover Database – Alaska 2001 coverage (http://www.epa.gov/mrlc/nlcd-2001.html).

For all of these areas, we also calculated the percentage of burned area that burned two or more times during the 58-year study period. Similarly, we calculated the fire cycle (i.e., the number of years required to burn over an area) (*Kasischke et al. 2002*) for these areas. We calculated this by dividing 1 by the proportion of area burned and multiplied the dividend by the duration of the study period (58 years). We used linear regression to test whether annual area burned and number of wildfires were increasing over time and for correlation among variables. We used average monthly temperature and total monthly precipitation data (Western Region Climate Center, http://www.wrcc.dri.edu/summary/climsmal.html) from 1950-2005 to develop climatic models to explain the variance in the amount of area that burned annually. Sets of station averages were compared to find the grouping that provided the most predictive power. We also transformed these data, by exponentiation, and included measures of the strength (average of January and February as per *Duffy et al. 2005*) of the PDO (University Of Washington, http://jisao.washington.edu/pdo/PDO.latest) to develop alternative models. The best models were chosen based on predictive ability and parsimoniousness using Akaike's Information Criterion (AIC_c).

Results

More than 10.5 million acres (4.3 million hectares) burned in northwestern Alaska between 1950 and 2007, covering approximately 10.9% of the region. Of these burned acres, 7.9% have re-burned during this time period. We determined the fire cycle for the region to be 535 years. The range of the WAH covers this entire region, excluding the western most extremes of the Seward Peninsula. The percent area burned and re-burned, and fire cycle for other portions of the WAH's range, conservations units, and ecoregions are shown in *Table 1.1*. Ecoregion fire cycles were similar to those reported by Kasischke et al. (*2002*). North of 68° latitude, only 1.1% of the area had burned with no re-burning at all. More than half of the burned area was attributed to the 2007 Anaktuvuk River fire. In the boreal forest ecosystem, 24.1% had burned but only 1.5% of those burned areas had re-burned during the past 58 years. The fire cycle for forested areas was 240 years. Burns only covered 9.2% of the tundra ecosystem, but 7.0% of that area had re-burned in the past 58 years. The fire cycle for tundra areas was 630 years.

The average of the seven weather stations (Bettles, Big Delta, Fairbanks, McGrath, Nome, Northway and Tanana) that Duffy et al. (*2005*) used to model area burned in Interior Alaska provided the most explanatory power versus various subsets or the inclusion of the Barrow and Kotzebue station data. Average June and July temperatures have increased over time in northwestern Alaska (*Figure 1.3;* $F_{1,55} = 18.67$, $P = 0.001$, $F_{1,55} = 10.99$, $P = 0.002$*)*, though average August temperature and precipitation from June through August did not show a significant relationship. The average August temperature for Barrow, Kotzebue and Nome was 46.7°F (8.2°C) +/- 2.6°F (1.4°C).

There was no significant relationship with time and area burned (i.e., there was no evidence that burned area is increasing over time; $F_{1,57} = 0.61$, $P = 0.439$). However, when we omitted years with more than 200,000 acres (81,000 hectares) burned, the amount of burned area did increase over time ($F_{1,42} = 9.95$, $P = 0.003$). There were 15 years where the annual burn area was greater than 200,000 acres (81,000 hectares) and they were clustered into 4 groups – each group contained three to five years of high burn area, spanned four to nine years and were temporally separated by an average of 16.3 years (sd = 0.72). All but 1 (i.e., 14 of 15) of these high fire years were associated with average August temperatures exceeding 53°F (11.7°C). The exception occurred in 1969, which had the lowest June precipitation on record – well less than half the normal for the month. Although burned area was more than double in years with average August temperatures > 53°F (11.7°C), 226,000 versus 96,000 acres (91,000 versus 39,000 hectares), the difference was not statistically significant ($P > 0.1$).

The number of wildfires in northwestern Alaska and in the tundra ecosystem significantly increased from 1950 to 2007 ($F_{1,57} = 11.50$, $P = 0.001$, $F_{1,57} = 11.40$, $P = 0.001$, respectively). These trends disappeared when the analysis was limited to 1988-2007 ($F_{1,18} = 0.73$, $P = 0.404$, $F_{1,18} = 0.72$, $P = 0.406$, respectively). Dry weather in August was significantly associated with high August temperatures ($F_{1,55} = 7.42$, $P = 0.009$). A 6-factor (June-September precipitation and July-August temperature) model explained the most variation, approximately 31%, in annual burn area in northwestern Alaska. Explanatory power was increased when non-linear factors were added; a 5-factor model (June and August precipitation, exponential of June precipitation, exponential of August temperature and PDO) explained 55% of the variance in annual burned area. This model plus the exponential of June temperature explained 67% of the variance of average annual burned area within tundra ecosystems. The single factor of the exponential of August temperature explained 28% of the variance in burned area for northwestern Alaska and 47% for tundra fires in this region. For more on models, see *Table 1.2*.

Discussion and Conclusions

We found that wildfire is a common occurrence in northwestern Alaska, in the range of the WAH and in tundra ecosystems. Burn acreage tended to decrease with latitude and longitude as, historically, fires have been rare events north of the Brooks Range and in maritime climates. Nearly 20% of the WAH core winter range has burned during the past 58 years. With current high population densities and declining lichen cover, the herd may seek out additional winter ranges (*Joly et al. 2007b*). We found that the WAH's outer range has burned even more extensively than its core winter range. Potential future winter range, further to the east, was one of the most extensively burned areas in the region and also appeared to have one of the highest incidences of re-burning - likely because it is in the warmer and drier continental Interior climate zone. We believe this level of burning may prove to be an impediment for the herd to expand its winter range possibilities as extensive, mature lichen mats are unlikely to be found in these areas.

The extent of burned area within Selawik National Wildlife Refuge (28%) came as a surprise, as well as the fact that this area had the highest percentage of re-burned area of any sub-region within northwestern Alaska. These facts may help explain why the herd has largely abandoned its historic winter range in the refuge, though density-dependent factors are likely to have also played an important role. The Seward Peninsula ecoregion had the second highest re-burn percentage, but is still utilized by the herd (*Joly et al. 2007a*). One possible reason for the continued use of this region as winter range is that it contains > 4 times more area that has not burned in the past 58 years than the Selawik. Re-burn estimates need to be cautiously interpreted, however, as fire perimeters in the AFS database do not account for unburned inclusions and in earlier years were often based on rough maps produced by firefighting crews.

As we expected, wildfire affected a greater percentage (~ 25%) of forested areas than tundra areas (< 10%) in northwestern Alaska during the past 58 years. We found that burned tundra was 4.5 times more likely to re-burn than burned forest during our 58-year study period. This finding is intuitive because grasses and sedges that dominate tundra ecosystems recover very quickly (*Jandt et al. 2008*), and produce an important surface fuel (dead leaf litter) to carry new fires. Conversely, surface fuel loads (dominated by feather mosses) in the boreal forest can take decades to return (*Kasischke and Stocks 2000, Camp et al. 2007*). The fire cycle for tundra areas was more than 2.5 times longer than for forested areas. We did not find any examples of forested areas being re-burned more than once, while we found 11 cases in the tundra where a patch had re-burned more than once and one location on the central Seward Peninsula was mapped as burned in 1971, 1990, 1997 and 2002.

Using the large fire database, we were unable to detect a trend of increased annual burn area over time. This may be because climate warming is not yet strong enough to impact northwestern Alaska's fire regime or is intertwined with other factors that may suppress wildfires. However, when we omitted large fire seasons, we found a strong increasing trend *(Figure 1.4)*. This may be explained in at least two ways. First, changes in the accuracy of fire maps and in fire suppression capability and management over the period of record may affect apparent burn acreage trends. Alternatively, climate warming may indeed be increasing annual area burned, but some other factor may induce pulses of large fire seasons that mask this overall trend when included in the regression analysis. The number of wild and tundra fires in northwestern Alaska appear to have significantly increased during the past 58 years. This, however, may also be an artifact of the fact that fires less than 1,000 acres (405 hectares) were not regularly mapped prior to 1988. We did not find evidence that the number of wild and tundra fires have increased since 1988 – a time period when all of these factors should be equivalent in the database. Increases in both the area burned and number of fires in the boreal forest have been identified (*Kasischke and Turetsky 2006*).

Our addition of non-linear and Pacific Decadal Oscillation factors greatly improved our model's ability to predict annual burned area. The model was even stronger at predicting the amount of burned tundra. The effects of climate change, potentially warmer and drier summer weather, may have non-linear effects on the fire regime of northwestern Alaska. For northwestern Alaska and tundra, the exponential of August temperature had the greatest explanatory power. For Interior boreal forests, June temperature was the single most important factor explaining variance in burned area (*Duffy et al. 2005*). Part of this difference may be explained by phenology differences between the ecosystems – in other words, summer simply comes later to northwestern Alaska than it does to the Interior, therefore temperatures later in the year are more important in determining annual burn area. Additionally, August is on average the coolest of the summer months but has the greatest variability. Warm temperatures in August were correlated with dry weather and thus it is not surprising that they are associated with increased annual burned area in northwestern Alaska (*Miller 1985*).

Management Implications

The management of wildfires is a contentious issue, not least of all because of its implications for caribou winter range. Our findings are based on the large fire database maintained by the Alaska Fire Service and thus should be viewed carefully. We believe our preliminary findings provide a starting point for understanding the importance of wildfire in northwestern Alaska and tundra ecosystems in general. While fires are less common in tundra ecosystems than in boreal forests, tundra ecosystems are capable of burning much more frequently. Understanding the fire regime of this region and its impacts on the WAH will be critical information utilized in the development of a fire management plan for the winter range of the herd.

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Table 1.1. Percent area burned and re-burned from 1950-2007 and fire cycle for various regions within northwest Alaska.

Region	**% Area Burned**	**% Area Re-burned**	**Fire Cycle (years)**
Caribou			
Core winter range	19.6	8.7	296
Outer range	34.9	6.8	166
Potential winter range	42.8	13.2	136
Conservation Units			
Cape Krusenstern (NPS)	0.1	0.0	53349
Gates of the Arctic (NPS)	2.7	5.0	2173
Noatak (NPS)	4.7	5.2	1237
Bering Land Bridge (NPS)	4.9	0.0	1188
Kobuk Valley (NPS)	6.9	3.5	844
Nulato Hills (BLM)	19.9	1.3	292
Selawik (FWS)	28.0	15.7	207
Koyukuk (FWS)	45.1	11.0	129
Innoko (FWS)	57.6	12.7	101

Table 1.1 (continued). Percent area burned and re-burned from 1950-2007 and fire cycle for various regions within northwest Alaska.

Region	**% Area Burned**	**% Area Re-burned**	**Fire Cycle (years)**
Ecoregions			
Coastal Plain	0.0	0.0	n/a
Brooks Range	1.0	3.3	5917
Foothills	1.7	0.0	3467
Kotzebue Lowlands	6.8	2.9	859
Seward Peninsula	13.9	15.1	418
Nulato Hills	20.5	2.0	283
Kobuk Ridge and Valley	30.0	8.6	193
Yukon Lowlands	42.2	10.0	137

Table 1.2. Models explaining the annual amount of area burned for (A) all of northwest Alaska and (B) just for tundra (non-forested) ecosystems within northwest Alaska, 1950-2005. The term "exp" means the exponential of that variable was used. The term "PDO" is the average of the January and February values of the strength of the Pacific Decadal Oscillation.

A. All of northwest Alaska

Model Variables	Δ AIC$_c$	Adj. R^2
June and August precipitation, exp August temperature, exp June precipitation, PDO	0.00	0.547
June precipitation, exp August temperature, exp June precipitation, PDO	0.40	0.532
exp August temperature	20.12	0.284

B. Tundra

Model Variables	Δ AIC$_c$	Adj. R^2
June and August precipitation, exp June and August temperature, exp June precipitation, PDO	0.00	0.667
June precipitation, exp June and August temperature, exp June precipitation, PDO	0.16	0.656
June precipitation, exp August temperature, exp June precipitation, PDO	0.47	0.645
exp August temperature	19.10	0.467

Figure 1.1.A.

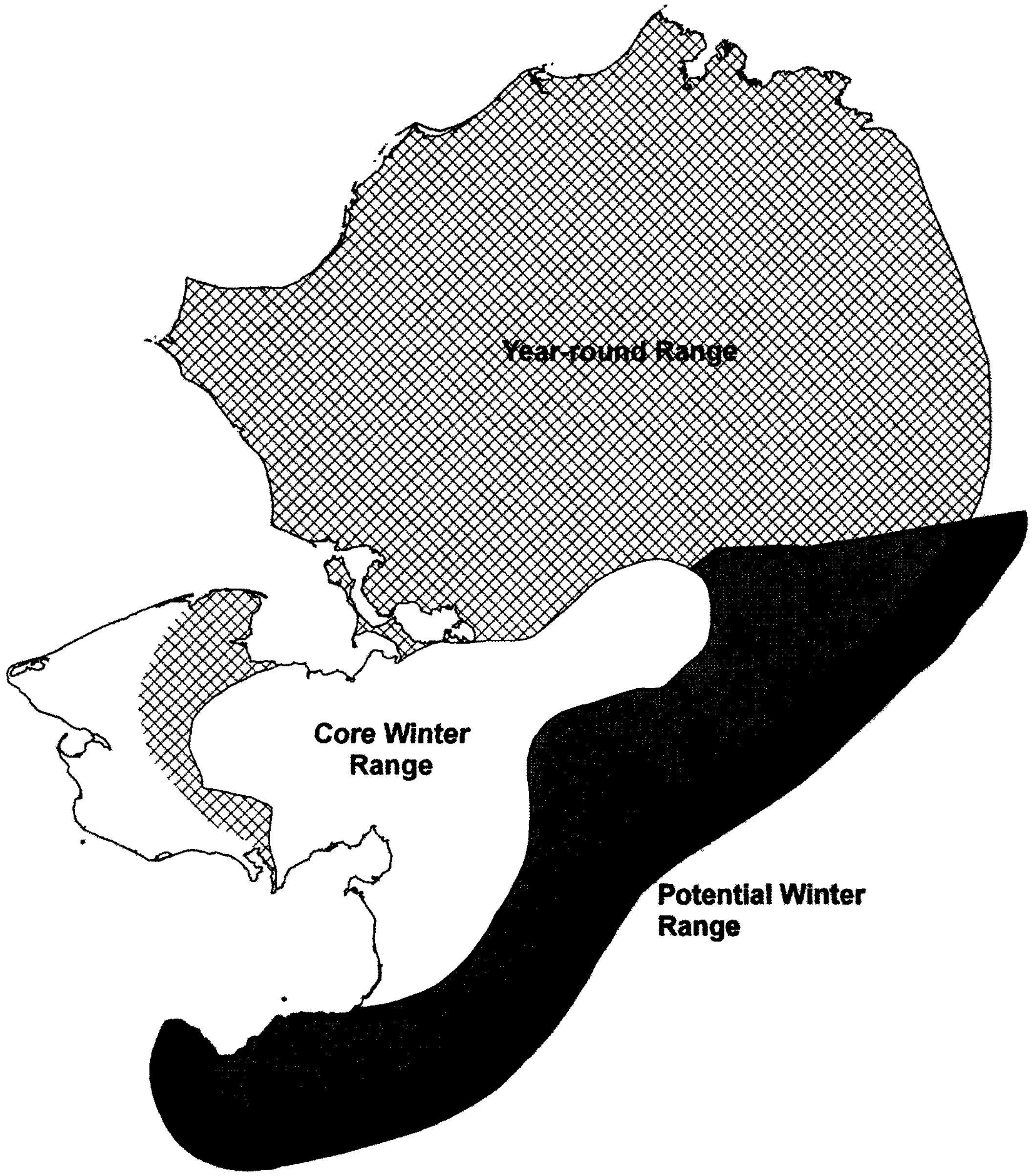

Figure 1.1. Study area. A) Range of the Western Arctic Caribou Herd. B) Conservation System Units of northwest Alaska: dark green, tan, pale yellow are managed by NPS, USFWS, and BLM respectively. C) Ecoregions of northwest Alaska.

Figure 1.1.B.

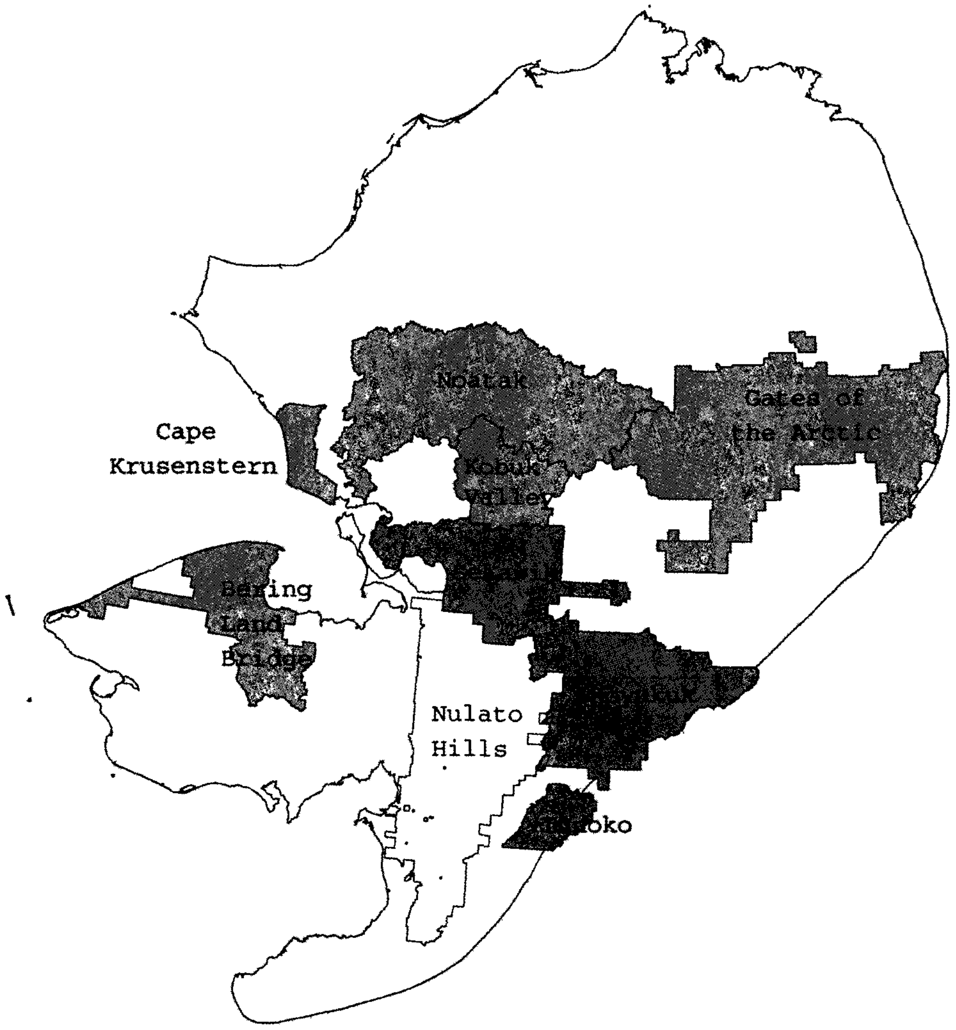

Figure 1.1. Study area. A) Range of the Western Arctic Caribou Herd. B) Conservation System Units of northwest Alaska: dark green, tan, pale yellow are managed by NPS, USFWS, and BLM respectively. C) Ecoregions of northwest Alaska.

Figure 1.1.C.

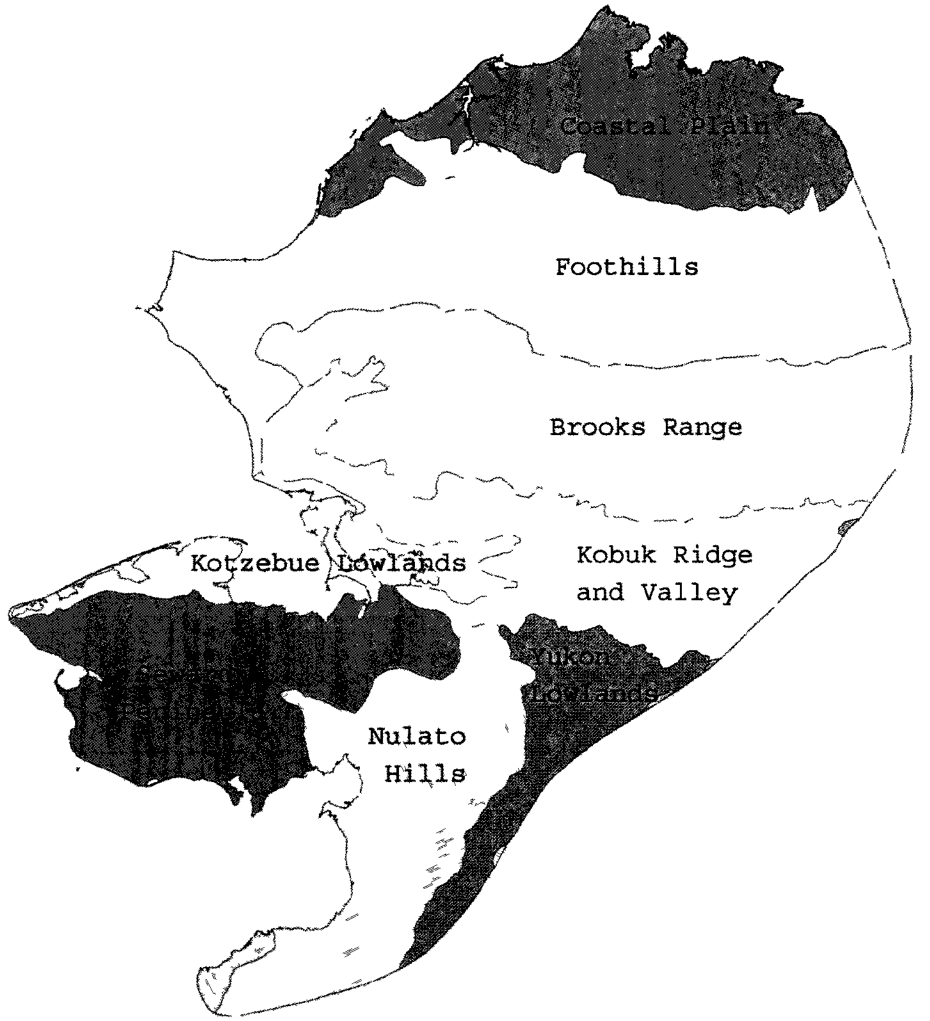

Figure 1.1. Study area. A) Range of the Western Arctic Caribou Herd. B) Conservation System Units of northwest Alaska: dark green, tan, pale yellow are managed by NPS, USFWS, and BLM respectively. C) Ecoregions of northwest Alaska.

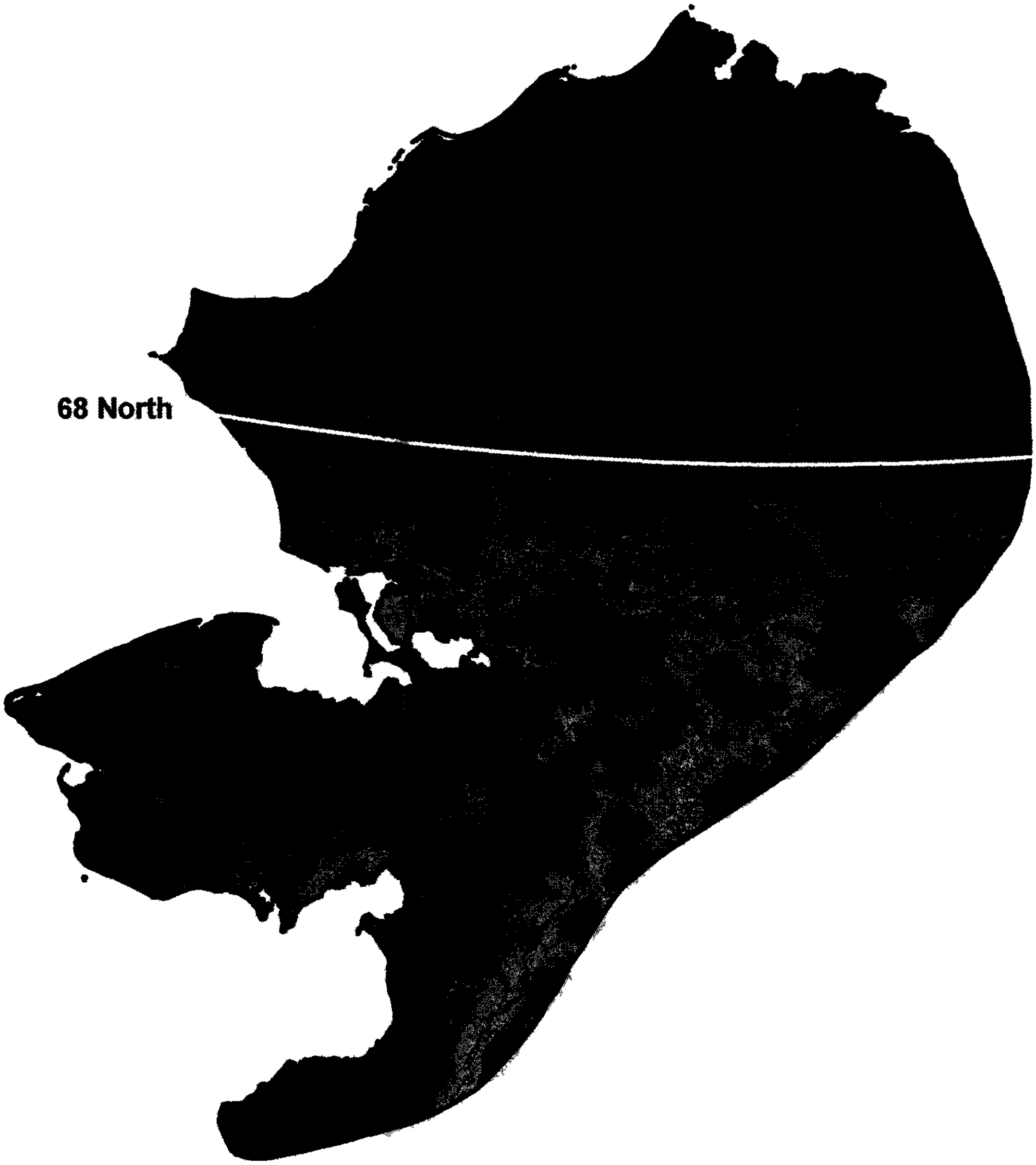

Figure 1.2. Wildfires on the landscape of northwest Alaska, 1950-2007. Red polygons are burned areas, brown areas are dominated by tundra (non-forested) habitats, green is boreal forest ecosystems and blue depicts water. The yellow line is the 68th latitude.

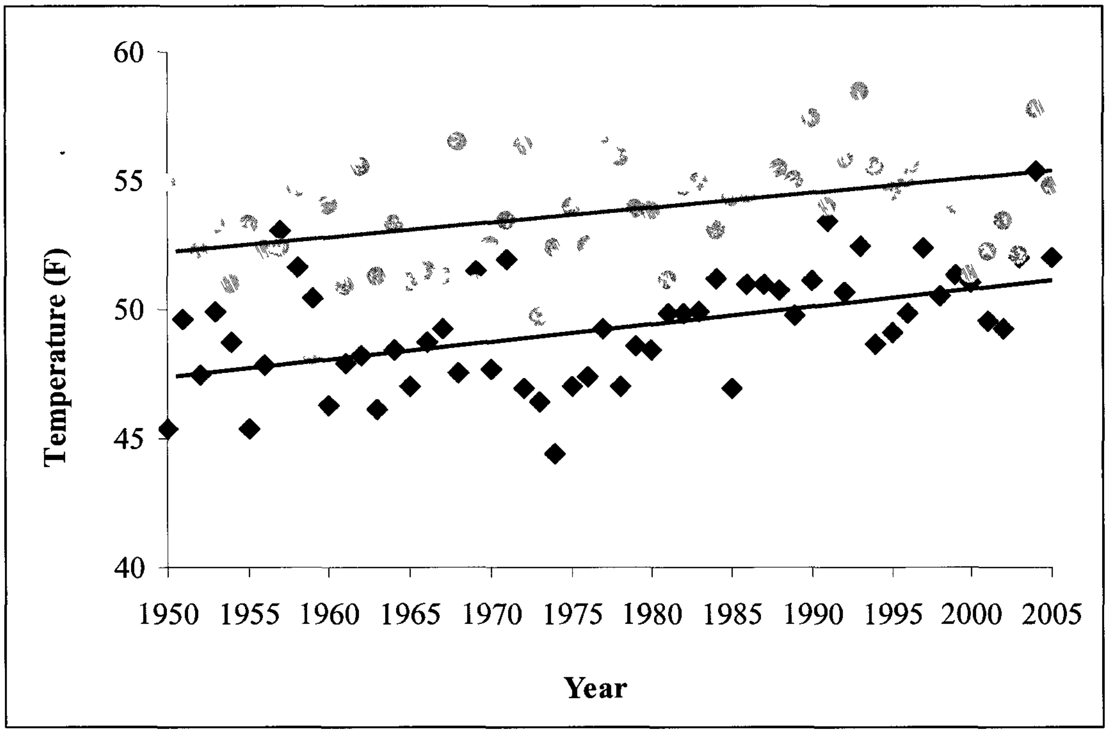

Figure 1.3. Significant increases in average June (blue diamonds) and July (green circles) temperatures from 1950-2005 in northwest Alaska. Corresponding regression lines are in depicted in red (lower line) and orange (upper line). Temperatures represent an average for the Barrow, Bettles, Kotzebue, McGrath, Nome and Tanana weather stations.

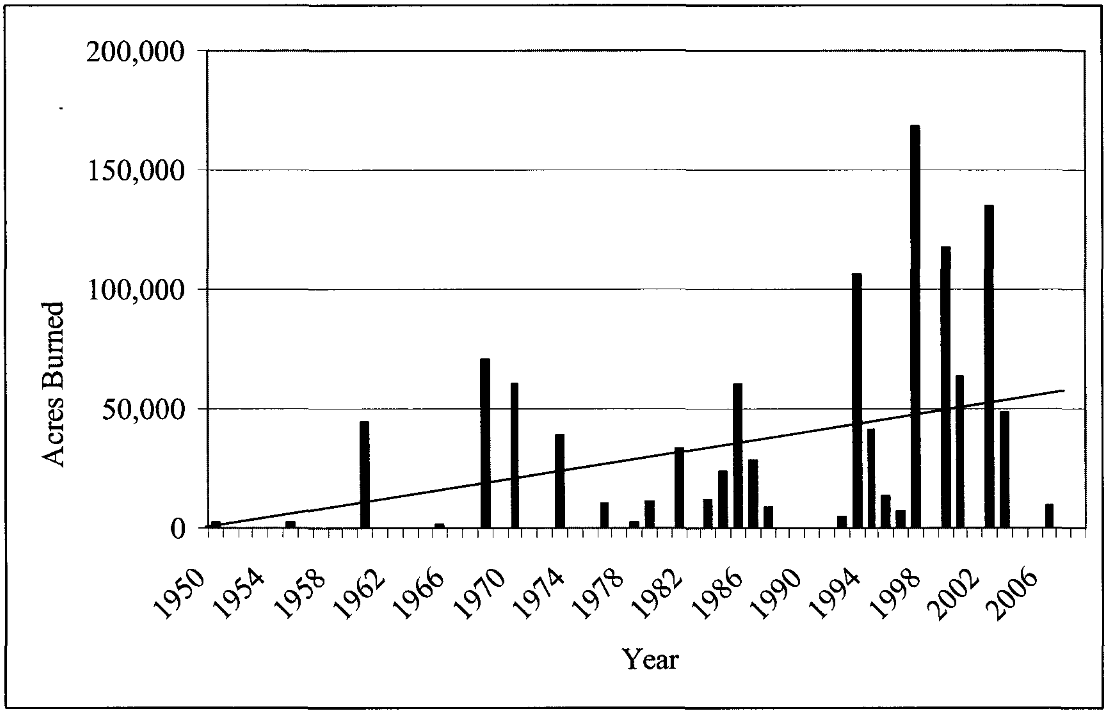

Figure 1.4. Amount of burned area in northwest Alaska, 1950-2007, excluding years with very high acreages (> 200,000, n = 15). Red line is the regression line showing a significant increasing trend in burned area. This trend may be real or an artifact of the large fire database due to varying fire suppression and mapping efforts over the years.

Chapter 2:

Modeling influences on winter distribution of caribou in northwestern Alaska through use of satellite telemetry[1]

Abstract: I hypothesize that the distribution of barren-ground caribou (*Rangifer tarandus granti*) is affected by multiple, interrelated factors. These factors include, but are not limited to, terrain and snow characteristics as well as predation pressure and habitat. To test this hypothesis, I attributed caribou locations derived from satellite telemetry over a 6 year period with terrain (elevation, slope, aspect, and ruggedness), habitat characteristics, and moose density – potentially an index of wolf predation pressure. These locations were compared to random locations, attributed using the same data layers, using logistic regression techniques to develop resource selection functions (RSFs). I found that caribou moved significantly less during mid-winter than early- or late-winter and that cows moved significantly more in April than bulls due to their earlier departure on their spring migration. Distribution was different between cows and bulls. Terrain variables were important factors but were scale-dependent. Cows avoided forested areas,

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[1] In press as: Joly, K. Modeling influences on winter distribution of caribou in northwestern Alaska through use of satellite telemetry. Rangifer.

highlighting the importance of tundra habitats, and selected for dwarf shrub, with relatively high lichen cover, and sedge habitat types. Bulls selected for dryas, coniferous forest and dwarf shrub habitats but against lowland sedge, upland shrub and burned tundra. Cow distribution was negatively correlated with moose density at the scale of the Seward Peninsula. My results support the hypothesis that caribou distribution during winter in northwest Alaska is affected by multiple, interrelated factors. These results may be useful for researchers to track and/or model changes in future patterns of range use over winter.

Introduction

I hypothesize that the distribution of Western Arctic Herd (WAH) caribou (*Rangifer tarandus granti*) is affected by terrain and snow characteristics, as well as habitat and predation pressure. Looking across the northern landscape, caribou ecotype and disturbance (*e.g.*, wildfire and/or industrial development) are also likely to be important factors in determining distribution (Mallory & Hillis, 1998; Johnson *et al.*, 2005; Joly *et al.*, 2007a). The importance of each factor is likely to depend on the scale of the analysis (Wiens, 1989; Rettie & Messier, 2000; Johnson *et al.*, 2004; Gustine *et al.*, 2006; Mayor *et al.*, 2007). Terrain, snow conditions, habitat characteristics and predation pressure are all interrelated to some degree. High elevation, steep slopes and open habitats often have less snow due to wind scouring than do protected valleys or forested habitats. Lichen biomass is typically greater in areas that have a protective snow cover as lichens are susceptible to desiccation and wind abrasion (Holt *et al.*, 2008). Terrain is an important factor in determining winter distribution because certain conditions may be correlated with preferred habitats, as noted above, or may provide improved sightability of predators. Snow characteristics are important because movement can be impeded by deep snow, while foraging efficiency can be reduced by either deep or crusted snow (Skogland, 1978; Fancy & White, 1985; Fancy & White, 1987; Collins & Smith, 1991; Joly *et al.*, 2010b). Habitat is an important factor because lichens comprise the majority of the winter diet of WAH caribou (Saperstein, 1996; Joly *et al.*, 2007b). Pregnant caribou should be the most reliant on high quality habitat during the winter months as their energetic demands are relatively higher than other classes of caribou (Cameron *et al.*, 1993; Barboza & Parker, 2008). Different habitat types may also offer varying levels of predation pressure. Similarly, different snow conditions can change the relatively vulnerability of caribou to predation (Telfer & Kelsall, 1984). Predation pressure is an important factor, as caribou not judging this risk correctly will be killed. However, if a caribou is weakened from poor nutrition and killed by a predator, the ultimate factor in its death is habitat quality - predation would be its proximate cause. Too often, this distinction is not made.

The WAH experienced a population crash in the 1970's, rapidly declining from approximately 242 000 individuals in 1970 to 75 000 individuals in 1976 (Dau, 2007). The herd rebounded, reaching a population apex of approximately 490 000 individuals in 2003 (Dau, 2007). At this height, the density of caribou was 1.35/km^2, which prompted concern about overgrazing (Joly *et al.*, 2007c). The 2007 photo-estimate revealed a 23 %

decline to 377 000 individuals – though the cause of the decline is unknown at this time (Dau, 2007). Significant declines in lichen cover within the core winter range (Joly *et al.*, 2007c) and/or severe winter events (Dau, 2005; 2007) are potential causes. Understanding the drivers of population changes in this herd is important because it serves as a subsistence resource for scores of villages that harvest more than 10 000 caribou annually from this 1 herd (Dau, 2007).

My goals were to 1) document winter distribution of caribou during the period of peak population and 2) determine factors that help explain why caribou go where they do during winter in northwestern Alaska. This information will provide valuable insight into the factors that shape caribou distribution as a basis for predictions of potential changes in caribou distribution if the population continues to decline and to model how the suitability of winter range may change for caribou under different climate-change scenarios.

Material and methods

Study area

The study area is the range of the WAH, which covers the entire 363 000 km^2 of northwestern Alaska (63° – 71° N and from 148° to 166° W; Dau, 2007) and contains 8 major ecoregions (Fig. 2.1, Nowacki et al., 2001). The region transitions from treeless arctic tundra in the north and west to black spruce (*Picea mariana*) stands and eventually to boreal deciduous forests in the south and east. At the northern extreme of the study area, the Coastal Plain is primarily a flat, poorly drained wetland that is underlain by continuous permafrost. The ground of the Brooks Foothills, to the south, is composed of thick continuous permafrost and supports no trees. Low shrubs, sedges, and tussock tundra dominate this region but extensive willow thickets line the many braided rivers and streams (Nowacki *et al.*, 2001). The steep angular peaks of the Brooks Range are largely barren, while alpine vegetation can be found at lower elevations (Nowacki *et al.*, 2001). Forests and woodlands dominate much of the Kobuk Ridge and Valley ecoregion on the southern flanks of the Brooks Range (Nowacki *et al.*, 2001). The Kotzebue Lowlands lie to the west of the Kobuk Ridge and Valley ecoregion and is dominated by tundra and coastal ecosystems. The Seward Peninsula ecoregion is a mosaic of extensive hills, coastal lowlands and isolated rugged mountain complexes (Nowacki *et al.*, 2001). The moist polar climate supports tundra, dryas, and shrub communities (Nowacki *et al.*, 2001). To the east is the Nulato Hills, an ecoregion dominated by low but often rugged hills. Vegetation varies widely with elevation, from well-forested areas in the river valleys to shrubs on side slopes and alpine communities on the ridges and summits (Nowacki *et al.*, 2001). The Yukon Lowlands is dominated by the confluence of the Yukon and Koyukuk Rivers which forms an expansive wetland system complex of deciduous and coniferous forests, tall shrub and muskeg communities (Nowacki *et al.*, 2001).

Caribou can be found throughout their annual range during winter, though use is more concentrated on the Nulato Hills, upper Kobuk River and eastern Seward Peninsula (Joly *et al.*, 2007a). These regions are diverse, with extensive areas of treeless tussock tundra (*Eriophorum vaginatum*, *Carex* spp.), rugged but low elevation (< 1100 m) mountains, and shrub-lined (*Salix* spp., *Alnus crispa*) riparian corridors. Lichens (*Cladina* spp., *Cetraria* spp.), mosses (*Sphagnum* spp., *Polytrichum* spp.) and shrubs (*Betula nana*, *Empetrum nigrum*, *Ledum palustre*, *Vaccinium uliginosum* and *V. vitis-idaea*) are important components in tundra habitats (Joly *et al.*, 2007c). Mean annual precipitation for the region is about 30 cm. Snow cover occurs throughout the winter (October through April), though some areas may be snow free due to wind scouring or uncommon weather events that bring above freezing temperatures and rain. Although average daily temperatures can drop to – 45^0 C during winter, the average daily temperature for the winter months is -3.3^0 C. Mean temperatures have risen significantly over the study period in this region, especially during the winter (Stafford *et al.*, 2000).

Data acquisition and derivation

Caribou were captured as they swam across the Kobuk River at Onion Portage, located within Kobuk Valley National Park, using motorboats. A total of 70 caribou (63 cows and 7 bulls) were instrumented with satellite telemetry collars. Caribou location data were not used for a year after deployment to ensure adequate mixing with the entire herd (Dau, 2007). A total of 7048 locations from the beginning of October through the end of April were collected from 1999-2005. A total of 20 000 random locations were developed using ArcGIS within the range of the herd. Both the satellite and random locations were attributed with the following data that had potential to affect caribou distribution. Elevation was directly obtained from a Digital Elevation Model (DEM). Slope, aspect and terrain ruggedness indices were derived from the DEM using ArcGIS (ESRI, 2006) tools. I converted aspect from degrees into a categorical variable covering the 8 cardinal directions. I created 2 terrain ruggedness coverages, 1 at a relatively fine scale (180 m cell-size) and the other at a relatively coarse scale (1 km cell-size), using a Vector Ruggedness Measure (VRM) developed by Sappington *et al.* (2007). This measure incorporates variability in both the aspect and gradient components of slope so that steep, broken terrain can be distinguished from steep, even terrain (Sappington *et al.*, 2007).

I obtained habitat classification data at 2 scales. The National Land Cover Database of 2001 (NLCD; data available from the Multi-Resolution Land Characteristics Consortium, www.mrlc.gov, accessed November 13, 2008) covers the entire study area with 30 m resolution. The development of this dataset relied heavily on remotely sensed data. The study area was covered by the following broad habitat categories; deciduous forest, coniferous forest, mixed forest, scrub, shrub, sedge, woody wetlands, herbaceous wetlands, bare ground and open water. Forested areas were defined as having > 20 % vegetation cover of trees > 5 m high. If there was > 75 % of one type (not species) of tree it was defined as that type of forest, if neither deciduous nor coniferous trees dominated, then it was defined as mixed. Scrub habitats generally had > 20 % cover of low (< 20 cm high) shrubs and were "often co-associated with grasses, sedges, herbs, and non-vascular vegetation". Shrub habitats were dominated by shrubs between 20 cm and 5 m high such as *Vaccinium uliginosum*, *Betula nana*, and *Salix glauca* but could include early successional or trees stunted by environmental conditions (*e.g.*, mesic black spruce stands overlaying permafrost). Sedge habitats were dominated (> 80 % cover) by sedges, grasses and forbs. This class included tussock tundra. Woody wetlands were areas of forest or shrubland whose soils were periodically saturated with water. Herbaceous wetlands were dominated by herbs (> 80 % cover) and had their soils periodically saturated with water.

The second coverage was a highly detailed habitat vector map, developed by the Soil Conservation Service (SCS; Swanson *et al.*, 1985), was based on extensive ground surveys and low-level photography of the entire Seward Peninsula. This coverage was utilized only when I was performing analyses dealing solely with the Seward Peninsula and represents a fundamentally different dataset and classification system. The SCS delineated over 150 different habitat types within the region. With assistance of local vegetation experts, I aggregated these types into 12 categories; dryas (*Dryas spp.*; 35 – 65 % cover), lowland sedge, lowland low shrub, tussock tundra, lichen (> 24 % cover), upland low shrub, tall shrub, forest, mountain meadow, burned tundra, burned forest and miscellaneous un-vegetated areas. Mountain meadow had > 30 % graminoid cover whereas upland low shrub had < 25 % graminoid cover. The lowland low shrub, mountain meadow, and tussock tundra can have a strong lichen component, with up to 25 % cover. These data were from the 1980's, so burned areas are > 25 years old and did not include recent burns.

Data on wolf densities were specious or nearly 20 years old in the study area and so were not analyzed. Existing data for moose density was much more comprehensive, collected annually concurrent with the study period, and may be an index of wolf density (Bergerud, 2007). I also calculated, using the Hawth's Analysis Tools (Beyer, 2006) ArcGIS extension, the distance from every satellite collar location and every random location to the nearest of the 44 villages within the study area.

Statistical analysis

I used Analysis of Variance (ANOVA) to detect differences among months, between sexes in movement rates, and between satellite location and random points. I employed a logistic regression – resource selection function (RSF) approach to assess factors that influence caribou distribution during winter (Manley *et al.*, 2002). I selected Thomas and Taylor's (1990) Design II, where the locations of individually marked animals are pooled to study population level patterns. Selection or avoidance by caribou was relative to the random locations. Using an information theoretic approach, the best models were determined using Akaike's Information Criteria (AIC_c) for small sample sizes to determine the most parsimonious models (Burnham & Anderson, 2002). The full model was compared to the full model minus one factor using ANOVA techniques to determine significance of individual model parameters. Using the results of these analyses, I developed a resource suitability map. Significant factors were multiplied by beta coefficients derived from the best model, summed and the exponential was taken of the resultant. The final number represents the relative probability of selecting a given location as determined by the RSF (Manley *et al.*, 2002).

Results

Cows moved significantly more than bulls throughout the winter (140 m/hour versus 97 m/hr, respectively; $F_{1,472} = 6.42$, $P = 0.01$; Fig. 2.2). Movement rates declined, for both cows and bulls, from October to December ($F_{1,424} = 112.56$, $P < 0.01$, $F_{1,42} = 21.65$, $P < 0.01$, respectively). Movement rates were lowest during mid to late winter. Cow movement rates (124 m/hr) were significantly greater than bulls (45 m/hr) during the month of April ($F_{1,63} = 5.61$, $P = 0.02$). Cows were found at lower elevations (298 m) and gentler slopes (18^0) than bulls (365 m, 23^0), but due to low sample sizes these differences were not significant ($F_{1,68} = 2.06$, $P = 0.16$, $F_{1,68} = 3.33$, $P = 0.07$, respectively). Because of these differences, I analyzed resource selection separately for bulls and cows.

The best resource selection function model for WAH cow distribution over the entire winter range incorporated slope, aspect, elevation, fine scale (180 m cell-size) terrain ruggedness, habitat and moose density (Table 2.1a). Cow distribution was positively correlated with slope and fine scale terrain ruggedness but negatively with elevation (Table 2.2a). Correlation with moose density was not significant. Aspect and habitat were significantly correlated with cow distribution as well (Table 2.2a). Cows significantly selected southwest to northwest aspects over others and avoided flat (no aspect) terrain (Table 2.2a). Scrub, shrub and sedge habitats were significantly preferred, while deciduous and mixed forests and perennial snowfields were used significantly less than expected. The resource suitability map, depicted in Fig. 2.3, reveals extensive areas of relatively high quality winter habitat in the western (Seward Peninsula ecoregion) and southern Nulato Hills. Areas with lower probability of use include the central Brooks Range and the Yukon Lowlands.

Limiting the analysis to the Seward Peninsula, and using the more detailed SCS habitat map, the best model for cow winter distribution incorporated aspect, elevation, fine scale (180 m cell-size) terrain ruggedness, coarse scale (1 km cell-size) terrain ruggedness, habitat, and moose density (Table 2.1b). Cow distribution was positively associated with elevation but negatively with coarse scale terrain ruggedness and moose density (Table 2.2b). Aspect and habitat were significantly correlated with cow distribution (Table 2.2b). Cows significantly preferred northeastern aspects. Cows used lowland low shrub, tussock tundra, and mountain meadow habitats preferentially.

The differences between the analysis of the distribution of cows for the entire range and that focusing on the Seward Peninsula included: a change in the correlation with elevation from positive to negative, and negative correlations with moose density and coarse scale terrain ruggedness on the Seward Peninsula. By conducting a second analysis utilizing the range-wide (NLCD) vegetation classification, I was able to directly compare habitat selection for the entire winter and the Seward Peninsula. Selection was very similar for both regions. Cows significantly preferred dwarf scrub and sedge habitats and avoided coniferous forests in both regions. Correlations with deciduous forest (-), mixed forest (-) and dwarf shrub (+) were not significant for the Seward Peninsula, but showed the same tendency as the correlations did for the entire winter range.

Analyses of bull distribution should be viewed with caution due to limited sample size (n = 7). The best resource selection function model for bull distribution over the entire winter range incorporated slope, elevation, fine and coarse scale (180 m and 1 km cell-size) terrain ruggedness, habitat, and moose density (Table 2.1c). Bull distribution was positively correlated with slope and coarse scale terrain ruggedness, but negatively correlated with elevation (Table 2.2a). Habitat was significantly correlated with bull distribution (Table 2.2a). Bulls selected scrub and coniferous forest habitats. Bull distribution differed from cows in that they were 1) positively associated with coarse scale, not fine scale, terrain ruggedness, and 2) did not show avoidance of deciduous forests and 3) associated with fewer habitat classes.

Limiting the analysis to the Seward Peninsula and the SCS habitat map, the best model for bull distribution incorporated slope, elevation, fine and coarse scale (180 m and 1 km cell-size) terrain ruggedness, habitat, and moose density (Table 2.1d). Bull distribution was positively correlated with elevation but negatively with coarse scale terrain ruggedness (Table 2.2b). Bulls showed significant preference for dryas communities, while avoiding burned tundra, lowland sedge, and upland low shrub communities (Table 2.2b). Similar to cows, the range-wide analysis for bulls revealed a negative correlation between distribution and elevation whereas on the Seward Peninsula the correlation was positive. Also, the correlation with coarse scale terrain ruggedness changed from positive to negative moving from the range-wide to Seward Peninsula analyses. Caribou locations (49.7 km ± 0.5 km) were significantly closer to villages than random locations (68.6 km ± 0.3 km) within the study area ($F_{1,27047} = 1272.25$, $P < 0.01$).

Discussion

A complex interaction of multiple, interrelated factors drive the winter distribution of WAH caribou. My results suggest that studies that focus on a single factor as the presumed determinant of caribou population distribution or dynamics may fail to capture the full, actual situation except under rare cases. The relative importance of predators, habitat, and other factors will be very case specific (Skogland, 1991). For the WAH, all 3 general factors I analyzed (terrain, habitat and predation pressure) were correlated with caribou distribution in winter. Other factors, such as disturbance from wildfire (Joly *et al.*, 2007c; Joly *et al.*, 2010a) and industrial development (Vistnes & Nellemann, 2008), which I did not analyze, might also be important for the WAH and other northern caribou herds. By analyzing multiple factors, researchers also garner insight into the cumulative effects these factors may have on caribou (see also Nellemann & Cameron, 1998; Johnson *et al.*, 2005).

The nature and relative importance of terrain features on WAH caribou distribution depended on scale – both of the landscape features themselves and of the extent of the study area. Caribou preferred relatively lower elevations across their winter range but relatively higher elevations on the Seward Peninsula. Average elevation was significantly higher on the winter range outside the Seward Peninsula than within it. Thus selection or avoidance of certain terrain features depends on the landscape available to WAH caribou. Two factors that may help explain these results are vegetation and snow, which are related to both elevation and differ between the entire range and just the

Seward Peninsula. Higher terrain is common throughout the herd's range (e.g., the Brooks Range) and is associated with sparsely or non-vegetated areas; providing little forage and thus caribou would utilize relatively low terrain. Relatively high terrain is much more limited on the Seward Peninsula. Furthermore, the Seward Peninsula is a maritime climate and receives more snow on average than most of the range which experiences climate conditions more typical of continental areas. Deep snows accumulate in the lowlands of the Seward Peninsula and would explain caribou preference for relatively higher elevations there as ridges tend to be more windswept and have lower snow depths in general. Ridges with low snow accumulation tend to enhance the predictability of winter range use (Russell *et al.*, 1993). A similar, but opposite, relationship was found with coarse scale terrain ruggedness between these regions. This suggests that there may be threshold values of terrain features where caribou usage will be greatest. WAH cows showed a positive relationship with fine scale terrain ruggedness over the entire winter range. This uneven terrain may provide a diversity of habitats for foraging and softer snow conditions that allow access.

Cow distribution on the Seward Peninsula was negatively correlated with moose density. This result may seem intuitive as caribou tend to avoid habitat that has recently burned (Joly *et al.*, 2007a; Joly *et al.*, 2010a), whereas moose select for it (Maier *et al.*, 2005). Furthermore, high moose densities could support high wolf densities which would reduce its suitability for caribou (Bergerud, 2007). However, moose density was not well correlated with cow distribution throughout the winter range or bull distribution at either scale, and these relationships were positive in nature. A positive correlation between caribou and wolf density could develop if wolves were successful in areas that had consistently high caribou densities during winter. Thus the lack of significant correlations among moose density and cow (entire winter range) and bull (both over the entire winter range and the Seward Peninsula) distribution may indicate that moose density may not be an adequate index of wolf density and/or the effects of predator densities on caribou distribution is more complicated than simple selection or avoidance.

WAH cows avoided forested areas across the winter range and preferred scrub, shrub and sedge habitats, highlighting the long-known importance of tundra habitats (Murie, 1935; Skoog, 1968). I found a strong agreement between the habitat associations throughout the winter range and those found on the Seward Peninsula for WAH cows. These habitat types typically have relatively high lichen cover (Swanson *et al.*, 1985). Lichens are an important component of the winter diet of WAH caribou, making up a majority of their forage (Saperstein, 1996; Joly *et al.*, 2007b). Concurrent with major declines in lichen cover within the core winter range of the WAH (Joly *et al.*, 2007c) and the percentage of lichens in their winter diet (Joly *et al.*, 2007b), the size of the WAH peaked and has declined for the first time in 30 years. Though only anecdotal, this evidence supports the theory (Klein, 1991) that lichens may be a critical component of the winter diet of large migratory herds in North America (see also Holleman *et al.*, 1979). This does not, however, refute the importance of predators on *Rangifer* population dynamics, especially at lower densities. Nor does it preclude the possibility that other factors, such as severe winter weather (Dau, 2007; Joly *et al.*, 2010b), are the major driver or have had additive effects.

The distribution of bulls differed from that of cows. Preference of habitat types was muted in comparison to cows, though bulls avoided lowland sedge habitats. Bulls were found at higher elevations and steeper slopes than cows. These conditions are often associated with more open habitats, as was seen with the affinity for dryas community types on the Seward Peninsula by bulls. Also, bull distribution was not correlated with fine scale terrain ruggedness, as cow distribution was. These differences in distribution point to the use of alternative overwintering strategies between the sexes.

Though hampered by low sample sizes, my analyses suggest that bulls may be adopting an energy conservation strategy that favors reducing exposure to predation, whereas cows are sacrificing exposure to predators in return for maximizing energy intake by utilizing habitats with greater lichen forage. Higher movement rates by WAH cows, as compared to bulls, throughout the winter months supports this theory of differing overwintering strategies (Roby & Thing, 1985). Vigilance alone does not explain these differences as bulls found in higher, open habitat could identify approaching predators at a greater distance than foraging cows but the large group sizes of cow and young caribou would improve vigilance relative to the smaller bull groups. The smaller group sizes would allow bulls to utilize smaller patches and exert less grazing pressure within an area. Cows, which retain their antlers over the winter, would also have a competitive advantage in maintaining and/or usurping optimal foraging locations and feeding craters (see Holand *et al.*, 2004).

Ultimately, the trade-offs between predatory exposure and forage intake are likely due to differing energetic demands. A vast majority of cows are pregnant during the winter months; this extra energetic demand may induce cows to try to maximize energy intake through foraging rather than adopting an energy conservation strategy utilized by bulls. These strategies may be reversed in spring when cows head towards calving grounds with lower predator densities and bulls lag behind consuming emergent green vegetation high in protein content (Heard *et al.*, 1996).

The RSF map (Fig. 2.3) reveals higher probability of use in the Nulato Hills and Seward Peninsula. Use of the northern Brooks Foothills by WAH caribou has been limited despite moderately high probability of use as determined by the RSF (Fig. 2.1, Fig. 2.3). This lends further support to the argument that lichens are an important winter forage for WAH caribou, as forage lichen abundance is very low in this ecoregion but snow depths and wolf densities are favorable (both low) for caribou compared to other portions of the winter range. However, limitations in the RSF cannot be ruled out as an explanation for this discrepancy. Expansion of the winter range to the southeast, into the Yukon Lowlands ecoregion seems unlikely as the probability of use as determined by the RSF was quite low. Furthermore, this area already supports high wolf densities without having regular or extensive usage by the WAH, more wildfire, and lower biomass of lichens (Joly *et al.*, 2010a). The western reaches of the Seward Peninsula have not been extensively used by the herd, had high probability of use and thus represent an area that has potential as an area for the herd to expand its winter range. This portion of the Seward Peninsula includes the largest towns and remaining reindeer (*Rangifer tarandus tarandus*) herds in the region, which could present problems if the herd did expand its range there (Dau, 2000).

Management implications

In order to better understand caribou distribution in winter, better information on predator densities, habitat, snow conditions, and weather should be collected. While efforts are currently underway to improve our understanding of most of these factors, it cannot be said for predator densities. To better understand caribou distribution and population dynamics in northwest Alaska, improved information is needed on predator distribution, predator abundance, predation rates and the factors that regulate them. A transition from traditional satellite collars to GPS-satellite collars will improve researchers' ability to analyze caribou movements, distribution and habitat use within the region (Joly, 2005; Joly *et al.*, 2010a). Dramatic changes are taking place rapidly in the Arctic and on the winter range of the WAH specifically (ACIA, 2005; Joly *et al.*, 2007c). The analyses presented here provide a useful foundation for modeling the effects of future potential climate regimes on the abundance and quality of caribou winter range in northwest Alaska.

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Table 2.1. Model selection for Western Arctic Herd caribou distribution during winter (October through April) from 1999-2005, northwest Alaska. Analyses were conducted for cows and bulls for the entire winter range and just the Seward Peninsula.

A) Cows throughout the winter range

Model Parameters	df	AIC	ΔAI
Aspect, Slope, Elevation, Ruggedness (180m), Habitat, Moose	25	28687.8	-
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat,	26	28688.0	0.26
Aspect, Slope, Elevation, Ruggedness (180m), Habitat,	24	28688.0	0.26
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat	25	28688.3	0.48
Aspect, Slope, Elevation, Ruggedness (1km), Habitat, Moose	25	28699.4	11.6

B) Cows on the Seward Peninsula

Model Parameters	df	AIC	ΔAI
Aspect, Elevation, Ruggedness (180m and 1km), Habitat,	24	8093.46	-
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat,	25	8094.55	1.09
Aspect, Slope, Elevation, Ruggedness (1km), Habitat, Moose	24	8094.69	1.23
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat	24	8096.75	3.29
Slope, Elevation, Ruggedness (180m and 1km), Habitat, Moose	17	8099.63	6.18

Table 2.1 continued.

C. Bulls throughout the winter range

Model Parameters	df	AIC	ΔAIC
Slope, Elevation, Ruggedness (180m and 1km), Habitat, Moose	18	4329.08	-
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat	25	4330.56	1.48
Aspect, Slope, Elevation, Ruggedness (1km), Habitat, Moose	25	4330.92	1.84
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat,	26	4332.02	2.94
Aspect, Elevation, Ruggedness (180m and 1km), Habitat,	25	4340.28	11.20

D. Bulls on the Seward Peninsula

Model Parameters	df	AIC	ΔAIC
Slope, Elevation, Ruggedness (180m and 1km), Habitat, Moose	17	1309.64	-
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat	24	1317.40	7.76
Aspect, Elevation, Ruggedness (180m and 1km), Habitat,	24	1317.70	8.06
Aspect, Slope, Elevation, Ruggedness (180m and 1km), Habitat,	25	1319.36	9.71
Aspect, Slope, Elevation, Ruggedness (1km), Habitat, Moose	24	1319.43	9.79

Table 2.2. Comparison of coefficients of selection (β_i) and standard errors (SE) of factors in the best models describing Western Arctic Herd caribou distribution in winter from 1999-2005, northwest Alaska. (+) indicates a positive correlation while (-) a negative one.

A) Entire winter range

	Cows (n = 63)		Bulls (n = 7)	
Factors	β_i	SE	β_i	SE
Aspect - SW	0.154 *	0.060		
Aspect - W	0.269 **	0.057		
Aspect - NW	0.145 *	0.058		
Aspect - Flat	-0.581	0.090		
Slope	0.021 **	0.001	0.016 **	0.004
Elevation	-0.001	0.001	-0.004 **	0.001
Ruggedness 180m	3.318 **	0.641		
Ruggedness 1km			4.044 **	0.870
Perennial snow	-2.890	1.010		
Deciduous forest	-0.717	0.220		
Coniferous forest			0.938 *	0.413
Mixed forest	-1.187	0.243		
Dwarf scrub	0.727 **	0.109	0.946 *	0.394
Shrub/scrub	0.436 **	0.112	0.813 *	0.400
Sedge	0.615 **	0.109		
Woody wetlands	0.269 *	0.136		

* $P < 0.05$, ** $P < 0.01$

Table 2.2 continued.

B) Seward Peninsula

	Cows (n = 63)		Bulls (n = 7)	
Factors	β_i	SE	β_i	SE
Aspect - NE	0.239 *	0.109		
Elevation	0.001 **	0.001	0.001 **	0.001
Ruggedness 1km	-5.670	0.780	-8.169 **	2.861
Burned tundra			-1.320 *	0.560
Dryas			0.817 *	0.365
Lowland low shrub	1.016 *	0.516		
Lowland sedge			-1.327 **	0.408
Tussock tundra	1.276 *	0.507		
Upland low shrub			-1.148 *	0.481
Moose density	-0.273 *	0.134		

* $P < 0.05$, ** $P < 0.01$

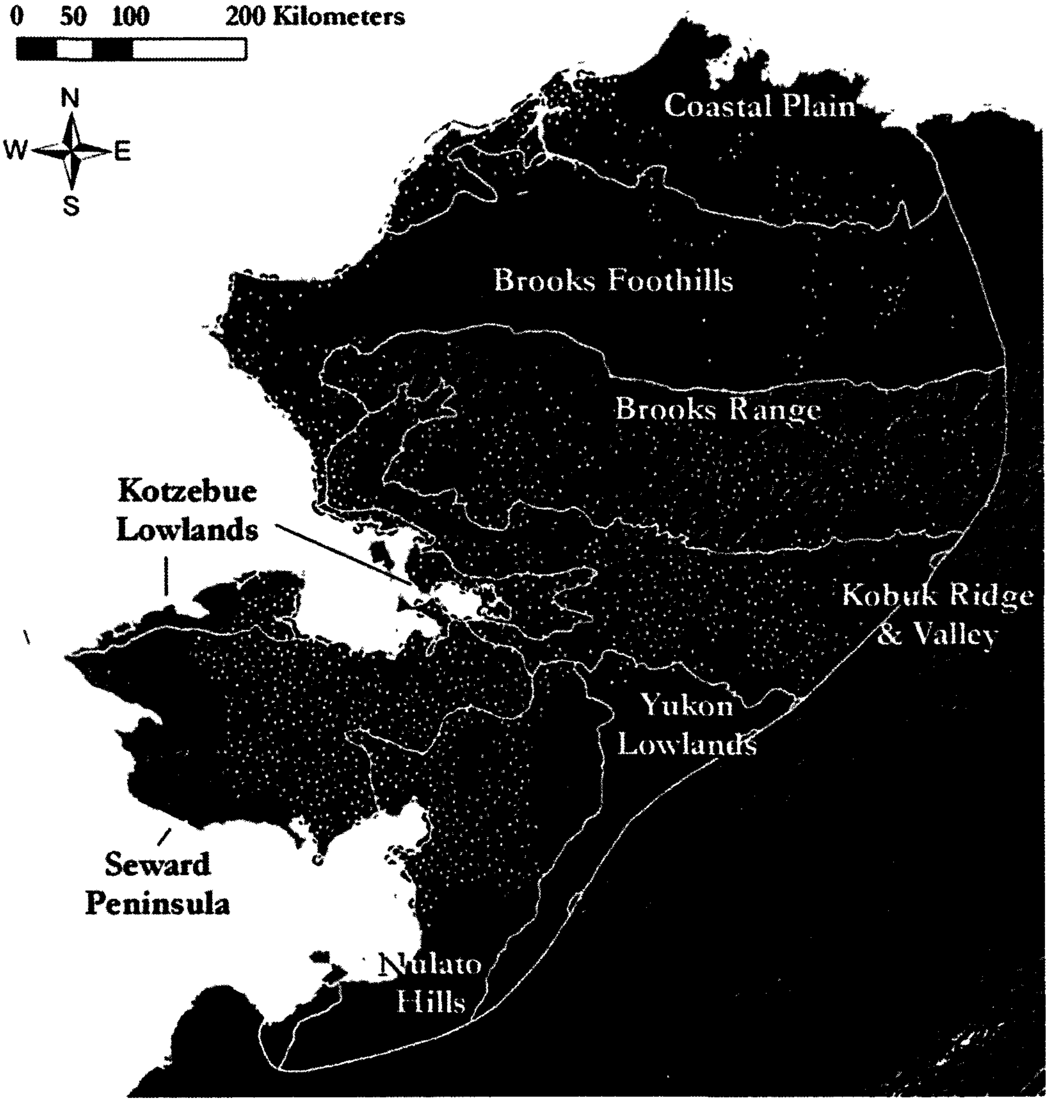

Fig. 2.1. Distribution of Western Arctic Herd caribou, 1999 – 2005, during winter (October through April), northwest Alaska. Caribou locations acquired by satellite telemetry from 63 cows and 7 bulls are represented by light-colored dots. The ecoregions covering the range of the herd are labeled and outlined in light gray.

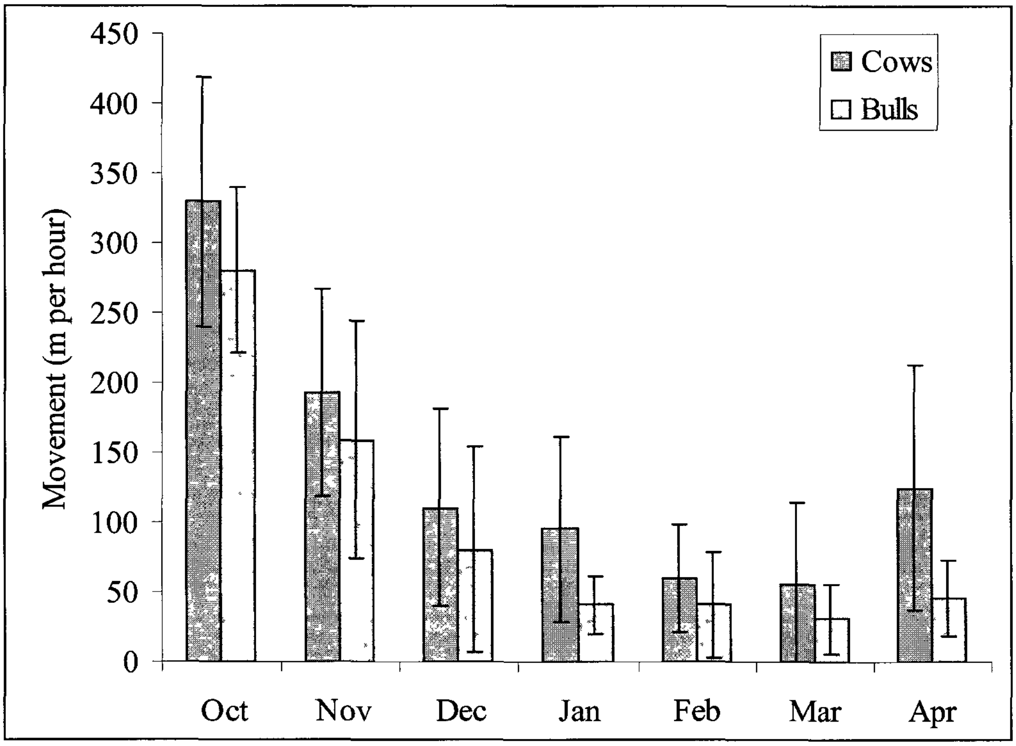

Fig. 2.2. Winter time movement rates of satellite collared Western Arctic Herd caribou from 1999-2005, northwest Alaska.

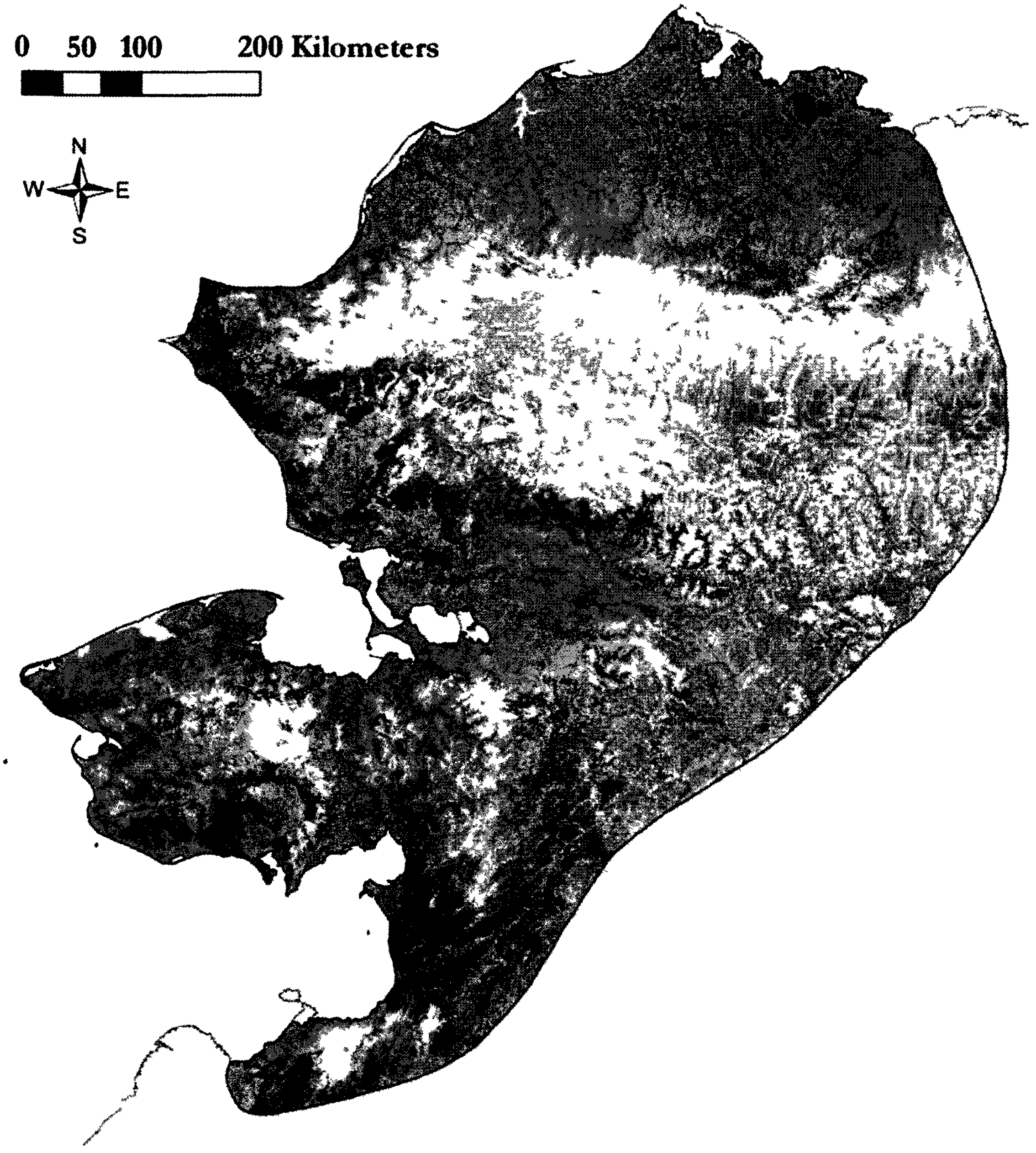

Fig. 2.3. Resource suitability map for Western Arctic Herd cow caribou during the winters (October through April) from 1999-2005, northwest Alaska. Red shades represent greater suitability (relative probability of selection) and blue indicates lower suitability.

CHAPTER 3:

Winter habitat selection by caribou in relation to lichen abundance, wildfires, grazing and landscape characteristics in northwest Alaska[1]

Abstract Lichens are an important winter forage for large, migratory herds of caribou (*Rangifer tarandus granti*) that can influence population dynamics through effects on body condition and in turn calf recruitment and survival. We investigated the vegetative and physiographic characteristics of winter range of the Western Arctic Herd in northwest Alaska, one of the largest caribou herds in North America. We made 3 broad comparisons: habitats used by caribou versus random locations, burned versus unburned habitats, and habitats within the current winter range versus those in the historic winter range and potential winter ranges. We found that lichen abundance was more than 3 times greater at locations used by caribou than found at random. The current winter range does not appear to be overgrazed as a whole, but continued high grazing pressure and consequences of climate change on plant community structure might degrade its condition. Within the current winter range, lichen abundance was more than 4 times

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greater at unburned locations than at recently (< 58 years) burned locations. Other than lichen abundance, there were few vegetative differences between burned ($\bar{x} = 37 \pm 1.7$years) and unburned locations. The historic winter range has low lichen abundance likely due to sustained grazing pressure exerted by the herd, which suggests that range deterioration can lead to range shifts. Recovery of this range may be slowed by continued grazing and trampling during migration of caribou to and from their current winter range, as well as by high wildfire frequency and other consequences of climate change. The area identified as potential winter range is unlikely to be utilized regularly by large numbers of caribou primarily due to low lichen abundance associated with extensive deciduous stands, large areas of riparian habitat, high moose (*Alces alces*) densities, and greater prevalence of wildfire. Our results suggest that lichens are important in the overwintering ecology of caribou that face the energetic costs of predator avoidance and migration.

Nomenclature: Brodo *et al.,* 2001; Feldhammer *et al.*, 2003; Viereck & Little 2007

Introduction

Caribou (*Rangifer tarandus*) distribution and population dynamics are driven by the complex interactions of numerous influences including climate, predation, habitat, year-round forage quality, quantity and availability, insects, parasites, diseases, density-dependence, disturbance, industrial development and other factors (Bergerud, 1980; Messier *et al.*, 1988; Klein, 1991; Post & Stenseth, 1999; Johnson *et al.*, 2005; Brown *et al.*, 2007; Joly *et al.*, 2007c; Pederson *et al.*, 2007; Briand *et al.*, 2009; Couturier *et al.*, 2009; Hughes *et al.*, 2009; Vors & Boyce, 2009). While it is important to understand that caribou population dynamics are a product of complex interactions among these factors, it is difficult to conduct research that simultaneously incorporates all of them. The quality of winter range has been a frequent focus of caribou research because range quality can affect body condition, fetal development, birth weights and growth rates of calves, and milk production (White, 1983; Parker *et al.*, 2005). Calf weights, in turn, are associated with herd productivity (Couturier *et al.*, 2009). Thus, poor winter range conditions can have a pronounced impact on caribou population dynamics through its effects on recruitment.

Body condition, inclusive of available energy and protein stores, affects the ability of cervids to survive and reproduce (Parker *et al.*, 2009). High-protein forage is limited in its availability to cervids during late winter when gestational demands are high (Parker *et al.*, 2005; Brown *et al.*, 2007). Caribou are unique among cervids in this respect because of the primary role lichens generally play in their winter diet. Lichens, though high in available energy, have a protein content that, by themselves, is below maintenance levels for most northern cervids (Person *et al.*, 1980; Parker *et al.*, 2005). Although dietary protein requirements during winter are low in contrast to the summer period (White *et al.*, 1981), an inadequate protein balance can cause catabolism of body tissues, reduce growth or compromise in utero development in cervids (Parker *et al.*, 2005). Caribou acquire most protein during summer, when high-protein forage is available, and make greater use of lichens, which are high in digestible energy, in winter, when energy demands are high (Parker *et al.*, 2009). Caribou distribution in winter must balance the competing demands of acquiring enough quality forage to meet their energetic demands and avoiding predation (Brown *et al.*, 2007; Parker *et al.*, 2009).

Large, migratory herds of barren-ground caribou generally rely heavily upon terricolous, fruticose lichens, especially those of the *Cladina* genus, during winter (Person *et al.*, 1980; White *et al.*, 1981; Klein, 1982, 1991; Heggberget *et al.*, 2002). The Western Arctic Herd (WAH), which ranges over 360,000 km^2 of northwest Alaska, represents a quintessential herd of this type (Davis *et al.*, 1980; Dau, 2007). Terricolous lichens constitute the majority of forage for WAH caribou during winter (Saperstein, 1996; Joly *et al.*, 2007b). Some have suggested, however, that lichens are not essential for caribou during winter and that a graminoid-dominated diet may be adequate for sustaining caribou and reindeer (also *R. tarandus*) populations (Bergerud, 1974; Heggberget *et al.*, 2002; van der Wal, 2006). However, this assertion is based on low-density, insular, non-migratory and/or predator- and parasitic insect-free populations. Recently (< 55 years old) burned habitats exhibit rapid and vigorous re-growth of graminoids (Jandt *et al.*, 2008), though low lichen cover, but are avoided during winter by WAH caribou (Joly *et al.*, 2007a). An analysis of the winter diets of the WAH revealed that graminoids were selected against despite their increasing prevalence in the landscape (Joly *et al.*, 2007b). These findings support the hypothesis that lichens are an important forage for this herd during winter.

The abundance of forage lichens is typically associated with particular plant community types, soil characteristics, topography, grazing pressure and time since disturbance (Morneau & Payette, 1989; Swanson, 1996; Racine *et al.*, 2004; Holt *et al.*, 2006, 2008; Jandt *et al.*, 2008). Abundance is greatest where competition from vascular species is low (such as on acidic or sandy soils) and where there is low frequency of disturbance by flooding or wildfire (Swanson, 1996; Holt *et al.*, 2006, 2008). Following a wildfire, it can take several decades for forage lichens to return to their former levels (Racine *et al.*, 2004; Dunford *et al.*, 2006; Holt *et al.*, 2008; Jandt *et al.*, 2008). Grazing and trampling by large numbers of caribou or reindeer can also reduce lichen abundance at the landscape level (Klein, 1968; Moser *et al.*, 1979; Morneau & Payette, 1989; Arseneault *et al.*, 1997; Joly *et al.*, 2007c).

Declining populations of *Rangifer* have been linked to climate change (Vors & Boyce, 2009). The influences of climate change are already apparent in the Arctic and are predicted to become more pronounced over time (Callaghan *et al.*, 2004). Indeed, the largest changes in faunal populations of the western hemisphere are projected to occur in arctic tundra habitats (Lawler *et al.*, 2009). Experimental, field, and theoretical studies suggest that climate change negatively impacts lichen abundance (see review by Joly *et al.*, 2009a). Climate change-related factors that act to reduce lichen abundance include competition with vascular plant species (which will increased shading and leaf litter), wildfire, warming and associated drying, and sustained grazing by *Rangifer*. Future reduction of lichen abundance may negatively affect caribou population dynamics.

Our main goal was to identify characteristics of primary caribou winter range by comparing sites used by caribou to random locations. Secondarily, we quantified differences between burned and unburned habitat and among the herd's current, historic and potential future winter ranges. Finally, we identified landscape-level and site-specific factors that were associated with the abundance of forage lichens. We hypothesized that caribou would seek out and locate areas of high lichen abundance, that unburned habitat would provide greater lichen abundance than recently (< 58 years) burned habitat and that the herd's current winter range would provide greater lichen abundance than either the historic or potential winter ranges. This research has implications for the relative importance of winter forage for caribou, will inform critical and costly decisions regarding fire management, and assist in the management of caribou as a subsistence and economic resource for communities in the Arctic.

Methods

STUDY AREA

With a population size of 377,000 in 2007, the WAH is the largest caribou herd in Alaska (Dau, 2007). Herd size peaked in 2003 at 490,000 caribou (Dau, 2007). The annual range of the WAH covers all of northwest Alaska, some 360,000 km^2 (Fig. 3.1; Davis *et al.*, 1980). WAH caribou can be found throughout most of this range during the winter when at high population levels (~300,000 caribou; Joly *et al.*, 2007a). Winter use has been concentrated in the "current winter range" since at least the 1990s (Fig. 3.2; Dau, 2007). This range is dominated by the Nulato Hills, a region of rugged but low-lying hills. Recently (circa 1996), the herd expanded its winter range to include the Seward Peninsula (Dau, 2007). Vegetation ranges from coastal wet meadows to tussock tundra, alpine tundra, boreal forest, narrow riparian corridors, and brush (*Alnus* spp., *Salix* spp.) fields. In the past, winter use was concentrated further north, in the Selawik National Wildlife Refuge, the Kiana Hills and the upper Kobuk River – referred to henceforth as the "historic winter range" (Fig. 3.2). This region is more forested to the east and transitions to a vast wetland complex within the refuge. To the southeast of the current winter range lies the Koyukuk National Wildlife Refuge and the Yukon Lowlands (Nowacki *et al.*, 2001) where the Koyukuk River drains into the Yukon River. This region is dominated by boreal forest and contains vast riparian complexes. If the herd continues to expand its winter range, it might utilize this region which we refer to as its "potential winter range" (Fig. 3.2). For additional details, see Joly *et al.* (2009b).

VEGETATION PLOT LOCATION, DATA COLLECTION, AND STATISTICAL ANALYSIS

We developed 8 treatment types for our study plots. Within the current winter range (Fig. 3.2), we collected data at locations used by caribou that were unburned (T1) and burned (T2) and random locations that were unburned (T3) and burned (T4). In an area to the southeast of the current winter range, where the herd could potentially expand their winter range (hereafter potential winter range), we created plots using random locations that were unburned (T5) and burned (T6). Similarly, within the herd's historic winter range, in the Selawik National Wildlife Refuge, we created plots at random locations that were unburned (T7) and burned (T8). Plot locations used by caribou were randomly selected from a set of satellite telemetry locations from 1999-2005 (Joly *et al.*, 2007a). Caribou locations in the database were nearly unique (i.e., only 12 of 7049 [< 0.2 %] had the same latitude and longitude as another caribou location) and thus the influence of frequency of usage was minimized. Unique, random locations were generated using ArcGIS 9.2 (ESRI, 2006; Redlands, CA). Burned status was determined using the Alaska Fire Service's Large Fire Database which maps the perimeters of fires dating back to 1950 (data available at http://fire.ak.blm.gov/, accessed October 5, 2009). Thus, plots designated as burned have done so within the past 58 years. However, the fire perimeters include unburned patches, also called inclusions, within the perimeter. We changed the designation of plot treatment types if there was solid evidence that it was mis-categorized using the Geographic Information System (GIS). For example, if a plot was designated T3 (random/unburned) but there were obvious signs of burning (i.e., standing burned poles or char), the plot would be re-designated T4 (random/burned). Similarly, presence of obvious caribou sign (e.g., pellets, shed antlers, evidence of cratering) would cause the re-designation of a T3 plot to a T1 (used/unburned) plot type.

We used helicopters to access the plots, from 28 June to 22 July 2008, and hand-held Geographic Position System (GPS) units to navigate the plot origin. A 10x10-m grid was formed from the origin. Vegetation was identified to species level, where possible, at 4 vertical layers every meter within the grid for a total of 100 intercepts per plot. The layers were ground (e.g., lichens, mosses), herbaceous (e.g., sedges, grasses, herbs), shrubs <1m (e.g., *Ledum* spp., *Vaccinium* spp.) and canopy >1m (e.g., *Picea* spp., *Populus* spp.). We used these data to determine percent cover. We considered *Cladina arbuscula/mitis* (referred to *C. mitis* henceforth), *C. rangiferina/stygia* (referred to as *C. rangiferina* henceforth), *C. stellaris*, and *Cladonia uncialis* to be primary forage species and *Cetraria cucullata*, *C. ericetorum*, *C. islandica*, *C. nivalis*, *Cladonia amaurocraea* and *C. gracilis* to be secondary forage species (Joly *et al.*, 2007c). For each primary and secondary caribou forage lichen that was encountered, we measured its height to the nearest 0.5 cm using a blunt metal rod (3 mm ∅; Moen *et al.*, 2007; also see Johnson *et al.* 2001). Total forage lichen is the combination of both primary and secondary lichens; all other lichen species were considered non-forage. Following Moen *et al.* (2007), we determined lichen volume by multiplying percent cover by height by plot area. Lichen volume is highly correlated with lichen biomass, so we used the formulas provided Moen *et al.* (2007) to approximate lichen biomass from its volume. We did not develop correction factors for our study area, so the biomass estimates should be viewed cautiously. Additionally, we collected data on near-surface soil pH, depth of the organic layer and amount of caribou utilization. Utilization was determined by identifying signs of use (e.g., pellets, signs of cratering, trampling) within a 1-m arc at the end of each grid row (*sensu* Joly *et al.*, 2007c); recorded as the percentage of arcs with evidence.

We used preexisting GIS data and attributed each plot with its slope, aspect, elevation, habitat type (30-m National Land Cover Database of 2001, data available from the Multi-Resolution Land Characteristics Consortium, www.mrlc.gov, accessed October 5, 2009), and 2 measures of terrain ruggedness. These measures were determined using a vector ruggedness measure (VRM) at relatively fine (180 m) and coarse (1 km) scales (Sappington *et al.*, 2007). For plots classified as burned, we used the Large Fire Database to determine stand age. We subjectively conferred a conservative age of 100 years to unburned plots for regression analyses and display purposes. Although fire rotation can be as short as 40-80 years in the boreal forest under the influence of the continental climate of eastern Alaska (Yarie, 1981; Johnstone *et al.*, 2009), it is typically longer (100-200 years; Payette *et al.*, 1989; Lynch *et al.*, 2003; Kasischke *et al.*, 2006). Rotation period can be several hundreds of years in the tundra biome (Wein, 1976; Kasischke *et al.*, 2006). Mark *et al.* (1985) estimated the ages of unburned tussocks to be 150 years.

We used analysis of variance (ANOVA) to test for differences among treatment types, burned status, and winter ranges (i.e., current, potential and historic). We used linear regression to identify associations with plot lichen volume. Combinations of vegetation characteristics (herbaceous cover, moss cover, shrub cover, canopy cover), plot characteristics (latitude, longitude, pH, stand age) and GIS layers (elevation, slope, terrain ruggedness) were used, *a priori*, to develop potentially explanatory models. We used Akaike's Information Criteria (AIC_c) for small sample sizes and the change in AIC_c ($_\Delta AIC_c$; Burnham & Anderson, 2002) to determine the most parsimonious models. We calculate the variance inflation factor (VIF) to assess multicollinearity. We standardize the values of the covariates by subtracting their means and dividing by their standard deviations in order to show the relative influence of these factors (Neter *et al.*, 1996; Schimdt *et al.*, 2009). Significance for all tests was defined at $\alpha = 0.050$ level unless otherwise specified.

Results

REASSIGNMENT OF PLOT TREATMENT TYPE

Ground observation of plots indicated that only 6 of 15 (40.0 %) plots used by caribou within the mapped fire perimeters (T2) had actually burned. A total of 12 of 32 (37.5 %) of the satellite location-derived points were changed to a different plot type based on ground observations. Total forage lichen volume in plots reassigned to T1 (used/unburned) (1037.2 ± 256.4 dm^3 / 100 m^2) was significantly greater than those that remained as T2 (used/burned) plots (125.0 ± 296.1 dm^3 / 100 m^2; $F_{1,13} = 5.42$, $P = 0.038$). Total forage lichen volumes in the remaining T2 (used/burned) plots were not significantly different than T4 plots (random/burned; $P > 0.100$).

A full 60 % of plots determined by satellite telemetry to be within a burn but used by caribou did not fit this categorization. Part of this error rate is undoubtedly due to lack of precision in the large fire database, but it strongly highlights the need for much more accurate caribou locations. All 6 remaining T2 (used/burned) were based on caribou locations from outside the mid-winter (i.e., December through mid-April) period.

USED VERSUS RANDOM UNBURNED PLOTS WITHIN THE CURRENT WINTER RANGE

Total forage lichen volume was significantly greater in locations used by caribou that were unburned (1427.3 ± 194.0 dm^3 / 100 m^2) than at randomly located unburned sites (447.8 ± 206.2 dm^3 / 100 m^2; $F_{1,48} = 11.97$, $P = 0.001$). In addition to having greater cover of forage lichens, areas in the current winter range used by caribou had lower cover of moss, forbs, and shrubs compared to random points (Table 3.1).

UNBURNED VERSUS BURNED PLOTS

In the current winter range, using both used and random plots, total forage lichen volume was significantly ($F_{1,70} = 9.79$, $P = 0.003$) greater in unburned plots (967.52 ± 133.43 dm^3 / 100 m^2) than on burned plots (217.49 ± 1991.3 dm^3 / 100 m^2). Forage lichen cover was also significantly ($F_{1,70} = 11.97$, $P = 0.001$) greater in unburned plots (23.3 ± 2.9 %) than on burned plots (5.1 ± 4.4 %). We did not detect any other significant vegetation differences between burned and unburned plots. The average age of burned plots was 37 ± 1.7 years. Though total forage lichen volume was nearly twice as much in random unburned plots (447.79 ± 136.84 dm^3 / 100 m^2) than in random burned plots (252.18 ± 164.06 dm^3 / 100 m^2) within the current winter range, the difference was not statistically significant ($F_{1,38} = 0.84$, $P = 0.366$).

Forage lichen cover and volume in the potential winter range were not significantly different between random unburned and burned locations. Dwarf birch (*Betula* spp.) cover in the shrub layer was significantly greater ($F_{1,22} = 8.45$, $P = 0.008$) in burned (8.1 ± 1.5 %) than in unburned plots (1.5 ± 1.7 %). Tree ($F_{1,22} = 5.41$, $P = 0.030$) and total ($F_{1,22} = 8.01$, $P = 0.010$) canopy cover were significantly greater in unburned plots (24.5 ± 6.7 %, 31.9 ± 7.1 %, respectively) than burned plots (3.9 +/- 5.8 %, 5.2 ± 6.2 %, respectively) in the potential winter range. The average age of burned plots was 25 ± 3.4 years.

Forage lichen cover and volume were not significantly different between random unburned and burned locations in the historic winter range. Dwarf birch cover in the shrub layer was significantly greater ($F_{1,10} = 7.55$, $P = 0.023$) in burned (11.5 ± 2.4 %) than in unburned plots (3.1 ± 1.8 %) in the historic winter range, similar to the pattern found in the potential winter range. The average age of burned plots was 37 ± 0.1 years.

DIFFERENCES IN VEGETATION AMONG UNBURNED CURRENT, POTENTIAL AND HISTORIC WINTER RANGES

Although total forage lichen volume in the potential (274.97 ± 221.59 dm^3 / 100 m^2) and historic (108.77 ± 264.86 dm^3 / 100 m^2) winter ranges were only 62% and 24 %, respectively, of that of the current (447.79 ± 146.11 dm^3 / 100 m^2) winter range, the differences were not statistically significant ($F_{2,39} = 0.70$, $P = 0.505$) due to high variability. The same held true for forage lichen cover (Table 3.2). Small sample sizes (Table 3.2) likely hindered our statistical analyses. Although there were relatively few statistically significant differences in vegetation between the current winter range and either the historic or potential winter range, there were some ecologically important trends (Table 3.3). In the shrub layer, both dwarf birch and crowberry (*Empetrum nigrum*) had significantly ($P < 0.100$ for the latter) greater cover in the current winter range than in the potential or historic winter ranges, and total shrub cover was greater in the current winter range than in the historic winter range. In the canopy layer, there were no significant differences among the current, potential and historic winter ranges except for a lower tree canopy in the current than the potential winter range and greater tall shrub canopy cover in the current than the historic winter range (just as observed in the shrub layer). In general, these results indicate that the current winter range tended to have considerably more lichens than the historic or potential winter ranges and had significantly less tree cover than the potential winter range and more shrub cover than the historic winter range.

LICHEN USE BY CARIBOU

Total forage lichen volume was positively associated with signs of caribou use ($R^2 = 17.2$, $F = 21.44$, $df = 104$, $P < 0.001$). Only 4 of 53 plots (7.5 %) with forage lichen cover < 5 % showed any signs of utilization by caribou. Total forage lichen volume was negatively associated with latitude ($R^2 = 15.8$, $F = 19.30$, $df = 104$, $P < 0.001$). Primary and secondary lichen covers were negatively associated with latitude ($R^2 = 18.4$, $F = 23.20$, $df = 104$, $P < 0.001$ and $R^2 = 5.0$, $F = 5.44$, $df = 104$, $P = 0.022$, respectively) but non-forage lichen cover was not ($R^2 = 0.01$, $F = 0.59$, $df = 104$, $P = 0.444$).

ASSOCIATIONS WITH LICHEN ABUNDANCE

Stand age, moss cover, latitude, soil pH and herbaceous cover parameters were in all of the most parsimonious models ($_{\Delta}AIC_c < 4.0$) predicting lichen volume for classes containing primary forages lichens (i.e., total forage, primary forage, *Cladina rangiferina*, and *Cladina mitis*; Table 3.4). The most parsimonious model predicting *Cetraria cucullata* volume (the most common secondary forage lichen) included Stand age, moss cover, longitude, soil pH, canopy cover of shrubs, terrain ruggedness (1 km scale) and elevation parameters (Table 3.4). A complete listing of the candidate models is provided in Table 3.5. All VIF values were low (< 4.0) suggesting multicollinearity was not problematic. Using only preexisting covariates available as GIS layers, the best model explaining variability of forage lichen volume included latitude, longitude and stand age (Table 3.4). Elevation, terrain ruggedness (1 km scale) and slope were all significantly and positively associated (independently) with forage lichen volume. The relative influence of the covariates and directionality (positive or negative) of their associations are depicted in Fig. 3.3. Latitude had a relatively strong negative association with lichen abundance, while stand age showed the opposite relationship.

In relation to shrubs specifically, forage lichen volume was associated positively with crowberry and dwarf birch cover, even using just the data from unburned plots. Forage lichen volume was negatively associated with alder (*Alnus* spp.) and willow (*Salix* spp.) cover in the shrub layer and with the depth of the organic layer ($R^2 = 23.0$, $F = 5.24$, $df = 93$, $P < 0.001$). Forage lichen volume also varied among habitat classes (Table 3.6). Approximations of lichen biomass from the volumes we calculated, as well others we found in the literature, are listed in Table 3.7. Lichen biomass at caribou locations were much greater than found at random locations within Alaska but still much lower than climax lichen communities in Quebec and Scandinavia.

While increasing lichen abundance was associated with stand age (Fig. 3.4), there were a few somewhat anomalous study plots of interest. Plots #72 and #122, which burned in 1957 (stand age = 51 years) had 74 % and 45 % lichen cover, 31 % and 36 % of which were forage lichens, respectively. On the sandy soils of plot #130, which burned in 1972 (stand age = 36 years), lichen cover reached 50 % with 31 % forage (predominately *Cladina mitis*) lichen cover. Aspen (*Populus tremuloides*) was quickly colonizing this area and so the high lichen cover may be a short-lived phase. Finally, plot

#125, which burned in 1984 (stand age = 24 years) had a lichen cover of 42 % (primarily *Cladonia* spp.), but with only 4 % forage lichen cover.

Discussion

Lichens are a critical component of the winter diet of caribou in large, migratory herds with high energy expenditure for predation avoidance and migration, such as the WAH. The abundance of lichens can affect caribou body condition, which in turn can have multiplier effects that affect herd population dynamics (White, 1983; Parker *et al.*, 2009). WAH caribou selected sites that had more than 3 times greater lichen abundance (both the percent cover and volume) of terricolous forage lichens than was found at random, unburned locations within the herd's current winter range. Additionally, locations used by caribou had lower moss, forb, shrub, and canopy cover than these random locations. Caribou utilized habitats that had less cover of tall shrubs; it is unclear if this is because of the lower lichen abundance in these habitat types, deeper snow, increased risk of predation, or a combination of these factors.

Lichen abundance was greatest in the current winter range and lowest in the historic winter range. Moreover, we found much lower lichen abundance on the historic range than was found there less than 20 years ago (Table 3.7), though the two studies are not directly comparable because methodologies and study sites varied. Northern areas of the current winter range have endured sustained heavy grazing (Joly *et al.*, 2007c), which was followed by the expansion of the herd's winter range onto the Seward Peninsula around 1996. From these findings, we infer that range deterioration can lead to range shifts, which is congruous with other research (*e.g.*, Ferguson *et al.*, 2001). Our results reveal, however, that there are still large portions of current winter range with relatively high abundance of forage lichens.

We hypothesized that an indicator of severe overgrazing of winter range would be the lack of, or loose, association between forage lichen volume or biomass and lichen cover. Instead, we found that forage lichen volume was highly associated with forage lichen cover from which we infer that the winter range of the WAH has not been overgrazed. In addition, signs of caribou use were associated with forage lichen abundance, while areas containing less than 5 % cover of forage lichens had very low use, in agreement with other studies (Arseneault *et al.*, 1997; Joly *et al.*, 2007c). We found forage lichen cover to be 10.6 % and biomass to be 1260 kg/ha; this also suggests to us that the range is not overgrazed, but it does have substantially less lichen biomass than some *Rangifer* ranges (Table 3.7). Grazing and trampling by *Rangifer* are important factors in landscape-level reductions in lichen cover (Klein; 1968; Pegau, 1969; Moser *et al.*, 1979; Messier *et al.*, 1988; Arseneault *et al.*, 1997; Joly *et al.*, 2007c; Klein & Shulski, 2009). Forage lichen cover was reduced by a third in just 10 years in the northern portion of the WAH's current winter range (Joly *et al.*, 2007c); reductions of this order of magnitude could drive forage lichen abundance below desired thresholds in the current winter range (see Kumpula *et al.*, 2000). In view of the current low abundance of forage lichens in the historic winter range, the continuing damage to the lichens from trampling, associated loss of lichen biomass through foraging during the traditional semi-annual migration through the historic winter range, and reduced lichen growth due to climate warming, we expect that lichen cover and biomass will be slow to recover and this area will be used less as winter range than during the historic period (Kumpula *et al.*, 2000).

We hypothesized that associations between lichen abundance and latitude would be affected by caribou usage and/or climatic influences. Caribou can utilize northern portions of their winter range on both their fall and spring migration, thus these areas may be subject to greater usage. Since caribou cross these areas during migrations and are less energetically demanding to reach (Fancy & White, 1987), we should expect to see a negative relationship between forage lichen abundance and latitude if caribou usage was affecting lichen abundance. If lichens declined universally from south to north, this would be suggestive of climatic factors. Forage lichen abundance was negatively associated with latitude (Fig. 3.3) but non-forage cover was not significantly associated. While total lichen cover (12.5 %) was similar between an area of concentrated use in the northern portion of the current winter range (Joly *et al.*, 2007c) and random unburned plots in our study (14.8 %), forage lichen cover was less than half in this northern area (4.6 %) versus the present study (10.6 %). These results suggest caribou usage is negatively impacting lichen abundance – though climatic factors may as well. This therefore provides evidence that large, migratory herds of barren-ground caribou both affect and respond to lichen availability (also see Moser *et al.*, 1979; Arsenault *et al.* 1997).

The abundance of forage lichens was also negatively associated with soil pH, and the cover of moss, herbaceous layer and shrubs in the canopy. The negative association between lichen abundance and vascular plant species has been reported in other studies (Klein, 1987; Swanson, 1996; Cornelissen *et al.*, 2001; Graglia *et al.*, 2001; Joly *et al.*, 2007c; Holt *et al.*, 2008, 2009). Taller physiognomy shrubs with greater leaf biomass (e.g., alders and willows) appear to inhibit lichens through shading and smothering (from the shrub leaf litter) of lichens. Another indirect, negative impact is that these taller shrubs can alter patterns of snow accumulation and delay snow melt, which could further enhance the competitive advantage of vascular species (Sturm *et al.*, 2005; Forbes *et al.*, 2010) at the expense of lichens. In contrast, covers of crowberry and dwarf birch were positively associated with lichen abundance. This relationship was the same using just unburned plots and thus is likely not an artifact of both being positively associated with time since last burn. The negative association with pH was also anticipated as acidic soils restrict vascular plants that compete with lichens (Swanson, 1996; Holt *et al.*, 2007, 2009). However, our finding of a negative association with moss cover contrasts with that of Holt *et al.* (2008) on the Seward Peninsula. Our sampling universe covered a much greater range of habitat types which may explain this discrepancy. Plots with well-drained soils, which are uncommon on the Seward Peninsula, can have high lichen but low moss abundance resulting in the association that we found. The disparity may also be due to differences in methodology. Because we could only determine the ages of recently burned plots, we are not able to address the potential regenerative role of fire in lichen stands in the long-term (e.g., 200 years) and whether mosses may outcompete lichen on this time horizon as occurs elsewhere (Sulyma & Coxson, 2001).

Forage lichen abundance was positively associated with stand age, elevation, coarse scale terrain ruggedness and slope. Lichen abundance was very low for at least 30-35 years following fire, which is in agreement with other studies from this region (Swanson, 1996; Racine *et al.*, 2004; Holt *et al.*, 2006, 2008; Jandt *et al.*, 2008). Areas of low elevation, topographic complexity and slope tend to be associated with large riparian habitats. Terricolous lichens do not compete well in these habitats due to competition with vascular plants, smothering by leaf litter and burial by sediments (Swanson, 1996; Holt *et al.*, 2008). The positive association with elevation, terrain complexity and slope will have limits; as they get too great, exposure, unstable soils, and avalanches can reduce lichen abundance (Swanson, 1996; Holt *et al.*, 2008). Most of our plots were found at intermediate elevations, topographic complexity, and slopes and thus did not approach these upper limits that would restrict forage lichen abundance.

The abundance of forage lichens was more than 4 times greater in unburned plots as compared to burned plots within the current winter range. Limited differences, aside from lichen abundance, between burned and unburned plots support the hypothesis that caribou avoid burned habitat in this region because of limited availability of forage lichens during winter (Joly *et al.*, 2007a). Additionally, many caribou locations initially thought to be within burned areas were actually outside actual burn perimeters or were within unburned inclusions within the larger fire perimeter. Islands of unburned habitat may also be attractive feeding sites for caribou (Miller, 2000). The large proportion of T2 (used/burned) plots that had to be reassigned to T1 (used/unburned) plots suggests that the avoidance of burned habitat in winter by caribou in this region is likely much greater than previously reported. All T2 (used/burned) plots that were not reassigned were all based on caribou locations during more migratory (e.g., October or late April) periods.

Traditional satellite and GPS-based telemetry systems use different methodologies and satellite platforms to determine locations, with accuracies of approximately 500 m (Fancy *et al.,* 1989) for traditional satellites and 30 m (Joly, 2005) for GPS telemetry in caribou studies. We recommend transitioning from traditional satellite to GPS telemetry technology for determining caribou locations in habitat use studies due to the high percentage of misclassification of plot treatment type (i.e., the satellite location indicated it was not in a burn but ground-truthing revealed it clearly was in a burn).

Differences between forage lichen abundance at burned and unburned plots were not detected within the historic and potential winter ranges likely due to caribou reducing lichens on unburned plots in the historic range to the point that differences between grazing and burning impacts could not be statistically differentiated and that other factors (e.g., more tall shrubs) might have made unburned habitats less suitable for lichens in the potential winter range. Indeed, we detected few significant differences between burned and unburned plots in these areas which is likely related to the ability of vascular vegetation to recover quickly after fire (average stand age of burned areas was 37 and 25 years in the historic and potential winter ranges, respectively), though small samples sizes may have hampered ability to detect differences. We were able to determine that cover of dwarf birch was greater at burned locations that at unburned locations in both the historic and potential winter ranges. Due to its adaptations to fire, increased abundance of dwarf birch following a wildfire should be expected (de Groot & Wein, 1999). Furthermore, dwarf birch is predicted to increase in abundance and distribution under climate warming scenarios (de Groot & Wein, 1999; Bret-Harte *et al.*, 2001; Euskirchen *et al.*, 2009) and could enhance a positive feedback mechanism that would further increase wildfire in tundra ecosystems (Higuera *et al.*, 2008, 2009).

We believe that the likelihood of the WAH expanding its range into the potential range is low for many reasons. First, biomass of forage lichens is low. Although there are peat plateau habitats in this region that support high lichen abundance, their extent is limited and present only on the southeast side of the Koyukuk River (Fig. 3.2). This large, meandering river creates vast riparian habitats with low biomass of forage lichens, and they support high densities of moose (*Alces alces*) which could facilitate the maintenance of year-round high densities of wolves (*Canis lupus*), the primary predator of caribou during winter. The low elevation hills to the west of the Koyukuk River are blanketed with habitat types that caribou avoid in winter (e.g., deciduous forests, alder and willow thickets; this study). Tree cover in the potential winter range (25 %) was much greater than in the current (3 %) or historic (5 %) winter ranges. The extent and frequency of fires is also much greater in the potential winter range than the current winter range (Joly *et al.*, 2009b). So the herd would have to cross wide expanses of poor-quality winter range with potentially high predator densities to reach the spatially limited but higher quality peat plateau habitats. Thus, we infer it is unlikely that the herd will utilize the Koyukuk country as winter range regularly or in large numbers.

Large, migratory herds of caribou seek out winter range with abundant lichen biomass. These herds can substantially affect this resource. Once depleted, caribou may expand or shift their distribution to find new areas with high lichen abundance. The additional energetic expense of migrating further, combined with additional predation risk, may be detrimental to caribou populations. Moreover, recovery of depleted winter ranges may take decades (Henry & Gunn, 1990). This recovery period may be extended due to changes in climate (Joly *et al.*, 2009a; Klein & Shulski 2009). Increased wildfire activity and shrub abundance combined with expansion of deciduous forests, all of which are predicted under climate change scenarios, will further retard lichen growth. This may in turn negatively impact caribou and the subsistence users that rely upon this critical resource.

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Table 3.1. Differences in vegetation between locations used by caribou that were unburned and randomly selected unburned locations within the current winter range of the Western Arctic Caribou Herd, western Alaska, 2008.

	Caribou locations		Random locations		
Vegetation (layer)	% Cover	SE	% Cover	SE	$F_{1,48}$
Forage lichen (ground)	34.6	4.1	10.6	4.4	16.00 **
Moss (ground)	25.3	4.4	38.0	4.7	3.81 *
Forb (herb)	8.8	3.4	22.0	3.6	7.03 **
Graminoid (herb)	22.5	3.5	16.5	3.7	1.37
Alder (shrub)	0.2	0.5	2.0	0.5	6.35 **
Tall shrubs (shrub)	27.7	3.9	43.5	4.2	7.67 **
Willow (shrub)	1.9	2.6	12.2	2.8	7.28 **
Alder (canopy)	0.1	1.4	4.3	1.4	4.60 **
Willow (canopy)	0.3	2.3	11.2	2.5	10.33 **
Total shrubs (canopy)	0.7	2.7	17.3	2.9	18.06 **
Total tree (canopy)	1.0	0.8	2.6	0.8	1.86
Total (canopy)	1.7	2.9	19.9	3.1	18.70**

* $P < 0.100$

** $P < 0.050$

Table 3.2. Percent lichen cover by treatment type among winter ranges of the Western Arctic Caribou Herd, western Alaska, 2008. Total cover includes all types of lichens whereas forage lichen just includes primary and secondary forage species. "Used" notation under "Treatment" represents a plot that a caribou was located as determined by satellite telemetry data. "Burned" means that the location recently (< 58 years) burned.

Winter Range	Treatment	N	Total (%)	SE	Forage (%)	SE
Current	T1 – Used/Unburned	26	41.1	26.7	34.6	3.1
Current	T2 – Used/Burned	6	7.5	10.1	3.5	6.6
Current	T3 –	23	14.8	19.9	10.6	3.3
Current	T4 – Random/Burned	16	16.3	19.0	5.8	4.0
Potential	T5– Random/Unburned	10	7.6	13.7	4.9	5.1
Potential	T6 – Random/Burned	13	18.8	18.6	7.5	4.5
Historic	T7– Random/Unburned	7	6.4	5.9	3.7	6.1
Historic	T8 – Random/Burned	4	13.5	7.3	4.5	8.0

Table 3.3. Relative differences in vegetation among random unburned locations within the current (C), potential (P), and historic (H) winter ranges of the Western Arctic Caribou Herd, western Alaska, 2008. Significance level of $P < 0.100$ denoted by "*" and $P < 0.050$ by "**".

	Current		Potential		Historic			
Vegetation (layer)	% Cover	SE	% Cover	SE	% Cover	SE	$F_{2,39}$	Relative Differences
Dwarf birch (shrub)	9.5	1.4	1.5	2.1	3.1	2.5	6.27	C > P, H **
Crowberry (shrub)	3.0	0.7	0.2	1.1	0.6	1.3	3.23	C > P, H*
All shrubs (shrub)	57.5	4.6	52.1	6.9	31.1	8.3	3.87	C > H **
Broadleaf trees (canopy)	0.7	3.3	15.2	5.0	0.0	0.0	3.20	P > C, H*
Conifer trees (canopy)	1.9	1.9	9.3	2.9	5.4	3.5	2.27	
Tall shrubs (canopy)	17.3	3.3	7.4	5.1	1.0	6.1	3.27	C > H **
All trees (canopy)	2.6	3.4	24.5	5.2	5.4	6.3	6.21	P > C **

Table 3.4. The most parsimonious models ($_{\Delta}\mathrm{AIC}_{c} < 4.0$) predicting volume of different lichen classes within the winter range of the Western Arctic Caribou Herd, western Alaska, 2008. Abbreviations for the model parameters are; “Age” for stand age, “Moss” for moss cover, “Lat” for latitude, “Long” for longitude, “pH” for soil pH, “Herb” for total herbaceous cover, “Shrub” for shrub cover in the canopy layer, “A” for alder in the shrub layer, “VRM180” for terrain ruggedness at the 180 m scale, “VRM1K” for terrain ruggedness at the 1 km scale, and “Elev” for elevation. “N” is the sample size used in the analysis, “K” is the number of degrees of freedom in the model, and “w_i” is the Akaike weight of the model.

Lichen Class	Model Parameters	N	K	Adj. R^2	$_{\Delta}\mathrm{AIC}_{c}$	w_i
Total forage	Age, Moss, Lat, Long, pH, Herb, Shrub	104	8	53.4	0.0	0.33
	Age, Moss, Lat, pH, Herb, A	104	7	52.7	0.1	0.31
	Age, Moss, Lat, pH, Herb, A, VRM180	104	8	52.7	1.6	0.15
	Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	104	9	53.1	1.9	0.13
	Age, Moss, Lat, Long, pH, Herb, VRM180	104	8	52.1	2.9	0.08
Primary forage	Age, Moss, Lat, pH, Herb, A	104	7	51.8	0.0	0.39
	Age, Moss, Lat, pH, Herb, A, VRM180	104	8	52.9	1.7	0.17
	Age, Moss, Lat, Long, pH, Herb, VRM180	104	8	51.6	1.8	0.16
	Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	104	9	52.2	1.9	0.15
	Age, Moss, Lat, Long, pH, Herb, Shrub	104	8	51.5	2.0	0.14
Secondary forage	Age, Moss, Long, pH, Shrub, VRM1K, Elev	104	8	37.5	0.0	0.34
	Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	104	9	37.9	0.7	0.24
	Age, Moss, Lat, Long, pH, Herb, Shrub	104	8	37.1	0.7	0.24

Table 3.4 continued.

Lichen Class	Model Parameters	N	K	Adj. R^2	$_\Delta$AIC$_c$	w_i
Secondary forage	Age, Moss, Lat, Long, pH, Herb, VRM180	104	8	35.7	3.0	0.08
	Age, Moss, Lat, pH, Herb, A	104	7	34.5	3.6	0.06
Cladina rangiferina	Age, Moss, Lat, Long, pH, Herb, VRM180	104	8	38.7	0.0	0.34
	Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	104	9	39.1	0.7	0.24
	Age, Moss, Lat, Long, pH, Herb, Shrub	104	8	37.9	1.2	0.19
	Age, Moss, Lat, Long, pH, Herb	104	7	36.6	1.9	.013
	Age, Moss, Lat, pH, Herb, A, VRM180	104	8	38.6	3.4	0.06
Cladina mitis	Age, Moss, Lat, Long, pH, Herb, VRM180	104	8	44.4	0.0	0.49
	Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	104	9	44.3	1.4	0.24
	Age, Moss, Lat, pH, Herb, A,	104	7	42.5	2.1	0.17
Cetraria cucullata	Age, Moss, Long, pH, Shrub, VRM1K, Elev	104	8	30.7	0.0	0.79
Total forage[1]	Age, Lat, Long	105	4	25.7	0.0	0.26
	Age, Lat, Long, Slope	105	5	26.3	0.4	0.21
	Age, Lat, Slope	105	4	24.7	1.46	0.12
	Age, Lat, Long, Elev	105	5	25.4	1.59	0.12
	Age, Lat, Long, VRM180, Slope	105	6	25.8	2.32	0.08
	Age, Lat, Long, Elev, Slope	105	6	25.6	2.53	0.07
	Age, Lat, Long, VRM1K, Slope	105	6	25.6	2.62	0.07
	Age, Lat	105	3	22.7	3.01	0.06

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[1] Using only covariates that are available in existing GIS databases

Table 3.5. List of candidate models and number (K) of parameters it uses. Abbreviations for the model parameters are; "Age" for stand age, "Moss" for moss cover, "Lat" for latitude, "Long" for longitude, "pH" for soil pH, "Herb" for total herbaceous cover, "Shrub" for shrub cover in the canopy layer, "A", "Db", "W" for alder, dwarf birch, and willow (respectively) in the shrub layer, "VRM180" for terrain ruggedness at the 180 m scale, "VRM1K" for terrain ruggedness at the 1 km scale, and "Elev" for elevation.

Model	K
Age, Moss, Lat, pH, Herb, Shrub, VRM180, Elev	8
pH, Herb, Shrub, A, Db, W, VRM1K	7
Lat, Long, pH, Herb, Shrub, VRM180, Elev	7
Age, Moss, Lat, Long, pH, Herb, Shrub	7
Age, Moss, Lat, pH, Herb, A, VRM180	7
Age, Moss, Long, A, Db, W, Elev	7
Age, Moss, Lat, Long, pH, Herb, VRM180	7
Age, Moss, Long, pH, Shrub, VRM1K, Elev	7
Age, Moss, Lat, pH, Herb, A	6
Age, pH, Herb, Shrub, VRM180, Elev	6
Age, Moss, Lat, Long, pH	5
Age, Moss, Lat, Long	4
Age, pH, Shrub	3
Moss, Herb, Shrub	3
Age	1

Table 3.6. Lichen volume (dm^3 / 100 m^2) per plot by habitat type (from the National Land Cover Map of 2001).

Habitat	N	Volume	SE
Sedge/Herb	5	1461.1	375
Dwarf Scrub	41	892.5	131
Shrub/Scrub	30	365.5	153
Woody Wetlands	12	278.6	242
Herb Wetlands	3	130.0	484
Evergreen Forest	10	113.5	265
Grassland	1	60.0	0
Deciduous Forest	1	0.0	0
Mixed Forest	2	0.0	0

Table 3.7. Comparison of lichen biomass (kg/ha) among different ranges.

Study area	Range	Plot Type	Biomass
Northwest Alaska	Current	Used by caribou (Unburned)	3007
	Current	Random (Unburned)	1260
	Current	Random (Burned)	818
	Historic	Random (Unburned)	435
	Potential	Random (Unburned)	873
Northwest Alaska	Historic	Used by caribou (Unburned)	1400
East Alaska	Current	Used by caribou	1045
	Current	Random	414
	Historic	Random	224
Quebec			1223
Quebec		< 30 yrs old	530
Quebec		> 90 yrs old	8010
Finland		Ungrazed pasture	8000
Norway		Climax stands	11000

Reference
this study
this study
this study
this study
this study
Saperstein 1993
Collins *et al.*, *in press*
Collins *et al.*, *in press*
Collins *et al.*, *in press*
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Arseneault *et al.,* 1997
Väre *et al.,* 1996
Gaare and Skogland 1980

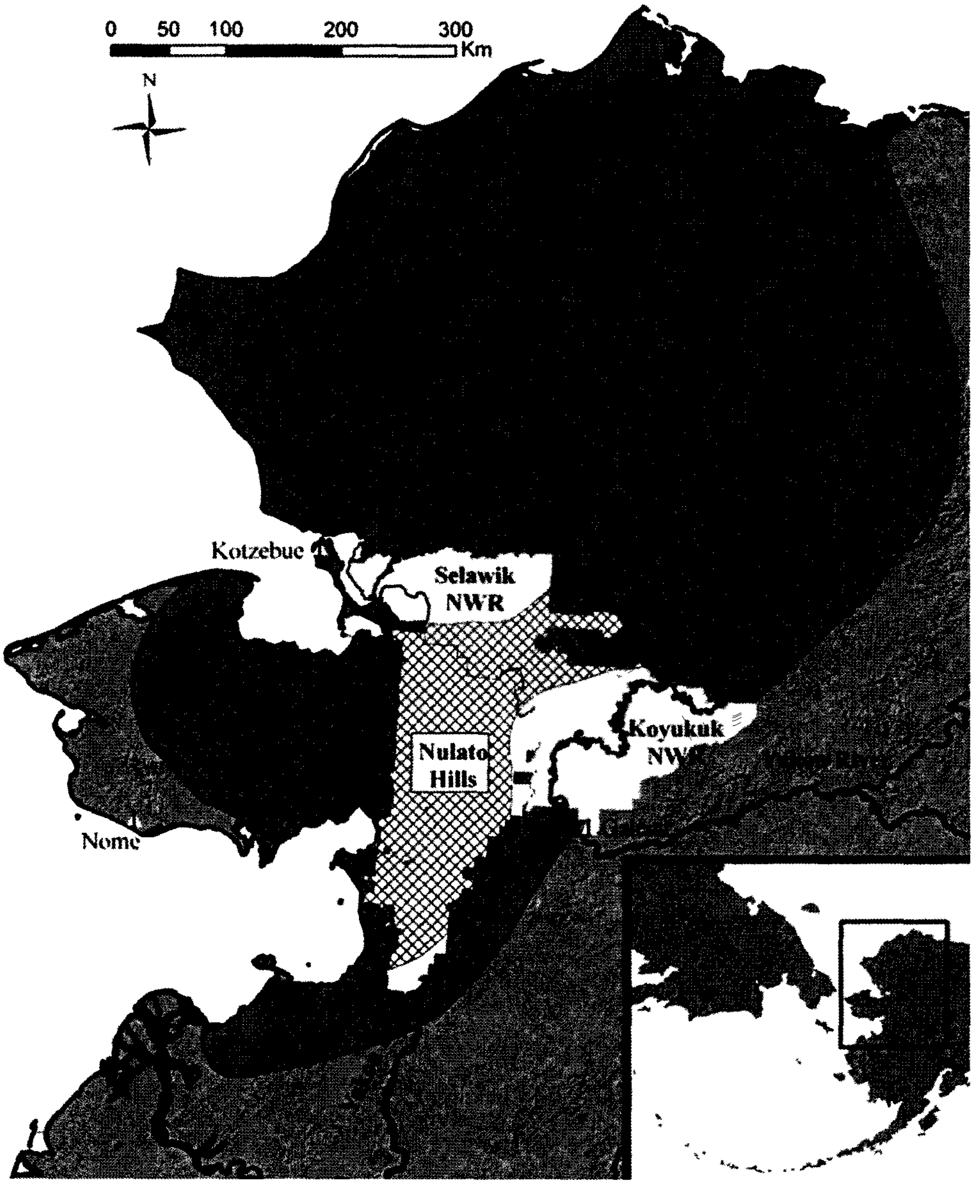


Fig. 3.1. Range of the Western Arctic Caribou Herd, northwest Alaska, 2008. Annual range is shaded dark while the current winter range is hatched, courtesy of the Alaska Department of Fish and Game.

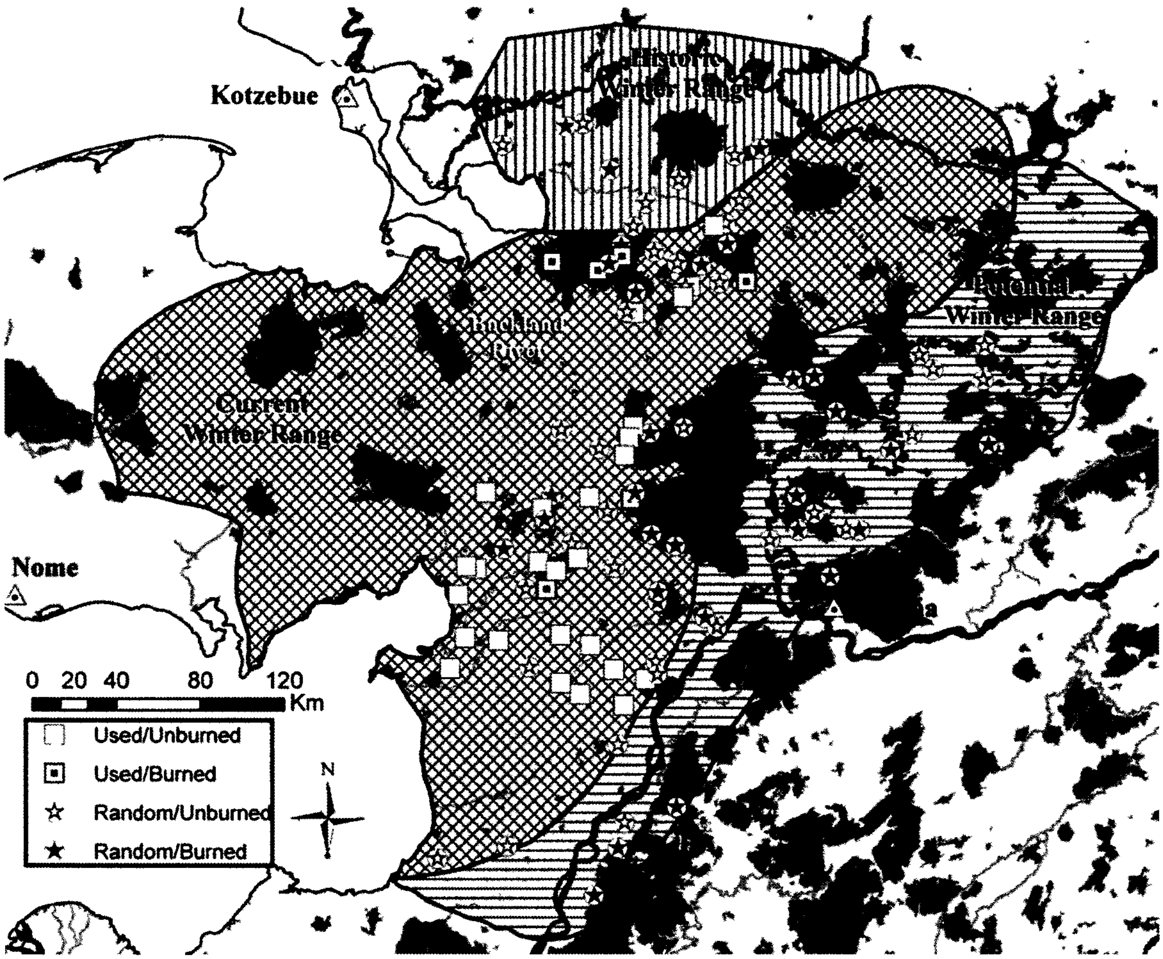
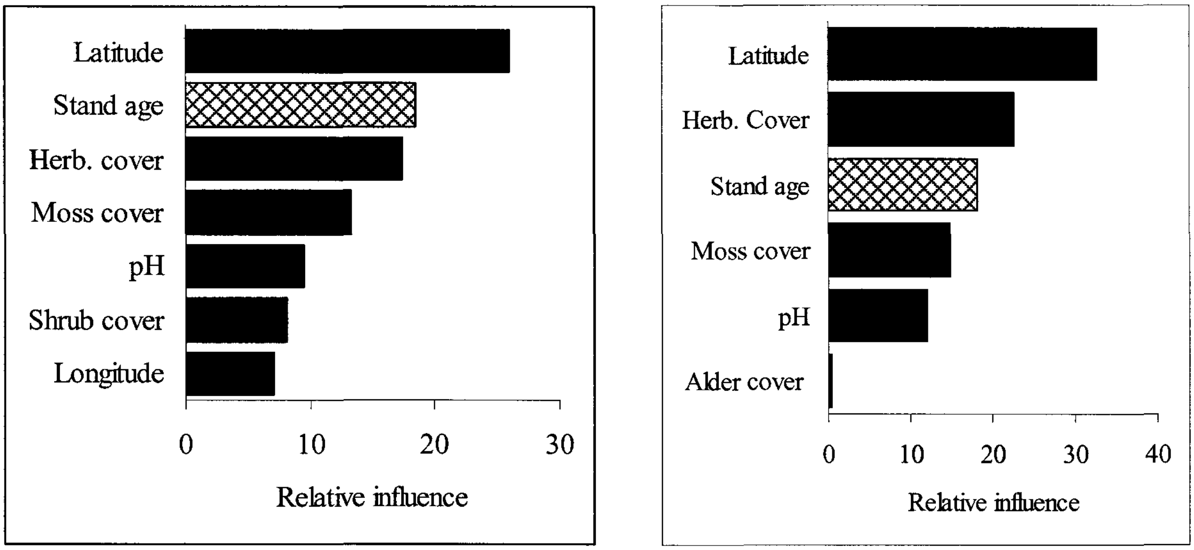


Fig. 3.2. Study plot locations and winter ranges of the Western Arctic Caribou Herd, northwest Alaska. Plots designated as "Used" were determined by satellite telemetry locations of collared caribou. Plots designated as "Burned" did so recently (< 58 years). The darkly shaded polygons are the outlines of these recent burns (courtesy of the Alaska Fire Service). The current winter range is hatched, the historic winter range is identified by vertical bars, and potential winter range (an area examined to see if caribou might expand their winter range there in the future) is identified by horizontal bars.


A. Total forage lichens

B. Primary forage lichens

C. Secondary forage lichens

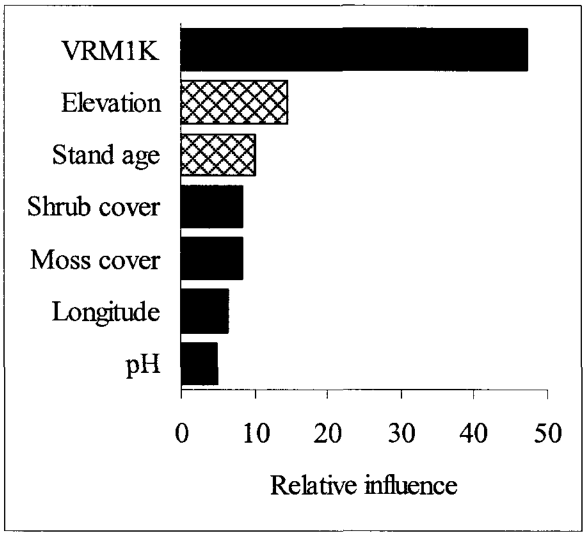


Fig. 3.3. Relative influence of covariates on modeling lichen abundance in Western Arctic Caribou Herd winter range, northwest Alaska, 2008. Regression coefficients standardize by subtracting their means and dividing by their standard deviations. Dark bars indicate a negative association while hatched bars indicate a positive association.

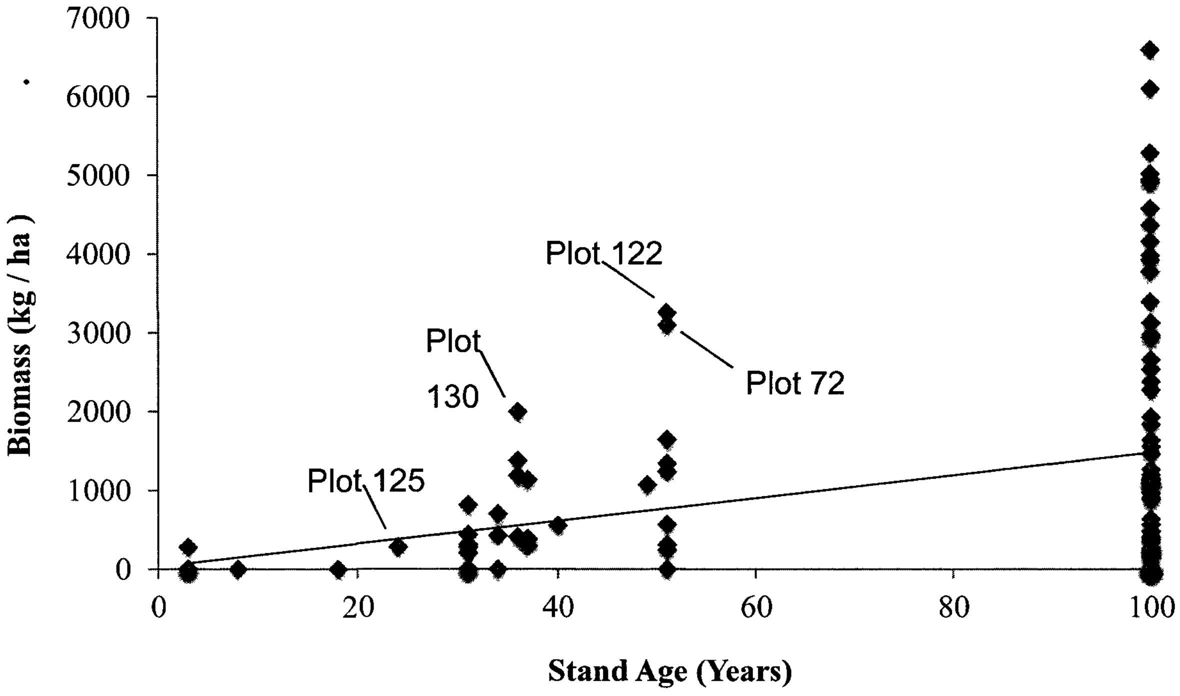


Fig. 3.4. Forage lichen abundance versus stand age in the winter range of the Western Arctic Caribou Herd, northwest Alaska, 2008. Plots determined to be "unburned" were subjectively assigned a stand age of 100 years (see Methods).

CHAPTER 4:

Simulating the effects of climate change on wildfire regimes in Arctic biomes: implications for caribou and moose[1]

ABSTRACT

Wildfire is the primary ecological driver of succession in the boreal forest and may become increasingly important within tundra ecosystems. Migratory barren-ground caribou (*Rangifer tarandus granti*) rely heavily on terricolous lichens to sustain them through the winter months. Lichens preferred by caribou can take 50 or more years to recover after being consumed by wildfires. We simulated effects of climate change on the fire regime within the winter range of the one of the largest caribou herds in the world, the Western Arctic Herd, to assess how forage quantity and quality may be impacted. We forecast that the total area burned (AB) in the near term (2008-2053) will be 0-30% greater than during our historic reference period (1950-2007) depending on the climate scenario (CGCM3.1 or ECHAM5) considered. Further into the future (i.e., 2054-2099), we forecast AB to increase 25-53% more than during our reference period. In contrast to the entire study area, which contains both tundra and boreal forest habitats, we forecast that the amount of AB in tundra alone will increase (0-61%) in the near term.

[1] To be submitted to *Ecological Modelling* as Joly, K., Duffy, P. A., Rupp, T. S., Simulating the effects of climate change on wildfire regimes in Arctic biomes: implications for caribou and moose

Simulated high-quality caribou winter range, as indexed by tundra and spruce habitat that had not burned in ≥ 50 years, decreased modestly (< 6%) in the near term over the entire study area. Simulated changes were more dramatic within the herd's core winter range, with near-term declines in high-quality caribou winter range of 3-16%. Further into the future, these declines ranged from 15-29%. Conversely, moose habitat was projected to increase by 19-64% within the core winter range in the near term. The simulated declines in the quality and quantity of winter range forage in the future due to larger and more frequent fires could impact caribou abundance through decreased nutritional performance and/or apparent competition with moose. These impacts would likely be detrimental to the subsistence users that rely on this resource. Additionally, changes in the fire regime and decreases in caribou abundance could amplify feedback mechanisms, such as decreasing albedo, by facilitating shrub growth that may hasten climate-driven changes to the composition and structure of vegetation communities in the low Arctic.

Introduction

Tundra covers over 5,000,000 km^2 of the Arctic (Walker et al., 2005). Additionally, vast areas of tundra are interspersed with boreal forest in the tundra-boreal forest interface that stretches around the globe north of the continuous boreal forest biome (Callaghan et al., 2002). The effects of climate change are already manifest (IPCC, 2007), especially in the Arctic (Callaghan et al., 2004). Mean annual temperatures have increased by 2-3° C in the region in recent decades, with larger increases apparent during the winter months (Stafford et al., 2000; Hinzman et al., 2005). Positive feedback mechanisms, including melting snow and sea ice, increasing shrubs, and greater prevalence of wildfires will likely amplify these changes via decreased albedo (Overpeck et al., 1997; Chapin et al., 2000; Qu and Hall, 2006; Balshi et al., 2009; Euskirchen et al., 2009).

Climate change impacts are predicted to be severe on the habitats of arctic land mammals (Lawler et al., 2009) and have already been implicated in the decline of caribou (*Rangifer tarandus*) populations across the Arctic (Vors and Boyce, 2009). However, the effects of climate change on caribou populations may not be uniform due to inherent complexities of the climate system (e.g., Zalatan, 2008; Joly et al., 2011). Furthermore, increased summer temperatures and longer growing seasons may prove to be nutritionally beneficial to caribou during the summer months (Jefferies et al., 1992; Griffith et al., 2002; Tews et al., 2007). Mechanisms by which climate change may negatively impact caribou include deeper snows, increased incidence of icing events, increased human disturbance, increased insect harassment, changes to habitat quality and quantity, and increased frequency of wildfire on winter ranges (Jefferies et al., 1992; Walsh et al., 1992; Griffith et al., 2002; Putkonen and Roe, 2003; Johnson et al., 2005; Rupp et al., 2006; Tews et al., 2007; Joly et al., 2011).

Climate warming in the Arctic could have several important consequences for its fire regime and caribou winter range. Warmer and drier summers predicted with climate change (Callaghan et al., 2004) are associated with increased burned area in boreal forests (Duffy et al., 2005; Balshi et al., 2009) and tundra (Joly et al., 2009b; Hu et al., 2010a). It is predicted that fire will increase in both of these biomes under climate warming scenarios (Flannigan et al., 2000; Higuera et al., 2008; Balshi et al., 2009). Wildfire is the dominant driver of change to ecosystem structure and function in the boreal forest (Payette, 1992; Chapin et al., 2006) but wildfires are less frequent and smaller in the tundra biome (Wein, 1976). However, the Seward Peninsula and Noatak River valley regions of Alaska (Figure 4.1) exhibit greater fire frequency than other areas within the tundra biome (Racine et al., 1985; Racine et al., 2004; Joly et al., 2009b; Hu et al., 2010a). While lichens are highly flammable, these regional differences are not likely due to lichen abundance because the Noatak region has relatively low lichen abundance, and graminoids provide fine fuels that increase the potential for reburning in tundra much sooner than feather mosses of the boreal forests (Cronan and Jandt, 2008; Jandt et al., 2008; Joly et al., 2009b).

Wildfires destroy terricolous fruticose lichens, a staple of the winter diet of migratory caribou (Klein, 1982; Bernhardt et al., 2011). Lichens are slow to recover after being burned, with primary caribou forage species often taking several decades or more to return to pre-fire levels (Holt et al., 2008; Jandt et al., 2008; Klein and Shulski, 2009; Joly et al., 2010; Collins et al., 2011). Caribou avoid burned areas in the boreal forest and tundra during the winter on a time scale coincident with lichen recovery rates (Joly et al., 2003, 2007a, 2010; Collins et al., 2011). Increases in fire frequency may limit the total available habitat for wintering caribou that is old enough to support high levels of preferred lichen species and tempers the argument that fire is not important to caribou because they can simply utilize alternative ranges (see Rupp et al., 2006). Further, the conclusions of Rupp et al. (2006) were likely conservative because they did not address the direct detrimental impacts climate change would have on lichens through warming and drying nor the indirect impacts on lichens by enhancing growing conditions for vascular plants such as shrubs and trees (see review by Joly et al., 2009a).

Warmer and longer growing seasons induced by climate change will likely degrade permafrost, deepen the active layer and provide vascular plants a competitive advantage over lichens (Chapin et al., 1995; Walker et al., 2006). Increases in shrub abundance have already been detected in the sub-Arctic and Arctic (Sturm et al., 2001; Joly et al., 2007c; Forbes et al., 2010). Shrubs may further inhibit re-generation of lichens by trapping snow, increasing leaf litter, competing for resources, and closing the canopy (Joly et al., 2009a; Čabrajić et al., 2010). Burn severity is an important factor that determines the successional trajectory after a fire (Racine et al., 2004; Johnstone et al., 2010; Berhardt et al., 2011). In more severe burns, stand replacement, which is common in Arctic ecosystems, is not predetermined. More shrubs, including fire-adapted dwarf birch (*Betula* spp.) are following fire, especially in moderate severity burns (de Groot and

Wein, 1999; Bret-Harte et al., 2001; Racine et al., 2004; Joly et al., 2010). This process could lead to an amplifying feedback where more shrubs lead to more and more severe fire (Higuera et al., 2009; Hu et al., 2010a, Xue et al., 2010). Similarly, greater burn severity could alter the black spruce (*Picea mariana*) self- replacement trajectory by consuming semi-serotinous seeds and reducing organic layers which results in an early successional deciduous phase (Johnstone et al., 2010). This would also be disadvantageous for overwintering caribou as they avoid these habitat types (Joly et al., 2010; Collins et al., 2011; Joly, in press). Likewise, additional shrub cover, such as *Salix* spp., could attract herbivores associated with boreal forests, such as moose (*Alces alces*; Bryant and Reichardt, 1992; Joly et al., 2010).

Unlike caribou in winter, moose select for areas with abundant early seral stage habitats (Weixelman et al., 1998; Maier et al., 2005). Thus, increased fire due to climate change may benefit this species. It has been hypothesized that more abundant moose may allow for increased wolf (*Canis lupus*) densities (i.e., a numeric response; James et al., 2004). Wolves are the main predator of caribou during winter months; thus, increases in moose densities due to climate change-induced higher fire frequency may indirectly lead to increased caribou predation by wolves (Latham et al., 2011). This would not impact neonatal survival as the herd's calving ground is north of the Brooks Range and far from the existing tree line.

Our goals were to identify the impacts of climate change on the fire regime in tundra-dominated landscapes of northwest Alaska. Using our simulations, we projected the amount and quality of caribou winter range available under climate warming scenarios. Additionally, we quantified how the amount of preferred moose habitat changed because this may have additional indirect impacts on caribou. We hypothesized that the fire regime in northwest Alaska in the next 50-100 years would be more intense (i.e., greater area burned (AB)) as compared to the current (1950-2007) regime. We hypothesized that the potential increase in AB in northwest Alaska would lead to decreased quality and quantity of caribou winter range as determined by the amount of available habitat over 50 years old (Joly et al., 2010; Collins et al., 2011), and to an increase of high-quality moose habitat as indexed by the amount of 10 to 30-year old deciduous vegetation (Weixelman et al., 1998; Maier et al., 2005).

Methods

Study area and wildlife

The study area is the range of the Western Arctic Herd (WAH) and the entire Seward Peninsula of northwest Alaska (Figure 4.1). The approximately 377,000 caribou in the WAH can be found distributed over some 360,000 km^2 (Dau, 2007). Overwintering caribou can be found throughout this range, though the areas of recent concentrated use have been in the Nulato Hills and Seward Peninsula (Joly et al., 2007a). While the region is dominated by arctic tundra, especially these areas of concentrated use, there are large expanses of tundra-boreal forest interface, boreal forest, alpine tundra, mountainous terrain and wetlands (Joly et al., 2010). Moose densities range from very low (~ 0.04/km^2) in the northwest portion of the study area to very high (~ 3/km^2) in southeast, but are low (~ $0.12/km^2$) over much of the study area (Alaska Department of Fish and Game (ADFG), 2008). The higher densities of moose correspond with expansive riparian complexes formed by the confluence of the large Yukon and Koyukuk Rivers.

Alaska Frame-Based Ecosystem Code (ALFRESCO)

The planet's climate system is inherently complex; how the climate impacts fire and vegetation at a local-scale is even more so. We used Alaska Frame-Based Ecosystem Code (ALFRESCO) to explore the interactions and feedbacks among fire, climate, vegetation, and caribou and moose habitat in northwest Alaska. ALFRESCO is a spatially-explicit cellular automata model that has simulated fire and successional dynamics in Alaska at a 1-km spatial resolution on a 1-year time step (Rupp et al., 2002). We modified the model to incorporate the tundra habitats of northwest Alaska (see Vegetation section below). ALFRESCO models the relationship between climate (i.e., monthly average temperature and total precipitation) and total annual area burned (i.e., the areal extent of fire on the landscape) rather than explicitly modeling fire behavior. We used a generalized boosting model (GBM) similar to that used by Hu et al. (2010b) to determine the effect of climate on the likelihood of cell ignition.

Annual fire occurrence, driven by climate, vegetation type and time since last fire, was simulated stochastically (Rupp et al., 2000b). The ignition of any given cell (pixel) is determined by comparing a randomly generated number against the flammability value of that cell. Fire may spread from an ignited cell to any of its eight surrounding neighbor cells. The burn algorithm in ALFRESCO employs a recursive cellular automaton approach, so fire spread depends on the flammability of cells in the first-order neighborhood and any effects of natural firebreaks including non-vegetated mountain slopes and large water bodies, which do not burn. The flammability coefficient is tied to vegetation type, so ALFRESCO allows for changes in flammability that occur through succession (Chapin et al., 2003). There are different fire regimes for different ecoregions (Joly et al., 2009b), therefore we subdivided our study area into 4 subregions (Interior, Seward Peninsula, North Slope and Yukon Lowlands) and assigned each a different relative flammability and probability of ignition. These values were determined by comparing model output to observed data during the calibration phase. Additional information regarding ALFRESCO can be obtained in existing literature (e.g., Rupp et al., 2000a, 2000b, 2002, 2006).

Vegetation

Our version of ALFRESCO reclassified the 1990 Advanced Very High Resolution Radiometer (AVHRR; agdcftp1.wr.usgs.gov/pub/projects/fhm/vegcls.tar.gz; accessed 07 June 2011) and 2001 National Land Cover Database (NLCD; www.mrlc.gov; accessed 07 June 2011) vegetation classification into three vegetation types: tundra, spruce, and deciduous vegetation. We categorized non-vegetated, non-flammable NLCD cover types (e.g., open water, perennial ice) as non-vegetated. Our tundra class contained dwarf scrub, grassland, sedge, lichen and moss NLCD cover types.

We categorized NLCD evergreen forests and woody wetlands (spruce bogs) as our spruce class. Our deciduous class contained NLCD deciduous and mixed forests categories. The shrub/scrub NLCD class was classified as either tundra or deciduous based on elevation, aspect and growing season temperature. White and black spruce have been differentiated using this technique in previous work (Rupp et al., 2002), but we did not for our simulations. Succession, occurring as either a transition from deciduous or spruce forest to early successional deciduous vegetation or the self-replacement of tundra, is initiated exclusively by fire (Rupp et al., 2000b). Conceptually, the deciduous vegetation type is an early successional stage of spruce forest. An exception to this successional trajectory can occur when repeated burning and/or climatic conditions preclude the transition from deciduous to spruce (Rupp et al., 2000a). We based transitional ages for deciduous to spruce succession from existing literature (Figure 4.2; Viereck et al., 1986; van Cleve et al., 1991; Rupp et al., 2006; Kurkowski et al., 2008) and observational data from the Joint Fire Science Program (JFSP; Duffy, 2006). Following fire in the boreal forest, flammability is low and stays low for several decades (Figure 4.2). Vascular tundra vegetation quickly re-sprouts after a fire and can be difficult to differentiate from nearby unburned tundra, with the exception of the absence of caribou forage lichens, after just a couple of years (Racine et al., 1985, 2004; Jandt et al., 2008, Joly et al., 2010). Thus, in a departure from previous modeling efforts, we developed a separate function for the flammability of tundra over time which reflects the speed with which fine fuels accumulate within the tundra (Figure 4.2). Successional transitions were determined stochastically in ALFRESCO (Rupp et al., 2002).

Climate Data

We employed the spatially explicit, global, gridded (0.5 degree by 0.5 degree) data provided by the Climatic Research Unit (CRU; Mitchell and Jones, 2005; http://badc.nerc.ac.uk/view/badc.nerc.ac.uk_ATOM_dataent_12562237733328276, accessed 07 June 2011). We extracted our study area (Figure 4.1) from this comprehensive dataset for the period 1901-2000. The dataset is comprised of the monthly averages for temperature and precipitation. We down-scaled the differences between CRU data and calculated CRU climate normals ("deltas") for 1961-1990 (Hay et al., 2000) using natural neighbor to 2 km^2 resolution and then added PRISM (Parameter-elevation Regression on Independent Slopes Model; http://www.prism.oregonstate.edu/, accessed 07 June 2011) climate normals. PRISM pixels were further interpolated to 1 km^2 using ArcGIS (ESRI) nearest neighbor re-sampling.

Fire and Fire Sizes

The amount of AB annually in Alaska is strongly correlated with monthly weather (average temperature and total precipitation; Duffy et al., 2005; Joly et al., 2009b). We developed a GBM to quantify the impact of monthly temperature and precipitation variables on annual area burned for our study region. The GBM is a non-spatial statistical model with annual AB as the response and the corresponding year's monthly temperature and precipitation values are the explanatory variables. The utility of the

GBM in this context is to provide a quantitative framework to inform the generation of spatially-explicit maps of flammability that account for the variation of temperature and precipitation across the simulation domain. The spatially-explicit maps of temperature and precipitation come from the PRISM downscaling of both CRU and GCM-base future climate scenarios. Quantifying the linkage between annual AB and monthly climate variables using this non-spatial approach is appropriate for large spatial domains where the climate signals conducive to fire are almost always sufficiently accompanied by lightning ignitions (Duffy, 2006). The climate-fire relationship quantified by this point model was then used as proxy for quantifying the relative flammability of pixels in spatially-explicit maps of annual landscape flammability. The climate-fire relationship is different for tundra and forest vegetation types; hence we used a different GBM-based functional linkage between climate and flammability for tundra and forest vegetation.

In order to calibrate our model we compared existing empirical data (1950-2007) to our simulated output. For our empirical fire perimeter data, we employed a spatially-explicit historical database maintained by the Alaska Fire Service (http://fire.ak.blm.gov/; accessed 16 February 2011). Estimates of AB based on these perimeters are presumed to be over-estimates due to the existence of unburned inclusions within fire perimeters (Duffy, 2006). Previous research has shown that the proportional size of inclusions increases with total fire size and fires > 2000 ha have approximately 5% inclusions (Eberhart and Woodard, 1987). Duffy (2006) determined the simulated AB should be roughly 5 % less than historical distribution based on that study and the examination of aerial photographs and remotely sensed imagery from interior Alaska. Comparable data do not exist for tundra ecosystems so we applied this factor to our analyses for northwest Alaska. The relative proportion of AB by fires of a given size is an important characteristic of fire distribution (Duffy, 2006). Our analysis of the historical database for our study area revealed that most fires are small (< 1000 ha) but the majority (> 56 %) of the area burned comes from a small percentage (< 5 %) of fires that are an order of magnitude larger (> 40,000 ha) than the rest. Our findings for northwest Alaska are similar to analyses conducted within the Interior region of Alaska (Kasischke et al., 2006). Therefore we focused our comparisons between historical and simulated fire size on the upper tail of the size distribution because the majority of the area burned occurs due to relatively few but large fires.

Model Spin-up and Validation

We developed 50 randomly-generated initial stand age/vegetation maps by running ALFRESCO for 400 years to approximate ecologically-realistic initial conditions with respect to the distribution and composition of vegetation (Duffy, 2006). This time period is approximately twice the duration of the longest reported fire return interval in interior Alaska (Yarie, 1981; van Cleve et al., 1991; Chapin et al., 2003). We assembled historical climate data for the model spin-ups consistent with Barber et al. (2004). We conducted different simulations from 1860-1949 using these initial age/vegetation maps and then subjected them to burning from 1950-2007, for comparison to historic, observed fire perimeter data, using the GBM-based flammability maps derived from historical climate. This process resulted in the generation of a final spin-up consisting of 50 different stand-age maps across northwest Alaska (Duffy, 2006). The spin-up phase therefore provided starting point conditions with realistic patch size and age-class distributions generated over multiple fire cycles that are used as input for simulations run into the future. We compared the simulated and empirical annual AB from 1950-2007 for each of the 50 replicate simulations generated by ALFRESCO (Figure 4.3). We used simple linear regression to assess the relationship between the average annual AB from the 50 replicate simulations to the historical data from 1950-2007. We selected the 5 most representative runs to seed the spin-ups for the 50 future replicate simulations based on vegetation type and ratios between modeled and observed data for AB for the years 2004 and 2005. These 2 years had climatic conditions that facilitated unrealistically large fire years in model simulation which could propagate landscape-level impacts throughout the simulation time period. The most representative runs were selected because they most closely represent realistic depictions of the current landscape based on observed fire activity. Among all of the stochasticity that is driven by random ignitions, burning, and patterns of vegetation succession, these maps best represented the starting point (2007) landscape for the future runs.

Projections with future climate scenarios

We extrapolated our models 92 years into the future (i.e., through the year 2099). This allowed us to compare the historic period (1950 - 2007), for which we have empirical data, to the next two 46-year periods (2008 – 2053 and 2054 – 2099). We imposed the intermediate A1B emissions scenario (Nakicenovic et al., 2000; IPCC 2007) on the ECHAM5 and CGCM3.1 general circulation models (GCMs) to provide a range that captures potentially high and conservative forecasts, respectively, for future climatically-driven fire activity (Walsh et al., 2008; Balshi et al., 2009).

Analyses

We calculated annual AB for the entire study area, as well as for both the tundra and boreal forest ecoregions separately, for the 3 time periods (1950-2007, 2008-2053, and 2054-2099) using R programming language scripts (www.r-project.org). Similarly, we calculated the areal extent of habitat in 1) deciduous stands 10-30 years old and 2) tundra and spruce forest ≥ 50 years old. WAH caribou utilize both tundra and boreal forest habitats during winter (Joly et al., 2007a; Joly, in press). We also calculated these extents for just the herd's core winter range (Figure 4.1).

Results

Mean AB from our 50 replicate ALFRESCO simulations were significantly correlated with historical, empirical data from 1950-2007 ($F = 13.19$, $P < 0.001$, $DF = 56$, $r^2 = 0.191$). While the empirical and simulated AB means were very similar (977 versus 953 km^2 per year, respectively), annual variability was much greater for the historical than the modeled data ($SE = 251$ versus 89, respectively). The historical data had an AB range of 0-10345 km^2, whereas the modeled output was 501-5183 km^2. Our attempts to restrain mean AB to historic levels came at the expense of conservative maximum AB and lower variability in the simulated data. We found that allowing the variability of simulated data to approach the actual data allowed for unrealistic maximum AB in warm years (e.g., 2005). Our model presents a compromise between realistic, yet conservative, maximum AB and variability. While the one-to-one relationship between actual and modeled mean AB was not strong, several other metrics (e.g., cumulative area burned over time, maximum fire size, and mean AB of tundra) indicated that our models performed relatively well (Figure 4.4). The strength of the correlation between of each of the 50 runs to the empirical, observed data was variable (r^2 mean = 0.092, range = 0.001 – 0.372).

The average annual AB within the study area from 1950-2007 was 953 ± 89 km^2 (mean, SE). Our projections reveal that within the near term (2008-2053) annual AB would increase (to 1235 ± 154 km^2) under the ECHAM5 GCM, but not for the CGCM3.1 (924 ± 110 km^2). Total annual AB was projected to continue to increase for both

CGCM3.1 (1188 ± 110 km^2) and ECHAM5 (1460 ± 154 km^2) in the 2054-2099 time period. These increases in AB incorporate the fire- and succession-induced changes in flammability through time.

Tundra comprised 73% of the study area; this proportion did not change because our model assumed burned tundra was self-replacing and did not incorporate treeline or successional changes. We did, however, allow the probability of tundra burning to vary as a function of time since last fire due to vegetation-induced changes in flammability (Figure 4.2). We projected increases in the amount of tundra AB using CGCM3.1 in the latter half of the century (2054-2099; 52%), but not the near term (Figure 4.5). In using ECHAM5, the amount of tundra AB increased by 61% during the near term and more than doubled in the 2054-2099 time period (Figure 4.5).

The projected annual AB of spruce showed only very modest increases over the coming century for either climate scenario (Table 4.1). However, the total amount of spruce on the landscape was predicted to decline substantially throughout the remainder of the century (Table 4.1).

AB in deciduous stands during the historic period was only 54 ± 9 km^2. Using CGCM3.1, substantial increases were detected for 2054-2099 (98 ± 10 km^2) but not the near term (64 ± 10 km^2). ECHAM5 projections revealed substantial increases in deciduous AB from 2008-2053 (85 ± 9 k m^2) and 2054-2099 (110 ± 9 km^2). The total amount of deciduous forest was projected to increase substantially from the historic period (48643 ± 156 km^2) for both future time periods (55444 ± 175 km^2 and 60746 ±

175 km^2, respectively) under CGCM3.1. ECHAM5 projected increases were also substantial for both time periods (59930 ± 227 km^2 and 65136 ± 227 km^2, respectively).

Potential high quality caribou winter range ≥ 50 years post-fire compromised 84% of the study area. This area was projected to decline to 82% during the near term and again to 79% during the 2054-2099 period using CGCM3.1 (Figure 4.6). Larger declines were projected using ECHAM5 (79% for 2008-2053 and 75% for 2054-2099; Figure 4.6).

Quality moose habitat, expressed as deciduous cover types 10-30 years old in the simulations, during the historic period averaged 11045 ± 268 km^2. Using CGCM3.1, moose habitat was projected to increase in the near term (13606 ± 301 km^2) and the 2054-2099 time period (14475 ± 301 km^2) compared to the historic period. ECHAM5 projected an even larger (47%) increase (16247 ± 248 km^2) in the near term. While lower than the near-term time period, the amount of quality moose habitat was projected to be substantially greater from 2054-2099 (14619 ± 248 km^2) than during the historic period.

Amount of AB, caribou winter range, and moose habitat within the WAH's core winter range (Figure 4.1) showed trends that were similar to those in the entire winter range (i.e., study area). AB within the core winter range was projected to substantially increase under the ECHAM5 in the near term (490 ± 82 km^2) and for both CGCM3.1 and ECHAM5 scenarios (476 ± 52 km^2, 531 ± 82 km^2, respectively) towards the end of the century (2054-2099) as compared to the reference period (281 ± 46 km^2), but not for CGCM3.1 in the near term (301 ± 52 km^2). The substantial declines (15-29%) in simulated high-quality caribou winter range ≥ 50 years old were more dramatic within the herd's core winter range (Figure 4.7). Simulated quality moose habitat increased substantially in the near term within the core winter range under both CGCM3.1 (2415 ± 39 km^2) and ECHAM5 (3346 ± 69 km^2) scenarios as compared to the reference period (2038 ± 35 km^2).

Discussion

Wildfire is difficult to model within the tundra and tundra-forest interface (Balshi et al., 2009; this study). We found that compromises had to be made among maximum, mean, and variability of AB to achieve realistic simulations. One of the most unheralded virtues of modeling effort is that they are highly effective at identifying knowledge gaps. In order to reduce model uncertainty in the future, we suggest that subsequent efforts develop parameters for multiple tundra types (e.g., lowland wet sedge tundra, upland dwarf shrub tundra), incorporate impacts from treeline advance and herbivory, and develop more detailed successional pathways that incorporate the potential for increasing shrub abundance. Moreover, developing models that differentiate years with little to no wildfire and those with substantial wildfire should alleviate some of the difficulties we had with balancing inter-annual variability with mean AB. On a practical side, the paucity of climate stations in the Arctic, especially inland, is a serious concern and should be addressed. In addition, given the vast expanses of tundra in Russia and Canada, we suggest future modeling efforts analyze the impacts of climate change on tundra fire regimes in these regions.

Our simulations project that the amount of AB in northwest Alaska could increase by up to 30% in the near term (2008-2053), as compared to our reference period (1950-2007), using an intermediate-level emissions scenario (A1B). This region is dominated by tundra, which we project will see proportionately greater (0-61%) increases in AB in the near term. We forecast that these relatively modest increases in AB will continue to increase towards the end of the century (2054-2099). While the quicker rebuilding of fuels in tundra plays a role in our projections for greater proportions of tundra burning in the future, our simulations project that the amount of AB in spruce habitats in northwest Alaska will only modestly increase during the upcoming century. This projection seemingly runs contrary for predictions of Interior Alaska (e.g., Balshi et al., 2009). However, our simulations also project that the amount of spruce on this landscape will decline substantially, thus the proportion of spruce on the landscape that will burn is actually forecasted to increase in our simulations. The declines in spruce abundance we forecast are in line with other recent projections that predict major biome shifts within Alaska during the coming century (e.g., Gonzalez et al., 2010, Beck et al., 2011). However, spruce should be able to expand into tundra habitats as permafrost thaws and active layers deepen (Lloyd et al., 2003). Disturbance by caribou may also facilitate the expansion of spruce by exposing mineral soils (Tremblay and Boudreau, 2011). Our model is not currently calibrated to allow for this transition and thus less spruce is predicted on the landscape. Further, less forecasted burning in the latter half of the century is likely a consequence of increased burns in the earlier half of the century leaving spruce habitats less vulnerable to subsequent burning across the landscape. In other words, fuel loads are removed during fire events and, within spruce forests, these fuels can take decades to return to pre-fire levels.

Lichens are the major food source of overwintering caribou that incur the energetic expenses of migration and predation, such as the WAH (Klein, 1982; Russell et al., 1993; Joly et al., 2007b). They are a critical source of carbohydrates that help these caribou survive winter until emergent forage appears in the spring (Person et al., 1980; Parker et al., 2005). Unfavorable nutritional status can reduce growth, compromise in utero development of fetuses, and have multiplier effects (White, 1983; Parker et al., 2005). In tundra and boreal forest, caribou forage lichens can take 50-80 years or more to return to previous abundance following wildfire (Joly et al., 2010; Collins et al., 2011). Thus, declines in winter forage, induced by climate change and increased wildfire, could lead to lowered nutritional status of individual animals that can translate into population–level impacts.

We project that the combination of substantially more burning in tundra and less overall spruce habitat should lead to modest (2-6%) reductions in the quantity (areal extent) of quality caribou winter range, as indexed by tundra and spruce habitat > 50 years old, in the near term. The declines were only slightly greater for the latter half of the century (2054-2099; 5-10%). The declines were more dramatic within the herd's core winter range (e.g., 15-30% of > 50 year old habitat by the latter half of the century, Figure 4.7). At the current population size, reductions in the quantity of high-quality winter range of this magnitude within and adjacent to the herd's core winter range may limit the ability of WAH caribou to find alternative lichen-rich winter ranges. This would make mute the argument that fire is not a key factor for caribou population dynamics because caribou can just seek out and discover new, alternative, high-quality winter ranges. The projected levels of burning will also limit the amount of habitat to reach an age where mosses have the potential to out-compete lichens, as some have argued (see Coxson and Marsh, 2001). Smaller herd and group sizes, changes in range use, increased dispersal, and diminished reproductive output are all potential consequences of decreased habitat quality and quantity.

Our simulations project that the total amount and AB of deciduous habitat will increase substantially, at the expense of spruce habitat, throughout the remainder of the century. Consistent with these forecasts, we projected large increases (19-64%) in high-quality moose habitat in the near term within the herd's core winter range, which should facilitate increases in moose abundance there (Weixelman et al., 1998; Maier et al., 2005). In an example of apparent competition (Holt, 1984), resident moose populations on or adjacent to caribou winter range may facilitate increased wolf abundance and therefore the potential for increased wolf predation on caribou (James et al., 2004). Caribou are easier for wolves to take than moose due to their much smaller relative size. Latham et al. (2011) documented increases in wolf abundance and caribou predation in Alberta where deer (*Odocoileus* spp.) densities greatly expanded due to more development-induced early seral habitats. If increased fire did promote moose abundance and hence wolf predation on caribou, it would be a novel example of an indirect, detrimental impact of climate change on caribou population dynamics. The impact of this potential relationship would be greater if the WAH experiences large population declines. Further, numerous interior Alaska herds that are already small (< 2000 caribou) may be most at risk if wolf populations increase, because low-density caribou populations are thought to be more vulnerable to predation (e.g., Dale et al., 1994).

Conclusions

Climate change is predicted to impact caribou in many ways. Decreased forage accessibility during winter, either from icing or increased snow depths, may have a stronger impact than increased summer forage biomass (Tews et al., 2007). Although wildfires occur during the summer, they negatively impact caribou winter range, which will only exacerbate forage accessibility issues. Tundra has potential to re-burn much more quickly than boreal forests, so warmer summer conditions could lead to additional fires (Joly et al., 2009b). Increased warming and burning will also likely facilitate increases in the abundance of shrubs and trees in the tundra (Rupp et al., 2000a; Sturm et al., 2001; Joly et al., 2009a; Forbes et al., 2010; Beck et al., 2011) and thus increase the extent and severity of fires (Higuera et al., 2009; Hu et al., 2010a). Since our models did not account for potentially substantial amounts of tundra being converted or the direct, negative impacts of warmer and drier conditions on lichen growth, we believe that our results are fairly conservative.

The amount of increased wildfire on caribou winter ranges we simulated may intensify discussions of the need for fire suppression/management plans for conservation. These plans should incorporate traditional ecological knowledge, co-management input, and logistical realities (Beverly and Qamanirjuaq Caribou Management Board, 1994; Urquhart, 1996; Klein et al., 1999), as well as scientific information, including our results, and needs of other species of interest. These plans will likely have to be tailored to individual herds and updated regularly to take into account rapidly changing conditions. For the WAH, our forecasts for modest increases in AB over the herd's entire range suggest that any fire suppression efforts be focused on its core winter range.

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Table 4.1. Amount (km^2) of spruce (mean, SE) area burned and total area on the landscape in the observed-historic record and using two climate scenarios to project into the future.

Source	Historic	CGCM3.1	CGCM3.1	ECHAM5	ECHAM5
Time Period	1950-2007	2008-2053	2054-2099	2008-2053	2054-2099
Area Burned	593 ± 54	559 ± 61	624 ± 61	656 ± 59	621 ± 59
Total Area	54167 ± 156	47366 ± 175	42064 ± 175	42880 ± 227	37674 ± 227

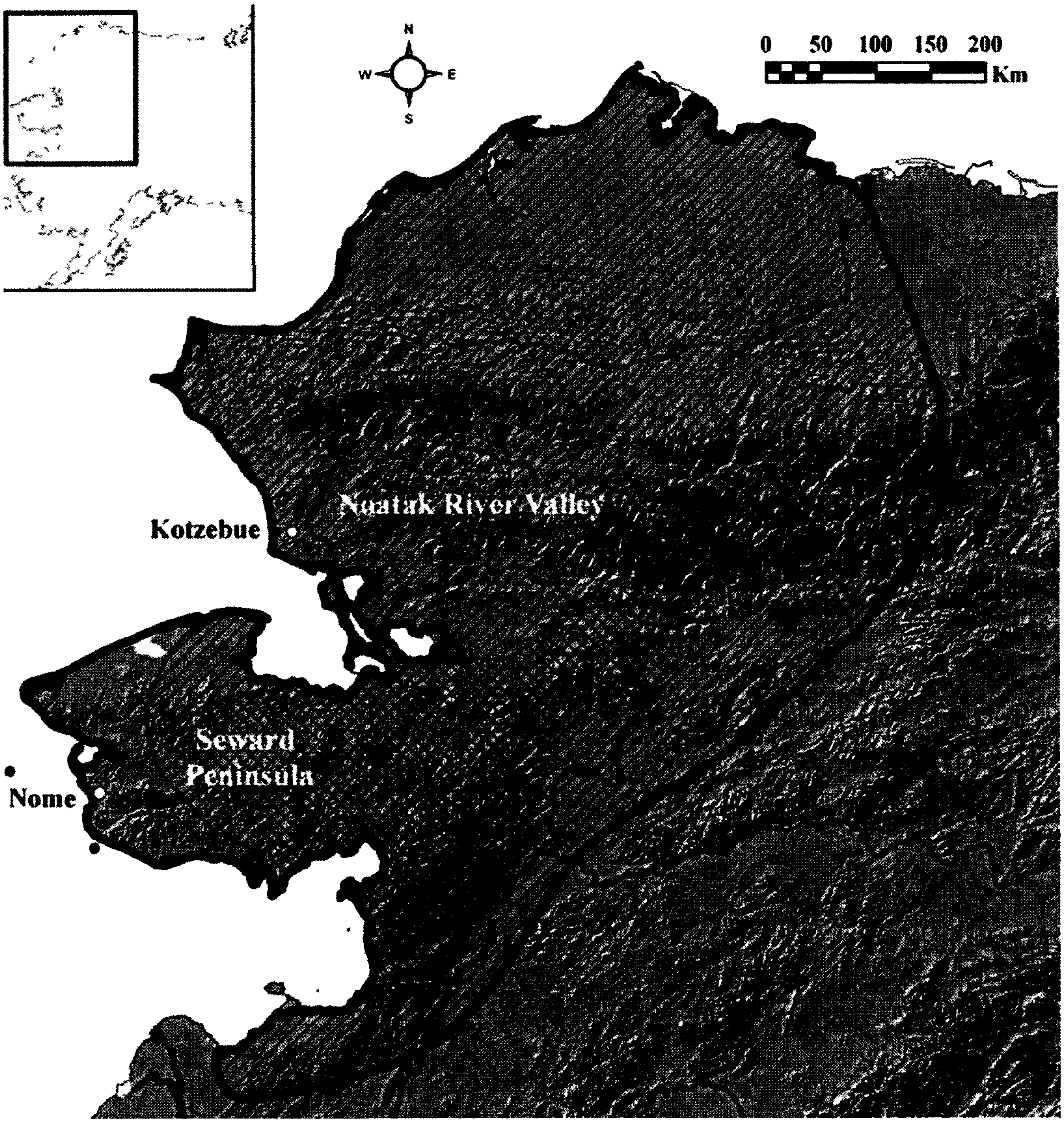

Fig. 4.1. Simulation domain (outlined by thick black line) in northwest Alaska. The range of the Western Arctic Caribou Herd is contained within the domain and is depicted with hatching, while the herd's core winter range is cross-hatched (courtesy of the Alaska Department of Fish and Game).

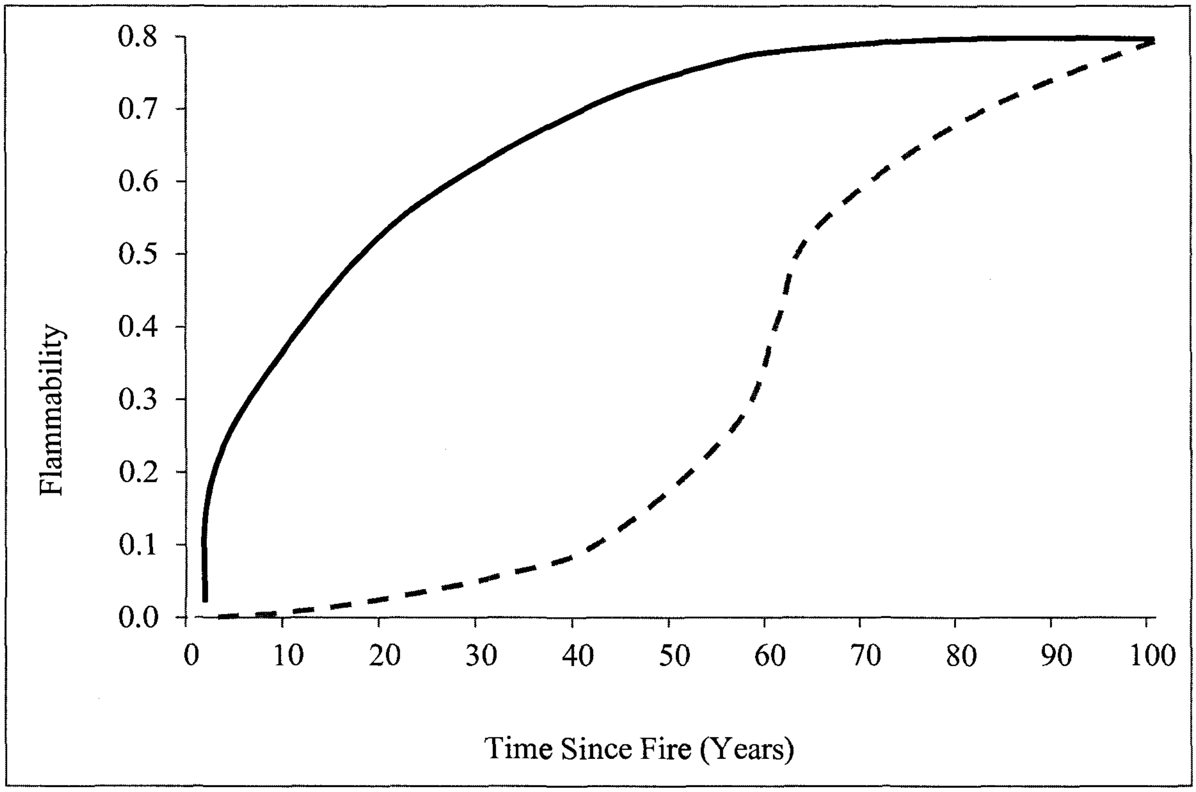

Fig. 4.2. Modeled flammability of pixels as a function of time since fire in northwest Alaska (boreal forest depicted with a dotted gray line and tundra with a solid black line).

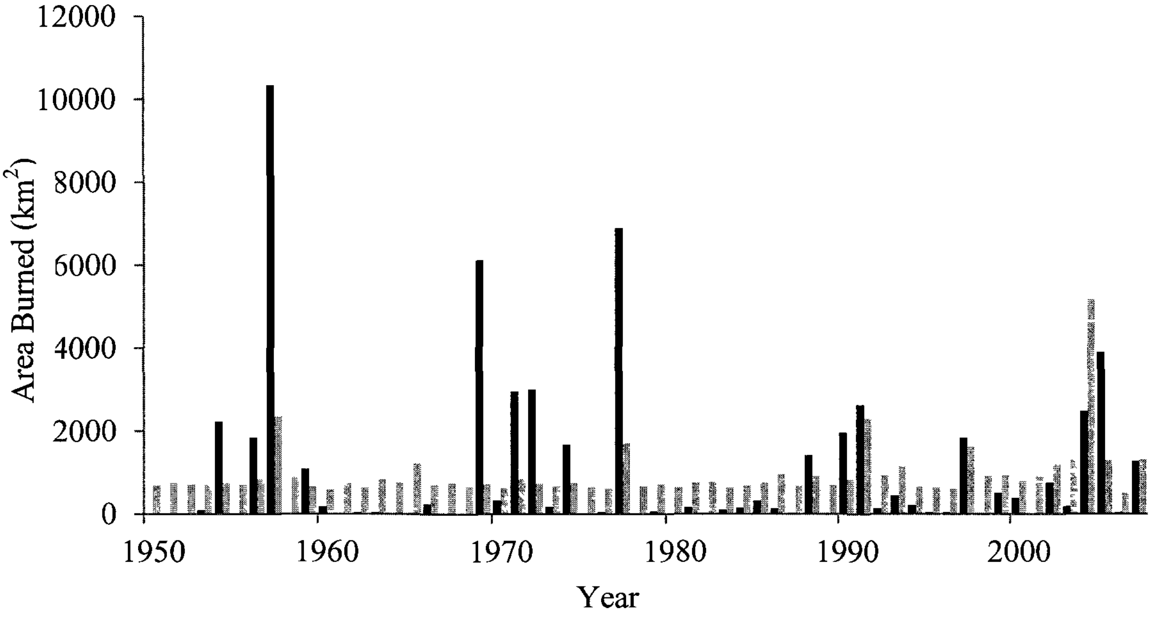

Fig. 4.3. A comparison of annual area burned (km^2) between observed (black bars) and the mean of 50 simulated ALFRESCO runs (gray bars) for northwest Alaska, 1950-2007.

Fig. 4.4.A

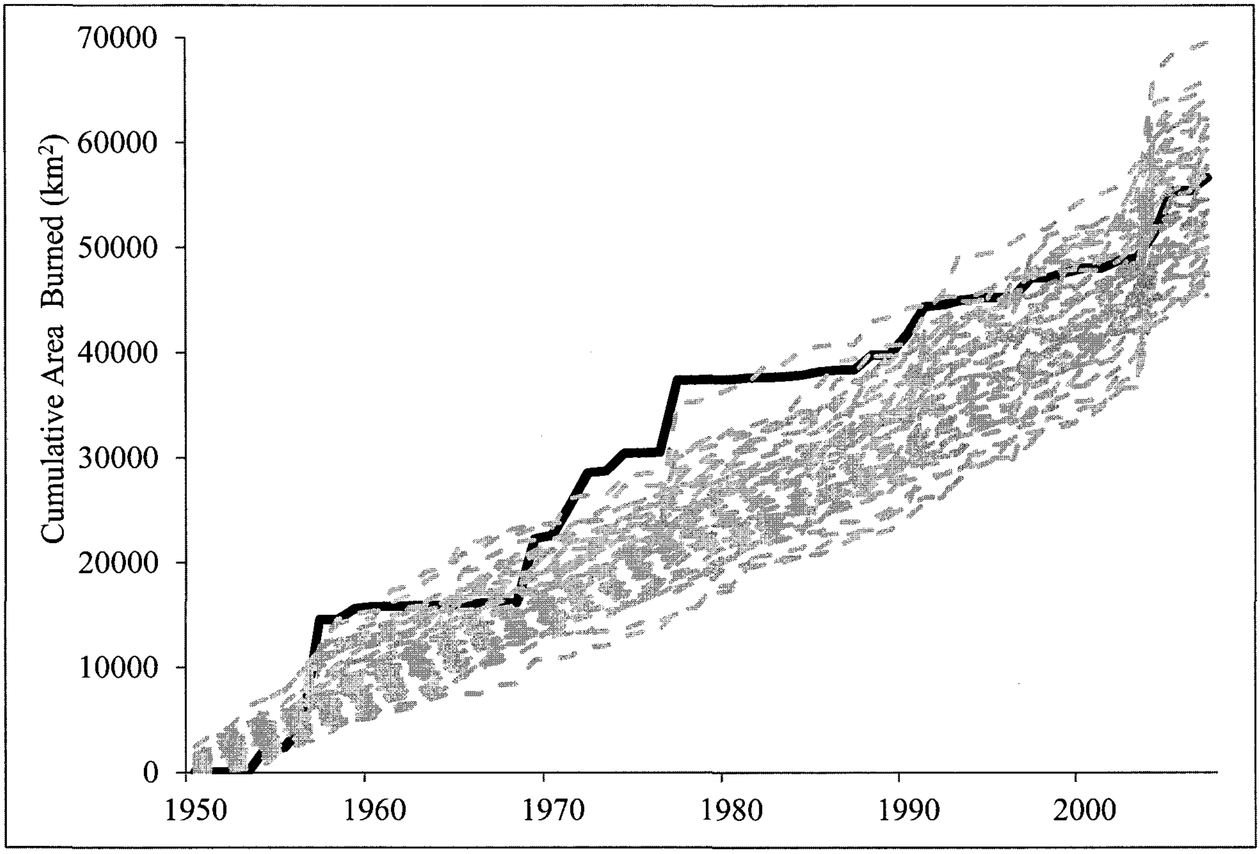

Fig. 4.4. Performance metrics comparing simulated (gray; n = 50) and historic-empirical (black line) wildfire data for northwest Alaska, 1950-2007. A) cumulative area burned, B) maximum fire size, and C) mean area burned of tundra.

Fig. 4.4.B

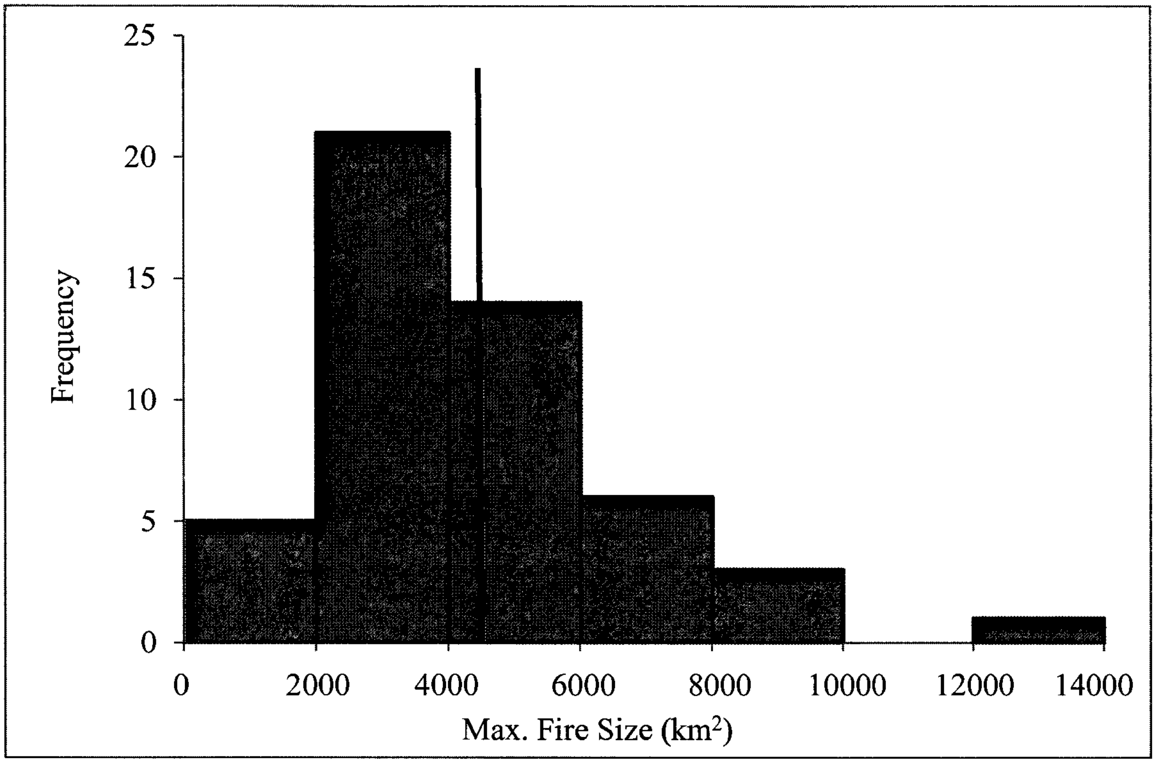

Fig. 4.4. Performance metrics comparing simulated (gray; n = 50) and historic-empirical (black line) wildfire data for northwest Alaska, 1950-2007. A) cumulative area burned, B) maximum fire size, and C) mean area burned of tundra.

Fig. 4.4.C

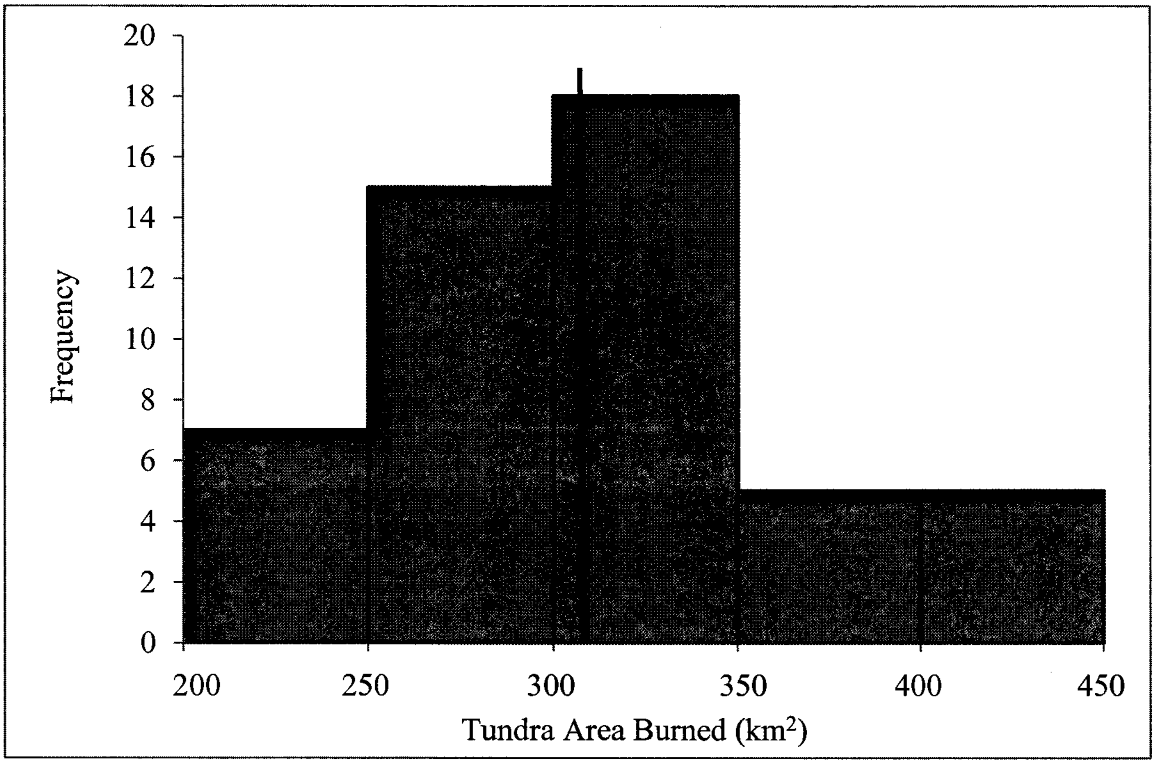

Fig. 4.4. Performance metrics comparing simulated (gray; n = 50) and historic-empirical (black line) wildfire data for northwest Alaska, 1950-2007. A) cumulative area burned, B) maximum fire size, and C) mean area burned of tundra.

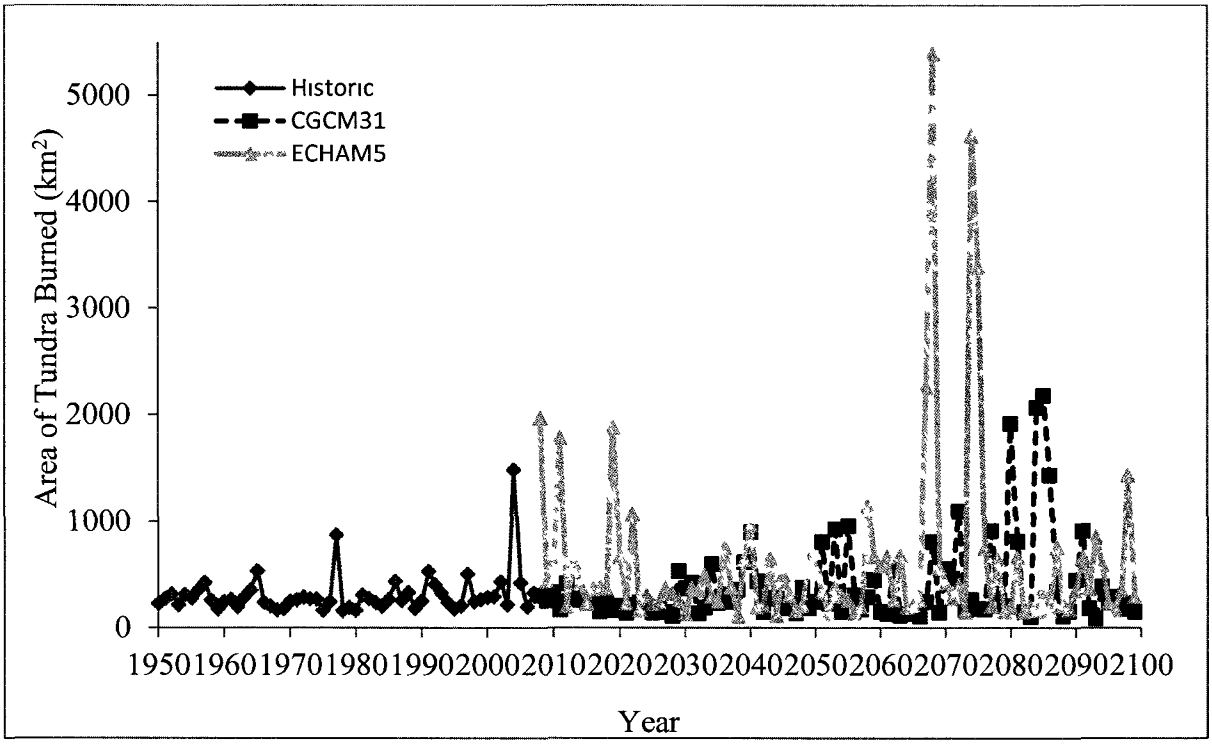

Fig. 4.5. Amount of tundra burned (km^2) in northwest Alaska for the historic reference period (1950-2007; thin solid line with diamond markers) and future projections using ECHAM5 (thick solid line with triangle markers) and CGCM3.1 (dotted line with square markers) general circulation models (GCMs) under the A1B emissions scenario.

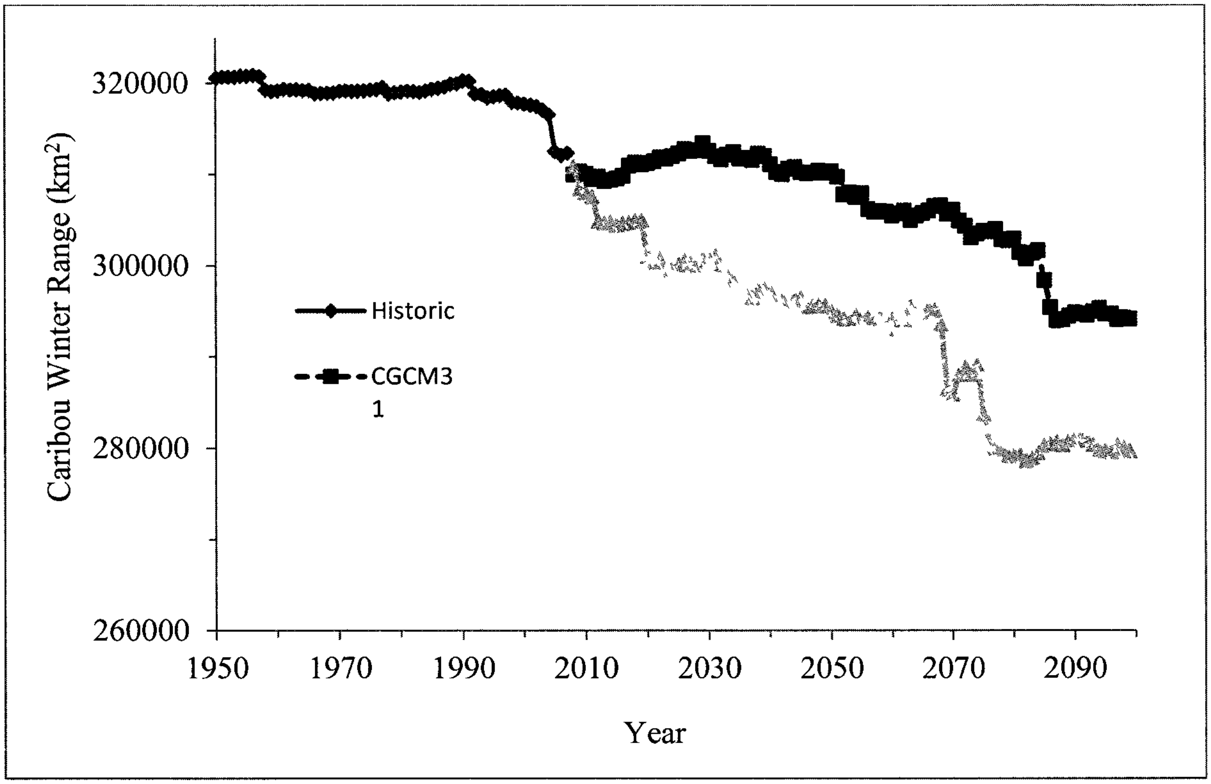

Fig. 4.6. Projected changes in areal extent of quality caribou winter range (> 50 years of age) in northwest Alaska. The historic reference period (1950-2007) is depicted with a thin solid line with diamond markers and future projections using ECHAM5 with a thick solid line with triangle markers (upper line) and CGCM3.1 with a dotted line with square markers (lower line). These general circulation models (GCMs) employed the A1B emissions scenario.

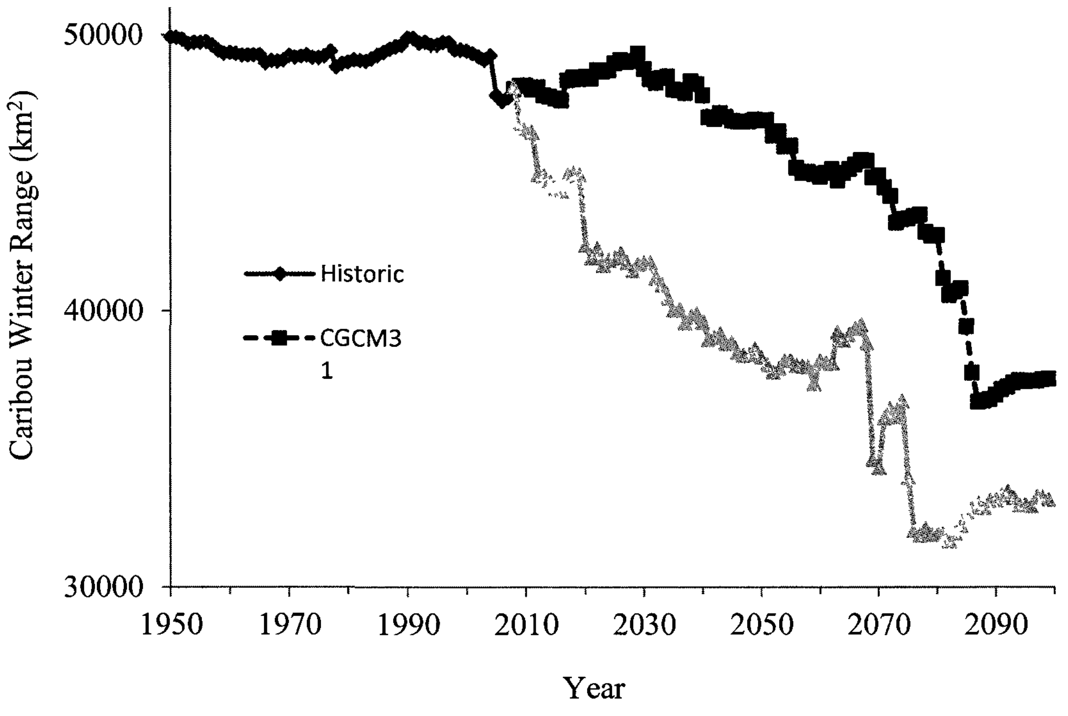

Fig. 4.7. Projected changes in areal extent of quality caribou winter range (> 50 years of age) solely within the core winter range of the Western Arctic Herd, northwest Alaska. The historic reference period (1950-2007) is depicted with a thin solid line with diamond markers and future projections using ECHAM5 with a thick solid line with triangle markers (upper line) and CGCM3.1 with a dotted line with square markers (lower line). These general circulation models (GCMs) employed the A1B emissions scenario.

CONCLUSION

Overview

My thesis advanced the state of knowledge on the effects of wildfire on caribou (*Rangifer tarandus*) tundra winter range and how climate change may alter this relationship – important research topics that have, heretofore, been largely ignored. Globally, there has been increased focus on caribou in recent years because of the endangered status of woodland caribou in Canada and the extirpations of small herds, including those within protected areas (Hebblewhite et al., 2010). My results add to a growing list of climate change-related impacts that may negatively affect caribou in the North (e.g., Jefferies et al., 1992; Tews et al., 2007; Vors and Boyce, 2009).

By analyzing the Alaska Fire Service's Large Fire Database, I determined that wildfire has covered over 10 % of northwest Alaska, including 9 % of the tundra in this region from 1950-2007 (CHAPTER 1), which revealed that tundra fires as not as uncommon as previously reported (e.g., Wein, 1976). The regional fire cycle (i.e., the number of years required to burn over an area) for tundra was 630 years, however it was much shorter for the tundra-dominated Seward Peninsula, Nulato Hills, and Kobuk Ridge and Valley ecoregions (418, 283, and 193 years, respectively; CHAPTER 1). Moreover, tundra was 4.5 times more likely to reburn during the study period (1950-2007) than boreal forest. This completely novel finding is intuitive because tundra re-vegetates so quickly and vigorously after a fire (Jandt et al., 2008; CHAPTER 3).

I was unable to detect an increasing trend of annual area burned in northwest Alaska using the entire database (CHAPTER 1). However, if large (> 81,000 hectares) were censured, a strong increasing trend in annual area burned was readily apparent. There were 15 years during the study period with large pulses of annual area burned (> 81,000 hectares), which were clustered into 4 groups (CHAPTER 1). I found that each group contained 3-5 years of high area burned spanning 4-9 years and were temporally separated by about 16 years (CHAPTER 1). Large fire years in northwest Alaska were correlated with warm dry summers (CHAPTER 1). However, unlike the boreal forests of the Interior Alaska, temperatures in August rather than June appeared to be of greater importance (Duffy et al., 2005; CHAPTER 1). Indeed, I found that the average annual area burned more than doubled in years where mean August temperatures exceeded 11.7 ^{o} C (53^{o} F) versus cooler years. Similar to the boreal forests of the Interior, annual area burned in northwest Alaska was correlated with the large-scale climate pattern known as the Pacific Decadal Oscillation (PDO; Duffy et al., 2005; CHAPTER 1).

I used resource selection function (RSF) techniques to determine the distribution of Western Arctic Herd (WAH) caribou during winter was influenced by habitat type and terrain, as well as other factors (CHAPTER 2). Unlike many other barren-ground herds, the WAH utilized tundra habitats extensively during winter (CHAPTER 2). Caribou cows selected habitats that are associated with high abundance of forage lichens (CHAPTER 2). Cows were found at significantly lower elevations and gentler slopes and moved significantly more throughout the winter than bulls (CHAPTER 2). These results support the theory that cows seek to maximize energy intake during the winter while bull opt to minimize energy output and reduce predation risk (Roby and Thing, 1985). These strategies likely change seasonally as I found that cows exhibit greater movement rates in April as they head to the calving grounds while the bulls linger further south, possibly to take advantage of emergent spring growth (CHAPTER 2). Moose (*Alces alces*) density, used as an index of predation pressure, was not strongly correlated with caribou distribution at the range-wide scale or for bulls (CHAPTER 2). Because logical arguments can be made for why either positive or negative correlations between moose/wolf (*Canis lupus*) and caribou densities can be made (i.e., wolves are successful where caribou overwinter and caribou avoid areas with high wolf densities, respectively), and the myriad of factors that can affect caribou distribution in winter (CHAPTER 2), I suggest that moose density may not be the most informative factor.

The RSF map I generated highlights the important wintering areas of the Nulato Hills and the Seward Peninsula (CHAPTER 2). The northern Brooks Foothills, an area with relatively low snowfall and wolf densities, was predicted to have moderately high probability of use (CHAPTER 2). However, this area consistently sees little use by the WAH (Joly et al., 2007a). These results lend support to the argument that terricolous lichens are an important winter forage and determinate of WAH caribou distribution in winter (CHAPTER 2; CHAPTER 3).

From vegetation plot data, I was able to determine that WAH caribou were able to seek out and find winter range with relatively high lichen forage abundance (CHAPTER 3). Forage lichen abundance was > 4 times greater in unburned areas versus recently (< 58 years) within the current winter range of the WAH (CHAPTER 3). Besides the significant difference in forage lichen abundance, there were few distinguishing vegetative characteristics between these two study plot types, suggesting that this is why caribou avoid recently burned areas in the tundra and boreal forest for decades following wildfire (CHAPTER 3; Joly et al., 2007a).

The impacts of herbivory by caribou can be considerable (e.g., Moser et al., 1979; Jefferies et al., 1992). I found that forage lichen cover was low in the WAH's historic versus current winter range (3.7 % and 10.6 %, respectively), which may help explain the herd's expansion into new areas (CHAPTER 3). I believe it is unlikely that large numbers of WAH will frequently use the heart of Koyukuk National Wildlife Refuge due to lower lichen abundance, increased wildfire activity, extensive deciduous forests and riparian habitats, and higher densities of moose (CHAPTER 1; CHAPTER 2; CHAPTER 3).

Plant productivity is not controlled by "*one or even a few factors, but is the aggregate response of many species reacting more or less independently to many environmental factors. Factors important to one species need not be important to another*" (Scott and Billings, 1964). The same can be said about lichen (or even caribou) productivity. I found lichen productivity was positively associated with stand age, but negatively with pH, shrub cover and herbivory pressure (CHAPTER 3). Irradiance, as determined by canopy cover, as well as precipitation and temperature were found to be key drivers of lichen growth in other studies (e.g., Holt et al., 2009; Čabrajić et al., 2010). After light, water is the major limiting factor on lichen growth rates the landscape scale (Holt et al., 2009; Čabrajić et al. 2010).

Climate change is likely to bring about changes in species composition in the tundra biome (McGraw and Fletcher, 1992; Chapin et al., 1992). I found that fire-adapted dwarf birch (*Betula* spp.) was more abundant in recently (<58 years) burned areas than unburned areas (i.e., > 58 years old; CHAPTER 3). Vegetation type is an important driver of fire regime and the abundance of dwarf birch is associated with greater burning (Higuera et al., 2008; Higuera et al., 2009; Hu et al., 2010a), thus even greater prevalence of wildfire should be expected in the future within the winter range of the WAH. Additional fire could help facilitate greater cover of graminoids, dwarf birch, willow and could advance treeline line bringing boreal forest into formerly tundra habitats (Bryant and Reichardt, 1992; Rupp et al., 2001; Racine et al., 2004; Jandt et al., 2008; CHAPTER 3). Increased proportion of boreal forest within the winter range of the WAH would likely lead to more plants with herbivore defenses and hence more herbivores tolerant of plant defenses such as moose (Bryant and Reichardt, 1992).

I modified the computer simulation model ALFRESCO (Rupp et al., 2002) to incorporate the rapid rate at which tundra fuels build following fire (CHAPTER 1; CHAPTER 3) and calibrated it to the ecoregions of northwest Alaska. I was able to simulate the impacts of an intermediate-level CO_2 emissions scenario (A1B) on the fire regime of northwest Alaska (Nakicenovic et al., 2000; IPCC 2007) using the ECHAM5 and CGCM3.1 general circulation models (GCMs) from 2008-2099. My ALFRESCO simulations forecasted that area burned (AB) in northwest Alaska is will increase by 0-30% in the near term (2008-2053) over my historic reference period (1950-2007), depending on general circulation model (CGCM3.1 or ECHAM5) employed. Further into the future (i.e., 2054-2099), projections continue to increase (25-53% greater AB than reference period). The increases in AB are largely accounted for by increases in the AB within the tundra ecosystem, where simulations project 0-61% increases in the near term over my historic reference period. The AB of spruce was not projected to significantly increase, but this was primarily due to significant declines in the areal extent of spruce habitats on the landscape.

The combination of less spruce forest and more tundra burning on the landscape drove significant declines in extent of quality caribou winter range (as indexed by tundra and spruce habitats > 50 years old; CHAPTER 3) in my simulations. These declines ranged from 2-6% in the near term for > 50-year old range in the near term. Declines (5-10%) were greater towards the latter part of the century (2054-2099). The reduction of quality winter range was greater within the herd's current core winter range, with declines of 3-16% in >50-year habitat. Declines of this extent have the potential to negatively impact the population dynamics of the WAH through changes in nutritional and reproductive performance.

Increased abundance of ungulates can facilitate numerical responses in wolf populations (Fuller, 1989; Latham et al., 2011). Ungulate populations can also respond numerically to increased abundance of early seral habitats (Maier et al., 2005; Latham et al., 2011). My simulations projected that high quality moose habitat (as index by 10-30 year old deciduous stands) would increase by 23-47% in the near term and by 19-64% within the herd' core winter range. Latham et al. (2011) documented increased predation of caribou by wolves where ungulate biomass increased indirectly from development-induced increases in early seral habitats. Thus, increased wildfire in northwest Alaska due to climate change could have indirect, negative impacts to caribou populations via increased predation to due increased abundance of high quality moose habitat. My simulation results are likely conservative because my models do not account for tree line advance or conversion of tundra to shrub lands.

The effects of climate change are readily apparent in the Arctic where mean annual temperatures have increased by 2-3° C in recent decades (Callaghan et al., 2004; Hinzman et al., 2005). These changes are predicted to be highly detrimental to arctic land animals (Lawler et al., 2009). Moderating temperatures in the Arctic could lead to suite of changes that could affect caribou ecology. Among the potential positive effects are that summer forage should be improved (Tews et al., 2007); for example, cottongrass (*Eriophorum* spp.) flowering increases after fire (Wein and Bliss, 1973). However, forage accessibility during winter will likely have a stronger impact than increased summer forage biomass (Jefferies et al., 1992; Tews et al., 2007). Rain-on-snow and icing events are the most commonly cited negative impacts of climate change for caribou (e.g., Jefferies et al., 1992; Putkonen and Roe, 2003; Vors and Boyce, 2009). Amplifying feedback mechanisms, such as increasing shrubs (Chapin and Shaver, 1985; Jefferies et al., 1992; Tape et al., 2006) will likely increase the impact these changes. Increased shrub cover will reduce the abundance of critical caribou forage lichens (Jefferies et al., 1992; Chapin et al., 1995; Cornelissen et al., 2001; Joly et al., 2009). One of the most obvious impacts of recent climate change has been the unprecedented reduction in Arctic sea ice cover (Comiso et al., 2008). The extent of sea ice reached a minimum in 2007, coincident with the largest ever reported tundra fire north of the Brooks Range, the Anaktuvuk River Fire (Hu et al., 2010b). Another impact of climate change will likely be increased human activity in the Arctic (Chapin et al., 1992; Jefferies et al., 1992; Johnson et al., 2005) and the potential for even more wildfires (CHAPTER 4). More fires, whether ignited by human activity or lightning strikes, will be an additional negatively impact caribou winter range exacerbating forage accessibility issues.

Caribou have tolerated many large changes in climate and have high physiological and behavioral plasticity (Jefferies et al., 1992; Klein, 1999), thus extirpation in near future (50-100 years) seems unlikely. However, the speed and magnitude of current climate change is unprecedented, and caribou may not be able to respond as well as they have previously (Jefferies et al., 1992; Callaghan et al., 2004), especially with the addition of the still relatively novel stressor of large-scale industrial development. Thus while caribou may be around for the foreseeable future, the spectacle of hundreds of thousands caribou migrating together for thousands of kilometers may be in jeopardy.

Preliminary Fire Management Plan for Western Arctic Herd Caribou Winter Range

My thesis represents the scientific component of a fire management plan for the WAH. There are, however, two other critical components that have not yet been addressed; Traditional Ecology Knowledge (TEK) and the management and logistical realities as defined by the Alaska Fire Service and land managers. Using the scientific results documented here and the fire management plan for the Beverly and Qamanirjuaq caribou herds (Beverly and Qamanirjuaq Caribou Management Board, 1994) as a foundation, I provide a preliminary fire management plan for WAH caribou from which a final plan that incorporates TEK, logistical realities, human safety, structures, cultural resources, moose and other wildlife habitat, and miscellaneous forest products can be developed. Such a detailed final plan is outside the purview of my thesis.

I estimated the fire cycle for the WAH's core winter range to be 296 years (CHAPTER 1). After a large population decline in the early 1970s, the WAH rebounded and maintained a large (> 200,000 caribou) population size for the past 25 years (Dau, 2007). So while there are some signs of winter range deterioration (Joly et al., 2007b; CHAPTER 3), the herd has been in generally good condition (Dau, 2007). Thus, maintaining a 296-year fire cycle within core winter range and a 16 km (10 mile) buffer around it may be sufficient to sustain the WAH at a large population level (> 200,000 caribou). An annual rate of burning of 0.334 translates to a 296-year fire cycle (100/296 = 0.334) and approximately 84 % of the range being > 50 years old (Table 3, page 18, Beverly and Qamanirjuaq Caribou Management Board, 1994). I therefore recommend that this should be the goal for fire management in this zone.

It is difficult to quantify or manage area burned in real time in the remote regions of northwestern Alaska. Further, weather, fire behavior and logistics often make it difficult or even impossible to cap the amount of area burned annually. Therefore, I recommend using a 3-year moving average of 0.334 which will allow for additional suppression in years where it is more practical and let surplus acreage burn in years where it is not feasible to fight them effectively. However, efforts should be taken to avoid the situation where vast areas are burned in 1 year, then extra efforts are made to suppress fires in out years because this will be more expensive but also set up conditions to promote further boom-bust fire cycles, at least in areas of boreal forest (Beverly and Qamanirjuaq Caribou Management Board, 1994). Annual area burned and the 3-year moving average within the core winter range, plus the buffer, as well as the herd's entire range should be tracked and reported annually to the WAH Working Group by the Bureau of Land Management office with the management responsibility for the majority of the Nulato Hills.

I recommend focusing suppression efforts on large (> 404 hectares [1000 acres]) blocks of habitat that are mapped in Alaska Fire Service's Large Fire Database as unburned or > 30 years old (Beverly and Qamanirjuaq Caribou Management Board, 1994). While 30 year old stand typically do not have high lichen abundance, they are well on there was towards recovery and thus have value as potential future caribou winter range (CHAPTER 3). Ideally, a resource advisor familiar with the lichen resources and snow patterns of the WAH's winter range would be available to fire management officers in charge of in-season suppression efforts. Areas with low forage lichen potential, such as riparian zones, should not be targeted for suppression even if they fall in the > 30 year old age category. Similarly, areas that typically receive very deep (> 1.5 m) snow cover should not be a primary target for fire suppression efforts (Beverly and Qamanirjuaq Caribou Management Board, 1994). Under current climate conditions and warming scenarios, the fire cycle in core winter range is fast enough to eliminate concerns about significant portions of the range with abundant forage lichen stands becoming 'over-mature' and being degraded by moss (Coxson and Marsh, 2001; CHAPTER 1; CHAPTER 4).

The herd's core winter range contains portions of the Selawik National Wildlife Refuge, Koyukuk National Wildlife Refuge, Bering Land Bridge National Preserve, and Gates of the Arctic National Park and Preserve which are administered by the US Fish and Wildlife Service and National Park Service. These conservation units are mandated to conserve natural processes within their boundaries so fire management decisions in these areas will have to balance those mandates with those of conserving caribou winter range.

Caribou are known for being unpredictable. The WAH may utilize novel wintering areas if range conditions deteriorate, the herd grows, or other reasons. Similarly, the WAH may contract their winter range, return to their historic ranges or utilize novel areas if herd declines. Caribou and fire managers should be prepared for these situations, and others facilitated by climate change, by allowing the fire management plan to be adaptable to changes in range distribution. Furthermore, the plan needs to be flexible and updated regularly (e.g., every 5 years) because of the potential for changing fire regimes (CHAPTER 4), treeline advance, increasing shrubs, altered suppression budgets, and/or other variables. Another reason for adaptive management is that future research may show that spring or summer ranges are limiting rather than winter range (Beverly and Qamanirjuaq Caribou Management Board, 1994).

Management recommendations

I recommend that agencies that manage wildfires in northwest Alaska utilize the scientific results of my thesis as a foundation to develop a formal fire management plan for the WAH. As the next step, information from TEK sources and fire suppression agencies, as well as input from the general public, should be collected and synthesized prior to the release of a final fire management for the herd because...

"*the success of a management program in a park or reserve encompassing people is dependent upon the establishment of both respect and understanding between administrators and local people*" (Klein 1975)

A plan that synthesizes science, TEK, public input and logistical realities is in the best interest of the herd. I documented high caribou forage lichen abundance in boreal forest and tundra habitats, as well as an array of slopes, aspects, and elevations (CHAPTER 3). The inability to identify or model with precision fine-scale lichen patches on the landscape level suggests that stand age should be used as a simple proxy for forage lichen abundance for now.

My simulations suggest that the impacts of climate change on the wildfire regime of northwest Alaska will be greatest on the WAH's core winter range (CHAPTER 4) and thus this is where suppression efforts, if deemed appropriate, should be focused. By limiting the annual area burned within the core winter range to 0.334 using a 3-year moving average, managers should be able to keep the fire cycle of the area at its known historic levels but also have some flexibility among years on the amount of suppression that is feasible. For now, stand age should be used as an index for forage lichen abundance, while noting some habitats, such as riparian corridors, will likely have low forage lichen abundance regardless of age. I recommend assigning resource advisors familiar with the WAH's fire management plan to fires in northwest Alaska (i.e., Galena Zone) to look for lichen resources, and suppress fire, where practicable, to protect large (> 404 hectares [1000 acres]) areas with high lichen abundance. I also recommend looking into developing priority zones for suppression if resource advisors are not available and/or suppression budgets necessitate more focused efforts.

My thesis provides a strong foundation for future research involving caribou winter range and the impacts of tundra fires. Long-term vegetation plots provide critical data to researchers and managers. I strongly recommend that agencies in charge of managing the WAH deploy additional permanent vegetation plots within the winter range of the WAH, especially in the Nulato Hills, Selawik National Wildlife Refuge and the upper Kobuk region, and continue to regularly (every 5-10 years) monitor existing plots.

By incorporating exclosures into plot design, future researchers will be able to tease apart the effects of climate change from herbivory. Future modeling of the impacts of wildfires and climate change on caribou winter range in the tundra should incorporate different tundra frames (e.g., wet tundra versus upland tundra), the impacts of herbivory (Chapin et al., 1992), the potential for an advancing tree and shrub line (Rupp et al., 2001) and species composition shifts that may alter the wildfire regime further (Higuera et al., 2008; Hu et al., 2010a).

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Appendix A:

Programming scripts used in Chapter 4

FIF INPUT FILE FOR FRESCO

```
; PROGRAM PARAMETERS
ClientFifPath = "X:\Western_Alaska_Files" ; The directory that contains this .fif file and is accessible by the ALFRESCO Clients.
ClientInputBasePath = "X:\Western_Alaska_Files" ; The ALFRESCO Client(s) must be able to READ files here.
ClientOutputBasePath = "X:\Western_Alaska_Files\Output" ; The ALFRESCO Client(s) must be able to WRITE files here.
ServerOutputBasePath = "X:\Western_Alaska_Files\Output"                    ; The ALFRESCO Server must be able to WRITE files here.
FirstYear                              = 0                                          ;
LastYear = 147 ; CRU=101 The number of years to run the simulation (starts at year zero ends after completing the x'th year).
MaxReps = 50 ; The number of replicates to run the model (i.e. x replicates total).
RandSeed = 1234763211 ; (IV=1236217828)(III=1262892765)(II=1231284591),(Orig=1129759571) A manually set number used to seed the random number generator. If not set the current time is used to seed the random number generator.
Output.DetailLevel = "MAXIMUM" ; MINIMAL: Show only run progress. MODERATE: Show some interim calculations along with run detail. MAXIMUM: Show maximum detail along with run detail.
```

```
; LANDSCAPE PARAMETERS
RequireAAEACProjectedInputRasters = FALSE ; (optional -- defaults to TRUE) if TRUE, each input raster file will be checked for the Alaska Albers projection.
ApplyAAEACProjectionToOutputRasters = TRUE ; (optional -- defaults to TRUE) if TRUE, each output raster file will include metadata for the Alaska Albers projection.
YSize = 981 ; The total number of rows in the landscape.
XSize = 872 ; The total number of columns in the landscape.
CellSize = 1000 ; The size of each cell. The current size is in meters
CropNeighbors = FALSE ; Should the neighbours algorithm crop calls to cells outside the circular boundary defined by size or should it include all the cells withing the square boundary defined by size.
NoVeg = 0 ; The species number for the default frame type - should be 0 if possible
XLLCorner = -656204.44 ; X Corner specified by ARCgis
YLLCorner = 2390529.456307 ; Y Corner specified by ARCgis
UseUniqueVegAndAgePerRep = TRUE ; Use unique input files for veg and age. This can be used to prevent a bias due to all reps starting from the same input data.
UseUniqueBurnSeverityPerRep = TRUE ; Use unique input files for burn severity per rep. This prevents a bias due to all reps starting from the same input data.
YearOfUniqueInputPerRep = 148 ; The year to be appended to the base filename when using unique veg, age and/or burn severity files per rep. File format: filename_REP#_YEAR#.txt
```

```
VegInputFile = "Input\Initialization\recycle_age\veg.tif" ; The vegetation input file
VegTransitionFile = "Input\Initialization\VegTransitions.txt" ; Base filename for forced vegetation transitions. Year is appended as _YR
IsForcedVegTransitions = FALSE ; TRUE: Succession trajectories are read from files as well as simulated. FALSE: Succession is simulated only.
AgeInputFile = "Input\Initialization\recycle_age\age.tif" ; The initial age input file - years before present so positive values
SiteInputFile = "Input\Initialization\cvr_centaksite.tif" ; The site input file
TreeDensityInputFile = "Input\Initialization\cvr_centakcanopy.tif" ; The initial tree density input file
TopoInputFile = "Input\Initialization\cvr_centakslope.tif" ; The topography input file. 0=flat, 1=complex
BurnSeverityInputFile = "Input\Initialization\recycle_age\burnseverityhistory.tif" ; The initial burn severity input file. 0=No Burn, 1=Low, 2=Moderate, 3=High w/ Low Surface Severity, 4=High w/ High Surface Severity

; MAP OUTPUT FLAGS
; 0x000001    On switch. Output is given only if this flag is specified.
;Formatting
; 0x000010    Output the data in a map (row,col) format rather than the default one column with fieldname.
; 0x000020    Add number of years/reps to header of output file for reference purposes
```

```
; 0x000040	Add ARCgis info to header of output file to facilitate import into ARCgis
;Frame Specific
; 0x000100	Decid: species trajectory;
; 0x000200	Tundra: basal area
; 0x000400	Not in use
; 0x000800	Not in use
;Data Type
; 0x001000	Vegetation type
; 0x002000	Stand age
; 0x004000	Site variables
; 0x008000	Subcanopy
; 0x010000	Fire age (time since the cell last burned)
; 0x020000	Temperature
; 0x040000	Precipitation
; 0x080000	The ID of each fire scar [If ignition cell, '-'][LastBurnYear].[FireID]
; 0x100000	Burn severity for burns occuring in the given year
; 0x200000	Burn severity for for the last known burn of each cell

; MAP OUTPUT PARAMETERS
MapFiles = {"Maps\BurnSeverityHistory.tif", "Maps\BurnSeverity.tif", "Maps\Veg.tif", "Maps\Fire.tif", "Maps\Age.tif", "Maps\FireScar.tif"} ; The base names for the output maps.
```

```
MapCodes = {"FSevHist", "FSev", "Veg", "Fire", "Age", "FScar"} ; The code to output so the user knows this map has been written.
MapFlags = {0x200051, 0x100051, 0x001051, 0x010051, 0x002051, 0x080051} ; The output codes for the map in question.
MapRepStart = {0, 0, 0, 0, 0, 0} ; The first replication that outputs maps.
MapRepFreq = {1, 0, 1, 0, 1, 0} ; The year frequency to output maps.
MapYearStart = {147, 0, 0, 0, 0, 0} ; The first year that outputs maps.
MapYearFreq = {1, 0, 1, 0, 1, 0} ; The year frequency to output maps.

; STAT OUTPUT FLAGS
;0x000 Nothing is output.
;0x001 Output the data averaged by year.
;0x002 Output the data averaged by replicate.
;0x004 Show row titles in output. IE "Year", "Mean", "Min", "Max", etc.
;0x010 Output the number of samples.
;0x020 Output the mean of the data.
;0x040 Output the standard deviation of the samples.
;0x080 Output the minimum of the samples.
;0x100 Output the maximum of the samples.
;0x200 Output a histogram of samples.
```

```
;0x400 Output the sample data for the statistic. Note that sample data may already be an average or sum of event data. For example, FireSize sample data is the sum of all fires in a given rep and year.
;0x800 Output the event data for the statistic. Event data is a record of individual events rather than an average or sum.
; 0, 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15
; 0, 1, 2, 3, 4, 5, 6, 7, 8, 9, A, B, C, D, E, F

; STAT OUTPUT PARAMETERS
Stat.Summary.File = "StatOut.txt" ; The name of the statistical data output file (will be created in the output directory specified under the General settings).
Stat.FireNum.Flags = 0x421 ; Output flags for the fire num stats.
Stat.FireSize.Flags = 0xc21 ; Output flags for the fire size stats.
Stat.VegDist.Flags = 0x431 ; Output flags for the vegetation stats.
Stat.VegResidence.Flags = 0x231 ; Output flags for the vegetation residence times.
Stat.FireSpecies.Flags = 0x431 ; Output flags for the fire by species stats.
Stat.FireInterval.Flags = 0x831 ; Output flags for the fire interval stats.
Stat.BurnsBySupp.Flags = 0x431 ; Output flags for the burn partitions by suppression zone stats.
Stat.Habitat.Flags = 0x421 ; Output flags for the burn partitions by suppression zone stats.
```

```
; STAT COLLECTION PARAMETERS
Stat.Habitat.Types = {"Caribou80","Moose","Caribou50"} ; The ID's for a habitat statistic. For each ID, provide Stat.Habitat.<ID>.VegTypes and Stat.Habitat.<ID>.AgeRange. For each ID, Stat.Habitat.Flags will be used to dictate the output.
; EXAMPLE: Stat.Habitat.Example.VegTypes = {1,3,4} ; The vegetation type criteria for a given habitat statistic.
; EXAMPLE: Stat.Habitat.Example.AgeRange = {5,101} ; The non-inclusive lower and upper bounds for the frame age criteria of a given habitat statistic.
Stat.Habitat.Caribou80.VegTypes = {1,2,3}
Stat.Habitat.Caribou80.AgeRange = {80,10000000}
Stat.Habitat.Moose.VegTypes = {4}
Stat.Habitat.Moose.AgeRange = {10,31}
Stat.Habitat.Caribou50.VegTypes = {1,2,3}
Stat.Habitat.Caribou50.AgeRange = {50,10000000}

; CLIMATE PARAMETERS
Climate.IsMonthly = TRUE ; If true, Climate.TempMonths and PrecipMonths must be provided and the new fire equation based on monthly climate will be used. Monthly data does not work with SPATIAL or CONSTANT climate.value.types.
Climate.NumHistory = 20 ; The number of years to remember climate history including the current year.
```

Climate.TransitionYears = {0} ; The years in which fire type transitions occur.

Climate.Values.Type = {"EXPLICIT"} ; CONSTANT: Single temp and precip value used for every cell over all years and reps. SPATIAL: Read in spatially unique temp and precip values from a file once and reuse it each year and rep. EXPLICIT: Read in spatially unique temp and precip values from a different file each year. Year is appended as _YR

Climate.Values.Temp.Constant = {0.} ; Constant temperature value for entire simulation on all land cells.

Climate.Values.Precip.Constant = {0.} ; Constant precipitation value for entire simulation on all land cells.

Climate.Values.Temp.File = {"Input\Climate\HIST_1860_06p_08t\1kmt.tif"} ; Base filename for temperature maps.

Climate.Values.Precip.File = {"Input\Climate\HIST_1860_06p_08t\1kmp.tif"} ; Base filename for precipitation maps.

Climate.Values.RandExplicit.MinYear = {0} ; For use with RANDEXPLICIT climate type: Minimum simulation year for the random selection of climate temperature and precipitation maps.

Climate.Values.RandExplicit.MaxYear = {101} ; For use with RANDEXPLICIT climate type: Maximum simulation year for the random selection of climate temperature and precipitation maps.

Climate.Values.RandExplicit.Replicate = {FALSE} ; If true the RANDOMEXPLICIT climate type will apply the same random sequence of climate files every rep. Otherwise a different sequence is genereated per rep.

Climate.Flammability.File = "Input\gbm_flamm_clipped_12132010plus6\gbm.flamm.tif" ;Climate Flammibility Map path

Climate.Offsets.Type = {"NONE"} ; NONE: No offsets applied. CONSTANT: Single temp and precip offset used for all cells on all years and reps. FILE: Offsets are read in from a file. RANDOM: Offsets are generated stochastically.

Climate.Offsets.File = {"na"} ; The file to use as input/output when using the FILE offset type.

Climate.Offsets.Constant.Temp = {0.} ; The temperature offset used when for CONSTANT offset type scenarios.

Climate.Offsets.Constant.Precip = {0.} ; The precipitation offset used when for CONSTANT offset type scenarios.

Climate.Offsets.Random.Temp.Mean = {0.} ; The mean of the offset from climate temperatures. Mean is of course expected to be 0 as the mean difference from the mean for that cell should be 0.

Climate.Offsets.Random.Temp.StdDev = {0.} ; The standard deviation of the offset from mean climate temperatures.

Climate.Offsets.Random.Precip.Mean = {0.} ; The mean of the offset from mean climate precipitation.

Climate.Offsets.Random.Precip.StdDev = {0.} ; The standard deviation of the offset from mean climate precipitation.

Climate.Offsets.Random.Replicate = {FALSE} ; If true RANDOM offset type will apply the same stochastic offset sequence every rep. Otherwise a different sequence is genereated per rep.

Climate.StepsAndRampsEnabled = FALSE ; If true Ramps and Steps will be applied to the temporal offsets

Climate.Offsets.TempStepYear = {} ; The year that temperature steps occur. Must be a multiple of TempStep to occur.

Climate.Offsets.TempStep = {} ; The amount of the step that occurs in the specified year.

Climate.Offsets.TempRampYear = {} ; The year that temperature ramps change i.e. the new ramp is applied if the year is > the specified year.

Climate.Offsets.TempRamp = {} ; The ramp (amount per year) that starts in the specified year.

Climate.Offsets.PrecipStepYear = {} ; The year that precipitation steps occur. Must be a multiple of TempStep to occur.

Climate.Offsets.PrecipStep = {} ; The amount of the step that occurs in the specified year.

Climate.Offsets.PrecipRampYear = {} ; The year that precipitation ramps change i.e. the new ramp is applied if the year is > the specified year.

Climate.Offsets.PrecipRamp = {} ; The ramp (amount per year) that starts in the specified year.

```
; FIRE PARAMETERS
Fire.Types = {"SPATIAL"} ; FIXED: Single ignition and sensitivity factor used. SPATIAL: Read ignition and sensitivity factors from files. HISTORICAL: Burns are scheduled across the landscape.
Fire.TypeTransitionYears = {0} ; The years in which fire type transitions occur.
Fire.Enabled = TRUE ; Enable the fire model - usually set to true.
Fire.SpreadRadius = 1000 ; The maximum distance fire can spread (meters) - if CropNeighbours is set to false and this is less than CellSize it will check the adjacent cells only
Fire.SpreadParms = {0., 500.} ; The mean and stddev of the fire spread weighting function. Mean should always remain 0.
Fire.Climate = {0.0, 0.0, 0.0, 0.0, 0.0, 0.0, 0.0, 0.0} ; {6.775, 0.1, -0.08, 0.17, 0.35, -0.01275, 0.002, 0.005}Relationship between fire probability and climate. Results of a regression analysis.
Fire.IgnoreFirstInterval = TRUE ; Ignore the first fire interval for a cell when calculating statistics - used to help eliminate startup bias in statistical calculations.
Fire.IgnitionFactor = {5.0e-05} ; Fire ignition multiplier - used to tweak fire ignition rates
Fire.Sensitivity = {5500.0} ; General fire sensitivity parameter - used to tweak over fire size/number for a region
```

Fire.MaxEmpiricalFireSizeEvent = 3000 ; Maximum fire size event as observed in empirical data.

Fire.MaxEmpiricalFireSizeEventWeight = 1.0 ;.7 Weight to be applied to fire spread probability when fire size exceeds the maximum.

Fire.Historical = {""} ; Base filename for historical fire maps.

Fire.HumanIgnition.Basename = "Input\Fire\HumanIgnitions\HIg_Default_0.txt" ; Base filename for human igntion fire maps. Map Legend: 0 No human ignition 1 Possible human ignition (dependent on Human Ignition Prob).

Fire.Spatial.IgnitionFactor = {"Input\Fire\Spatial\I2_Calib.tif"} ;ORIG: I_ER_017_012_032_115.txt*was075 Filename for spatially explicit ignition map.

Fire.Spatial.Sensitivity = {"Input\Fire\Spatial\S2_Calib.tif"} ;ORIG: S_ER_325_325_235_305.txt*Filename for spatially explicit sensitivity map.

Fire.Suppression.On = FALSE ; Apply suppression maps to fire spread caclulations?

Fire.Suppression.Basename = "Input\Fire\Suppression\old_mngt.txt" ; Base filename for suppression maps. Map Legend: 1-5 where 1 Most Suppression and 5 Least.

Fire.Suppression.Tran.Years = {0,34,56} ; Year of suppression transitions (## '0' redundant here, but would make it easier to keep track of multiplier arrays below ##)

Fire.Suppression.Tran.HasNewMap = {TRUE,TRUE,TRUE} ; FALSE = do not input new map, TRUE = input new map (must be supplied w/ corresponding <base>_tran.year.txt)

Fire.Suppression.Tran.Class1 = {1.0, 0.05, 0.05} ; Highest suppression level. Used as a multiplier in the fire spread function.

Fire.Suppression.Tran.Class2 = {1.0, 0.15, 0.15} ; Used as a multiplier in the fire spread function.

Fire.Suppression.Tran.Class3 = {1.0, 0.6, 0.6} ; Used as a multiplier in the fire spread function.

Fire.Suppression.Tran.Class4 = {1.0, 1.0, 1.0} ; Used as a multiplier in the fire spread function.

Fire.Suppression.Tran.Class5 = {1.0, 1.0, 1.0} ; Lowest suppression level. Usually 1 to give no suppression. Used as a multiplier in the fire spread function.

Fire.Suppression.Tran.Threshold.FireSize = {28461, 28461, 28461} ; The cutting point where suppressions stop being applied because the sum of the year's fire sizes exceeds the capability of available suppression resources.

Fire.Suppression.Tran.Threshold.Ignitions = {500, 500, 500} ; The cutting point where suppressions stop being applied because the amount of ignitions for the year exceeds the capability of available suppression resources.

BurnSeverity.FxnOfFireSize = {-3, 0.00333} ; The intercept and slope for logistic regression fxn returning probability of High Severity (level 3 or 4)

BurnSeverity.LSS-vs-HSS.wt = .5 ; (0,1); Probability that a burn that is assigned High Severity has HSS. Low value is more likely to be LSS (level 3), high value is more likely to be HSS (level 4).

```
BurnSeverity.Low-vs-Moderate.wt = .5          ; (0,1); Probability that a burn not assigned High Severity has Moderate Severity. Low value is more likely to give Low Severity (level 1), high value is more likely give Moderate (level 2)
BurnSeverity.FlatTopo.wt = .5                 ; (0,1); Prob that Topo spatial correlation overrides initial Severity assignment; eg takes on Severity of 'Spreader cell'
BurnSeverity.ComplexTopo.wt = .2        ; (0,1); Prob that Topo spatial correlation overrides initial Severity assignment

; PARAMETERS FOR THE TUNDRA FRAME
Tundra = 1 ; The species number for this species
Tundra.StartAge = {40., 1.} ; Parameters for the starting age function. If one parameter is specified assume a constant distribution. Otherwise parameters are the lifetime and shape parameters for a Weibull distribution.
Tundra.FireProb.IsAgeDependent          = TRUE
        ;If TRUE: Tundra.FirePorb requires an array of three parameters {k,a,b} in that order for a logistic function k/(1+exp(a-b*x)) used to translate age to fire factor. If FALSE: Tundra.FireProb requires a single constant value (no array).
Tundra.FireProb = {0.75 ,1.5, 0.05 }        ;{0.80,6.0,0.1}0.77,0.73 Probablity of this species type burning.
Tundra.HumanFireProb = 2.432667E-05 ; Probability of human causes will cause this species type to burn.
```

Tundra.Spruce.EstBA = 23.5 ; Estimate of the basal area of mature forest - used to calculate seed dispersal for the initial BA assignment routine.

Tundra.SeedRange = 1000 ; Distance to the maximum seed source in meters.

Tundra.SeedSource = {35., 0.95} ; The mean travel distance(meters) and fraction of thin tail (1-fraction fat tail) in the distribution kernel.

Tundra.Seed.BasalArea = 70000 ; The number of seeds/Ha produced per unit basal area (m2/Ha).

Tundra.Seedling = 5000 ; Seed to seedling ratio including viability factor.

Tundra.SeedlingBA = 7.854E-05 ; Initial basal area of a seedling (m2) - based on a 5mm diameter

Tundra.History = 20 ; The number of years of climate that a deciduous frame uses to make decisions. Includes the current year.

Tundra.SeedEstParms = {7.5, 0.085} ; Seedling establishment parameters - cuttoff temp avg degree days/year cutoff

Tundra.MeanGrowth = 0.00044 ; Mean spruce growth (m) - this discounts the known trend in growth through time.

Tundra.ClimGrowth = {0., 0., 0.} ; Three coefficients (Int Temp Precip) for the relative growth factor - from a regression against climate.

Tundra.CalFactor = {0., 0.} ; Calibration factors to move from cohort growth model to exponential growth model - growth seed set

Tundra->Spruce.BasalArea = 10 ; Basal area at which tundra transitions to spruce (m2/Ha). A mature spruce stand is assumed to be about double this.

Tundra.IgnitionDepressor = 0.1 ; [optional]

Value should be between 0 and 1. Decreases the likelihood of ignitions for any cell of this species.

; PARAMETERS FOR THE BLACK SPRUCE FRAME

BSpruce = 2 ; The species number for this species

BSpruce.StartAge = {3.59, 0.87} ; A constant distribution is used if only one parameter is specified. Otherwise two parameters should be specified: the lifetime and the shape parameters for a Weibull distribution.

BSpruce.FireParms = {1.000, 4., 0.08} ;0.95,0.97,0.90, 0.92,0.97Parameters for the fire age function : k Max Fire Factor a Age Coefficient (Age a/b) b Slope Coefficient

BSpruce.HumanFireProb = 0.00119969 ; Probability of human causes will cause this species type to burn.

; PARAMETERS FOR THE WHITE SPRUCE FRAME

WSpruce = 3 ; The species number for this species

WSpruce.StartAge = {11.3, 2.36} ; A constant distribution is used if only one parameter is specified. Otherwise two parameters should be specified: the lifetime and the shape parameters for a Weibull distribution.

WSpruce.FireParms = {0.9, 8., 0.08} ; 0.87Parameters for the fire age function : k Max Fire Factor a Age Coefficient (Age a/b) b Slope Coefficient

WSpruce.HumanFireProb = 0.002184466 ; Probability of human causes will cause this species type to burn.

; PARAMETERS FOR THE DECIDUOUS FRAME

Decid = 4 ; The species number for this species

Decid.StartAge.WSpruce = {9.} ; A constant distribution is used if only one parameter is specified. Otherwise two parameters should be specified: the lifetime and the shape parameters for a Weibull distribution.

Decid.StartAge.BSpruce = {4.} ; A constant distribution is used if only one parameter is specified. Otherwise two parameters should be specified: the lifetime and the shape parameters for a Weibull distribution.

Decid.FireProb.IsAgeDependent = TRUE

;If TRUE: Decid.FirePorb requires an array of three parameters {k,a,b} in that order for a logistic function k/(1+exp(a-b*x)) used to translate age to fire factor. If FALSE: Decid.FireProb requires a single constant value (no array).

Decid.FireProb = {0.33, 6.0,0.1 } ;0.15,0.125(.125),(.03).25 Probability of this species type burning.

Decid.HumanFireProb = 0.003060109 ; Probability of human causes will cause this species type to burn.

Decid.History = 20 ; The number of years of climate that a deciduous frame uses to make decisions. Includes the current year.

Decid->Tundra.Parms = {4., 19.32} ; Degree-years are integrated from base A (i.e. temp < A). A transition occurs if the integrand exceeds B*(The maximum number of steps i.e. History/TimeStep).

; The following 8 fields are sets of two coefficients, {A, B}, used depending on the trajectory and last burn severity of the given decid cell.

; The chosen values are used in a linear function relating age to the probability of transitioning to white or black spruce: A*Age + B.

Decid->WSpruce.BurnSeverity[1] = {0.001428571, -0.07142857} ;(***50:120***){0.001428571, -0.07142857}current: 40:90, 1 = Low Severity 20:40 years

Decid->WSpruce.BurnSeverity[2] = {0.001000000, -0.07500000} ;(***75:150***)(***75:150***)current: 40:90, 2 = Moderate Severity 40:60 years

Decid->WSpruce.BurnSeverity[3] = {0.001000000, -0.09000000} ;(***90:170)current: 90:170, 3 = High Severity + Low Surface Severity 60:90 years

Decid->WSpruce.BurnSeverity[4] = {0.001000000, -0.09000000} ;(***90:170)current: 90:170, 4 = High Severity + High Surface Severity 90:130 years

Decid->BSpruce.BurnSeverity[1] = {0.001818182, -0.06363636} ;(**NEW 35:90***) {0.001666667, -0.03333333}(***20:75***)current: 40:90, 1 = Low Severity 5:15 years

Decid->BSpruce.BurnSeverity[2] = {0.001428571, -0.07142857} ;(***50:120***){0.001666667, -0.03333333}(***20:75***)({0.001818182, -0.06363636}***35:90***)current: 40:90, 2 = Moderate Severity 15:30 years

Decid->BSpruce.BurnSeverity[3] = {0.001000000, -0.07500000} ;(***75:150***){0.001818182, -0.06363636}(***35:90***)(was***50:120)current: 75:150, 3 = High Severity + Low Surface Severity 30:50 years

Decid->BSpruce.BurnSeverity[4] = {0.001000000, -0.07500000} ;(***75:150***){0.001818182, -0.06363636}(***35:90***)(was***50:120)current: 75:150, 4 = High Severity + High Surface Severity 50:90 years {0.002857143, -0.1428571}

;##### Calibration VII (prior to Precip fix)

;Decid->WSpruce.BurnSeverity[1] = {0.001428571, -0.07142857} ;(***50:120***){0.001428571, -0.07142857}current: 40:90, 1 = Low Severity 20:40 years

;Decid->WSpruce.BurnSeverity[2] = {0.001428571, -0.07142857} ;(NEW*50:120***)(***75:150***)current: 40:90, 2 = Moderate Severity 40:60 years

;Decid->WSpruce.BurnSeverity[3] = {0.001000000, -0.09000000} ;(***90:170)current: 90:170, 3 = High Severity + Low Surface Severity 60:90 years

;Decid->WSpruce.BurnSeverity[4] = {0.001000000, -0.09000000} ;(***90:170)current: 90:170, 4 = High Severity + High Surface Severity 90:130 years

;Decid->BSpruce.BurnSeverity[1] = {0.001818182, -0.06363636} ;(**NEW 35:90***) {0.001666667, -0.03333333}(***20:75***)current: 40:90, 1 = Low Severity 5:15 years

;Decid->BSpruce.BurnSeverity[2] = {0.001818182, -0.06363636} ; (**NEW 35:90***) {0.001666667, -0.03333333}(***20:75***)({0.001818182, -0.06363636}***35:90***)current: 40:90, 2 = Moderate Severity 15:30 years

;Decid->BSpruce.BurnSeverity[3] = {0.001428571, -0.07142857} ;(NEW*50:120***) {0.001818182, -0.06363636}(***35:90***)(was***50:120)current: 75:150, 3 = High Severity + Low Surface Severity 30:50 years

;Decid->BSpruce.BurnSeverity[4] = {0.001428571, -0.07142857} ;(NEW*50:120***) {0.001818182, -0.06363636}(***35:90***)(was***50:120)current: 75:150, 4 = High Severity + High Surface Severity 50:90 years {0.002857143, -0.1428571}

;#####FBX, SummReport Success Parms

;Decid->WSpruce.BurnSeverity[1] = {0.001428571, -0.07142857} ;(***50:120***)current: 40:90, 1 = Low Severity 20:40 years

;Decid->WSpruce.BurnSeverity[2] = {0.001000000, -0.07500000} ;(***75:150***)current: 40:90, 2 = Moderate Severity 40:60 years

;Decid->WSpruce.BurnSeverity[3] = {0.001000000, -0.07500000} ;(***75:150***)current: 90:170, 3 = High Severity + Low Surface Severity 60:90 years

;Decid->WSpruce.BurnSeverity[4] = {0.001000000, -0.07500000} ;(***75:150***), 4 = High Severity + High Surface Severity 90:130 years

```
;Decid->BSpruce.BurnSeverity[1] = {0.001428571, -0.07142857}
;(***50:120***)current: 40:90, 1 = Low Severity 5:15 years
;Decid->BSpruce.BurnSeverity[2] = {0.001000000, -0.07500000}
;(***75:150***)current: 40:90, 2 = Moderate Severity 15:30 years
;Decid->BSpruce.BurnSeverity[3] = {0.001000000, -0.07500000}
;(***75:150***)current: 75:150, 3 = High Severity + Low Surface Severity 30:50 years
;Decid->BSpruce.BurnSeverity[4] = {0.001000000, -0.07500000}
;(***75:150***)current: 75:150, 4 = High Severity + High Surface Severity 50:90 years
{0.002857143, -0.1428571}

;Decid->WSpruce.BurnSeverity[1] = {0.003333333, -0.3000000} ;1 = Low Severity 20:40 years
;Decid->WSpruce.BurnSeverity[2] = {0.003333333, -0.3000000} ;2 = Moderate Severity 40:60 years
;Decid->WSpruce.BurnSeverity[3] = {0.003333333, -0.3000000} ;3 = High Severity + Low Surface Severity 60:90 years
;Decid->WSpruce.BurnSeverity[4] = {0.003333333, -0.3000000} ;4 = High Severity + High Surface Severity 90:130 years

;Decid->BSpruce.BurnSeverity[1] = {0.001428571, -0.07142857} ;(***50:120)1 = Low Severity 5:15 years
```

```
;Decid->BSpruce.BurnSeverity[2] = {0.001428571, -0.07142857} ;2 = Moderate Severity 15:30 years
;Decid->BSpruce.BurnSeverity[3] = {0.001428571, -0.07142857} ;3 = High Severity + Low Surface Severity 30:50 years
;Decid->BSpruce.BurnSeverity[4] = {0.001428571, -0.07142857} ;4 = High Severity + High Surface Severity 50:90 years
```

RECREATE IGNITION/SENSITIVITY MAPS SCRIPT

```
# order of the regions in the function arguments
#spat.tmp==2]=seward peninsula nulato hills
#spat.tmp[spat.tmp==3]=northslope
#spat.tmp[spat.tmp==4]=southwestalaska
#spat.tmp[spat.tmp==1]=interior

# Create ignition geotiff maps
spat.fire.fx("I2_Calib",0.0000367,0.0000067,0.000201,0.0000379)
in.map<-raster("X:\\Western_Alaska_Files\\Input\\Fire\\Spatial\\I2_Calib.txt")
writeRaster(in.map,filename="X:\\Western_Alaska_Files\\Input\\Fire\\Spatial\\I2_Calib.tif",
options="COMPRESS=LZW",datatype="FLT4S",overwrite=TRUE)

# Create spatial geotiff maps
spat.fire.fx("S2_Calib",1934.4,1614.9,1100.0,2034.3)
in.map<-raster("X:\\Western_Alaska_Files\\Input\\Fire\\Spatial\\S2_Calib.txt")
writeRaster(in.map,filename="X:\\Western_Alaska_Files\\Input\\Fire\\Spatial\\S2_Calib.tif",
options="COMPRESS=LZW",datatype="FLT4S",overwrite=TRUE)

# Run calibration function
```

```
calib.HighRep.fx("X:\\Western_Alaska_Files\\Output\\2011-02-28_PM1558",50,maps=T)
```

SCRIPT TO FIND MOST REPRESENTATIVE RUNS

```
###########
## FIND 10 Most representatives reps based on 2001 con/decid ratio, and 2004/2005
Area Burn
## Use: AK_CalibVII_MultiRep.RData
#########

## collect data:
infile<-"X:\\Western_Alaska_Files\\Output\\final_50reps_draft_kyle_1860to2007"
numrep<-100
alf.fs<-
as.matrix(read.table(paste(infile,"\\FireSize.txt",sep=""),skip=1,header=F))[,2:(numrep+1
)]

##Find Interior Ratio
inter.ratio<-c()
library(raster)
for(i in 1:numrep){
#alf.veg2001<-as.matrix(read.table(paste(infile,"\\Maps\\Veg_",(i-
1),"_141.txt",sep=""),skip=6,header=F))
alf.veg2001<-raster(paste(infile,"\\Maps\\Veg_",(i-1),"_2001.tif",sep=""))
```

```
alf.veg2001<-getValues(alf.veg2001,format='matrix')
inter.ratio[i]<-length(alf.veg2001[(alf.veg2001==2 | alf.veg2001==3) &
spat.fire.templateII == 1])/length(alf.veg2001[(alf.veg2001==4) & spat.fire.templateII ==
1])
}

#1. compute absolute distance from empirical ratio, then RANK

ratio.rank<-rank(abs(inter.ratio-1.276874))

#2. compute absolute distance from empirical 04/05 AB, then RANK

ab2004.rank<-rank(abs(alf.fs[145,]-26239))
ab2005.rank<-rank(abs(alf.fs[146,]-18399))

#3. Sum ranks across the 3 metrics; find 10 lowest rank sum

total.rank<-ratio.rank+ab2004.rank+ab2005.rank

best.rep.matrix<-cbind(1:100,total.rank,rank(total.rank))

best.rep.matrix[,1][best.rep.matrix[,3]<=10]
```

```
cbind(best.rep.matrix[,1][best.rep.matrix[,3]<=10],best.rep.matrix[,3][best.rep.matrix[,3]
<=10])

###VIEW reps in context of histograms

win.graph(width=12,height=8)
par(mfrow=c(1,3))

xmin<-min(c(inter.ratio,1.276874))
xmax<-max(c(inter.ratio,1.276874))
hist(inter.ratio,xlim=range(xmin,xmax),col='dark gray',xlab='Interior Conifer to Decid
Ratio',main='2001 Conifer to Deciduous Ratio (Interior)')
abline(v=1.276874,lwd=4)
legend("topleft",c("Empirical Ratio"),bty='n',cex=1,lwd=c(4))
legend("topright","ALFRESCO Ratio (X 100)",cex=1,fill="dark gray",bty='n')
abline(v=inter.ratio[best.rep.matrix[,1][best.rep.matrix[,3]<=10]],lty=2,col=2)


hist(alf.fs[145,],main='2004 AB/Yr')
abline(v=26239,lwd=3)
abline(v=alf.fs[145,best.rep.matrix[,1][best.rep.matrix[,3]<=10]],col=2,lty=2)
```

```
legend("topright","10 selected Reps",lty=2,col=2,bty='n')
hist(alf.fs[146,],main='2005 AB/Yr')
abline(v=18399,lwd=3)
abline(v=alf.fs[146,best.rep.matrix[,1][best.rep.matrix[,3]<=10]],col=2,lty=2)
#################################################################################################
######
####################### Duplicate 10 best reps to create 100 rep input suite######
#################################################################################################
######


infile<-
"X:\\Western_Alaska_Files\\Output\\final_50reps_draft_kyle_1860to2007\\Maps\\"
outfile<-"X:\\Western_Alaska_Files\\Output\\2007Inputs_10bestTO100\\"

##1. identify 10 best; reorder based on rank

rep.rank.mat<-
cbind(best.rep.matrix[,1][best.rep.matrix[,3]<=10],best.rep.matrix[,3][best.rep.matrix[,3]
<=10])
# rep* rank ; *subtract 1 for alfresco maps
#V19 18 7
```

```
#V20 19 3

#V26 25 8

#V31 30 6

#V35 34 5

#V38 37 2

#V60 59 10

#V73 72 1

#V79 78 9

#V93 92 4



library(raster)

for (i in 1:10){


in.map.veg<-raster(paste(infile,"Veg_",(rep.rank.mat[,1][rep.rank.mat[,2]==i]-
1),"_2007.tif",sep=""))
in.map.age<-raster(paste(infile,"Age_",(rep.rank.mat[,1][rep.rank.mat[,2]==i]-
1),"_2007.tif",sep=""))
in.map.bsh<-
raster(paste(infile,"BurnSeverityHistory_",(rep.rank.mat[,1][rep.rank.mat[,2]==i]-
1),"_2007.tif",sep=""))
```

```
writeRaster(in.map.veg,filename=paste(outfile,"Veg_",(i-
1),"_2007.tif",sep=""),options="COMPRESS=LZW",datatype="INT1U")
writeRaster(in.map.age,filename=paste(outfile,"Age_",(i-
1),"_2007.tif",sep=""),options="COMPRESS=LZW",datatype="INT4S")
writeRaster(in.map.bsh,filename=paste(outfile,"BurnSeverityHistory_",(i-
1),"_2007.tif",sep=""),options="COMPRESS=LZW",datatype="INT1U")

}

##### 2007Inputs_10bestTO100 KEY:

#reps 0 to 9 <==> original ranks 1:10

# rep* rank NEWFOLDERASSIGN; *subtract 1 for alfresco maps
#V19 18 7 6
#V20 19 3 2
#V26 25 8 7
#V31 30 6 5
#V35 34 5 4
#V38 37 2 1
#V60 59 10 9
#V73 72 1 0
```

```
#V79 78 9 8
#V93 92 4 3


### Duplicate to 100 reps

library(raster)

inc<-c(10,20,30,40,50,60,70,80,90)

for(d in 1:9){

for (r in 0:9){

in.map.veg<-raster(paste(outfile,"Veg_",(r),"_2007.tif",sep=""))
in.map.age<-raster(paste(outfile,"Age_",(r),"_2007.tif",sep=""))
in.map.bsh<-raster(paste(outfile,"BurnSeverityHistory_",(r),"_2007.tif",sep=""))

writeRaster(in.map.veg,filename=paste(outfile,"Veg_",(r+inc[d]),"_2007.tif",sep=""),options="COMPRESS=LZW",datatype="INT1U")
writeRaster(in.map.age,filename=paste(outfile,"Age_",(r+inc[d]),"_2007.tif",sep=""),options="COMPRESS=LZW",datatype="INT4S")
```

```
writeRaster(in.map.bsh,filename=paste(outfile,"BurnSeverityHistory_",(r+inc[d]),"_2007
.tif",sep=""),options="COMPRESS=LZW",datatype="INT1U")


}

}
```

SCRIPT TO LIMIT ANALYSES TO CORE WINTER RANGE

```
#**# 1. Define path to mask:  X:\\data\\corewinter.txt
kyle.mask<-as.matrix(read.table("W:\\Kyle\\corewinter_TRY.txt",skip=6,header=F))
kyle.mask[kyle.mask==-9999]=0
#**# 2. What value designates masked area
region<-1 #**# what value is assigned to the subregion? change if necessary
#**# 3. 50 reps, right?
numrep<-50
##identify 5 best; reordered based on rank in final input folder
#REp  Rank (out of 50); *must subtract 1 for alfresco maps
# 13   5
# 14   4
# 17   2
# 23   1
# 30   3
best.5map<-c(22,16,29,13,12)
#**#Initialize variables (only extract now for '5best')
age.g50.v123<-matrix(NA,148,5)
age.g80.v123<-matrix(NA,148,5)
age.10to31.v4<-matrix(NA,148,5)
ab.core<-matrix(NA,148,5)
```

```
#**#4.Define path's to your output folders (example:  out<-
"W:\\Kyle\\Historic\\final_50reps_draft_kyle_1860to2007")
out.1860to2007<-"W:\\Kyle\\Historic\\final_50reps_draft_kyle_1860to2007"
#**#5. These will have to be changed to your new future output folders using '5best'
inputs
out.echam<- "X:\\Western_Alaska_Files\\Output\\2011-03-28_AM1021"
out.cgcm<- "X:\\Western_Alaska_Files\\Output\\2011-04-07_PM1204"

library(raster)
#**# 1860:2007
##TODO:
#1. code:  ab.core[y]<-length( age.mat[!is.na(age.mat)][age.mat[!is.na(age.mat)]==0 &
kyle.mask==region])
#2. stagger years; eg via ab.total[2:148]-ab.stat[1:147] ==> 0

for (r in 1:5){#numrep rep loop
 for (y in 1:148){#148year loop


 veg.tmp<-raster(paste(out.1860to2007,"\\Maps\\Veg_",best.5map[r],"_",(y-
1),".tif",sep=""))
 veg.mat<-getValues(veg.tmp,format='matrix')
```

```
 age.tmp<-raster(paste(out.1860to2007,"\\Maps\\Age_",best.5map[r],"_",(y-
1),".tif",sep=""))
 age.mat<-getValues(age.tmp,format='matrix')



     ab.core[y,r]  <-length( age.mat[!is.na(age.mat)][age.mat[!is.na(age.mat)]==0 &
kyle.mask[!is.na(age.mat)]==region])
     age.g50.v123[y,r] <-length( veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]<4 &
age.mat[!is.na(age.mat)]>50 & kyle.mask[!is.na(age.mat)]==region] )
     age.g80.v123[y,r] <-length( veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]<4 &
age.mat[!is.na(age.mat)]>80 & kyle.mask[!is.na(age.mat)]==region] )
     age.10to31.v4[y,r] <-length( veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]==4
& age.mat[!is.na(age.mat)]>10 & age.mat[!is.na(age.mat)]<31 &
kyle.mask[!is.na(age.mat)]==region] )

     print(c(y,r))
     }
     }
#write to .csv
write.table(
cbind(1860:2006,ab.core[2:148,]),file="W:\\Kyle\\CoreExtract_MO\\CoreRange.ABYr.t
xt",row.names=F,col.names=F,sep=",")
```

```
write.table(
cbind(1860:2007,age.g50.v123),file="W:\\Kyle\\CoreExtract_MO\\CoreRange.AgeG50.
V123.txt",row.names=F,col.names=F,sep=",")
write.table(
cbind(1860:2007,age.g80.v123),file="W:\\Kyle\\CoreExtract_MO\\CoreRange.AgeG80.
V123.txt",row.names=F,col.names=F,sep=",")
write.table(
cbind(1860:2007,age.10to31.v4),file="W:\\Kyle\\CoreExtract_MO\\CoreRange.Age10to
31.V4.txt",row.names=F,col.names=F,sep=",")

#**# 2008:2099
#**#Initialize variables
age.g50.v123<-matrix(NA,92,numrep)
age.g80.v123<-matrix(NA,92,numrep)
age.10to31.v4<-matrix(NA,92,numrep)
ab.core<-matrix(NA,92,numrep)
gcm.loop<-c(out.echam,out.cgcm)
gcm.name<-c("ECHAM5","CGCM31")
for(g in 1:2){
for (r in 1:numrep){#numrep rep loop
 for (y in 149:240){#240year loop
```

```
veg.tmp<-raster(paste(gcm.loop[g],"\\Maps\\Veg_",(r-1),"_",(y-1),".tif",sep=""))
veg.mat<-getValues(veg.tmp,format='matrix')
age.tmp<-raster(paste(gcm.loop[g],"\\Maps\\Age_",(r-1),"_",(y-1),".tif",sep=""))
age.mat<-getValues(age.tmp,format='matrix')



    ab.core[y-148,r] <-length( age.mat[!is.na(age.mat)][age.mat[!is.na(age.mat)]==0 &
kyle.mask[!is.na(age.mat)]==region])
    age.g50.v123[y-148,r] <-length(
veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]<4 & age.mat[!is.na(age.mat)]>50 &
kyle.mask[!is.na(age.mat)]==region] )
    age.g80.v123[y-148,r] <-length(
veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]<4 & age.mat[!is.na(age.mat)]>80 &
kyle.mask[!is.na(age.mat)]==region] )
    age.10to31.v4[y-148,r] <-length(
veg.mat[!is.na(age.mat)][veg.mat[!is.na(age.mat)]==4 & age.mat[!is.na(age.mat)]>10 &
age.mat[!is.na(age.mat)]<31 & kyle.mask[!is.na(age.mat)]==region] )

    print(c(g,y,r))
    }
    }
```

```
    write.table( cbind(2007:2098,ab.core[1:92,]),
file=paste("W:\\Kyle\\CoreExtract_MO\\",gcm.name[g],".CoreRange.ABYr.txt",sep=""),
row.names=F,col.names=F,sep=",")
 write.table( cbind(2008:2099,age.g50.v123),
file=paste("W:\\Kyle\\CoreExtract_MO\\",gcm.name[g],".CoreRange.AgeG50.V123.txt",
sep=""),row.names=F,col.names=F,sep=",")
 write.table( cbind(2008:2099,age.g80.v123),
file=paste("W:\\Kyle\\CoreExtract_MO\\",gcm.name[g],".CoreRange.AgeG80.V123.txt",
sep=""),row.names=F,col.names=F,sep=",")
 write.table(
cbind(2008:2099,age.10to31.v4),file=paste("W:\\Kyle\\CoreExtract_MO\\",gcm.name[g]
,".CoreRange.Age10to31.V4.txt",sep=""),row.names=F,col.names=F,sep=",")


    }
```