Relationship between Climate-Induced Changes and Spatial-Temporal Trends of the Bathurst Caribou Herd During Rapid Population Decline (1997-2017)

by

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Abstract

The Bathurst caribou herd is a population of migratory barren-ground caribou (*Rangifer tarandus groenlandicus*) that inhabits the Northwest Territories and Nunavut, Canada. Similar to many caribou populations globally, the Bathurst herd has experienced a dramatic population decline, from an estimated 472,000 animals in 1986 to 6,240 in 2021. Concerns have arisen that climate change and land-use changes may be altering habitat characteristics, leading to shifts in range distribution. The summer range on the sub-Arctic tundra is crucial for caribou to access nutritious forage after long winter months and calving, and before the fall breeding season. This research analyzed collar telemetry data obtained by the Government of the Northwest Territories to assess the summer range distribution of Bathurst caribou from 1997 to 2017. Annual summer range distribution, home range extents, and core-use areas were determined using kernel density estimation for each year's data. Subsequently, a Theil-Sen regression analysis identified spatial trends in summer range distribution. These analyses revealed a significant northward shift and contraction of the herd over time, with increased use northwest of Contwoyto Lake. Results of the Theil-Sen analysis, combined with various climatic and environmental variables, informed the development of two random forest models examining the influence of habitat characteristics on changes in caribou use. Results indicated that increases in temperature trends above 0.10 degrees Celsius, delayed snow melt timing, delayed SOS and increased maximum EVI were all associated with decreases in relative summer habitat use by the herd. The findings of this study provide valuable insights for land-use decisions and the establishment of protected areas within the herd's range, aiming to prevent extinction.

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List of Abbreviations

AKDE	Autocorrelated kernel density estimation
ANN	Artificial neural network
BRT	Boosted regression tree
CARTs	Classification and regression trees
CPI	Conditional permutation importance
EOS	End of season
EVI	Enhanced vegetation index
GNDVI	Green normalized difference vegetation index
GNWT	Government of the Northwest Territories
GPS	Global positioning system
IncMSE	Increase in mean squared error
IncNodePurity	Increase in node purity
KDE	Kernel density estimation
LoCoH	Local convex hull
LOS	Length of season
M.A.S.L	Meters above sea level
MCP	Minimum convex polygon
MLR	Multiple linear regression
MODIS	Moderate resolution imaging spectroradiometer
mtry	Number of predictor variables to be tried at each split
NDVI	Normalized difference vegetation index
NIR	Near-infrared
ntrees	Number of bootstrapped samples
NWT	Northwest Territories
OOB	Out of bag
SOS	Start of season
TIE	Time-integrated enhanced vegetation index
UD	Utilization distribution

Chapter 1: General Introduction

1.1 Research Context

Arctic and sub-Arctic regions are particularly susceptible to climatic and environmental changes due to polar amplification, which results in temperature increases at high latitudes being significantly greater than the global average (Rantanen et al., 2022). The 20th century has been identified as the warmest in the last millennium (Folland et al., 2001), and the warming trend in the Arctic is projected to continue at twice the global average (IPCC, 2021). This has and will continue to induce profound alterations in a range of environmental conditions. Some of the changes already observed include increased frequency and intensity of wildfires (Flannigan et al., 2009), declines in snow cover and permafrost (Zhang et al., 2019), increased vegetation productivity in regions where forest fires are not prevalent (Dearborn & Danby, 2021), and extended growing seasons, due to an earlier onset of vegetation growth and delayed senescence (Post et al., 2009; Dearborn & Danby, 2021).

Many species of wildlife have been impacted by these drastic climatic and environmental changes. Caribou (*Rangifer tarandus*), known as reindeer in Eurasia, have experienced global population declines and changes in range distribution that are speculated to be related to environmental changes, including changes in plant and insect phenologies, increased frequency of extreme weather events, and overall warming temperatures (Vors & Boyce, 2009). The barren-ground caribou subspecies (*Rangifer tarandus groenlandicus*) is found throughout Alaska, northern Canada, and Greenland (COSEWIC, 2016). These caribou undertake the longest terrestrial migrations of any land mammal (Bergman et al., 2000), with their seasonal ranges differentiated primarily by forage availability and predation (Hebblewhite & Merril, 2009). This migratory behavior is a strategy of caribou to adapt to diverse environmental conditions, migrating between the Arctic tundra for calving, summer and fall ranges and the boreal forest in winter.

Barren-ground caribou play a crucial role in the regulation and ecology of plant and predator relationships in northern ecosystems (Musiani et al., 2007; Bernes et al., 2015). Specifically, through grazing, trampling, and defecation, caribou can suppress plant growth and reproduction (Kumpula et al., 2011), as well as actively contribute to nutrient cycling (Olofsson et al., 2004; Post & Klein, 1996). Additionally, they are integral to the traditional livelihoods of northern Indigenous communities, providing sustenance, cultural value, and income (Miller, 2003). The decline in both abundance and distribution of various herds poses significant threats to northern environments, making the conservation of their habitat critical.

1.2 Research Rationale

The Bathurst caribou herd, located in the Northwest Territories (NWT) and Nunavut, Canada, has experienced a drastic population decline in recent years. Specifically, the herd has declined by 98% since the 1980s, from approximately 470,000 in the mid-1980s to an estimated 6,240 individuals in 2021 (GNWT, 2019). While barren-ground caribou herds naturally fluctuate between population highs and lows (Zalatan et al., 2006), the decline of the Bathurst herd is unprecedented in recorded history (Kendrick et al., 2005). According to the "selfish herd theory" (Hamilton, 1971), caribou typically benefit from large herd sizes and tight herd formation as it increases their defense against adverse environmental factors, such as the ability to cope with insect harassment, move in deep snow, and escape predation (Pruitt, 1960; Nixon and Russell, 1990). Therefore, low population size makes individuals especially vulnerable to environmental changes, potentially leading to local population extinctions (Mallory & Boyce, 2018). Both Indigenous elders in the region (Kendrick et al., 2005) and scientific research (Virgl et al., 2017) have noted that the decline of the Bathurst herd has been accompanied by changes in typical range use patterns. Understanding how the herd's range is changing and identifying potential influencing factors is crucial for conservation efforts aimed at mitigating the herd's population decline.

Given the highly synchronized migratory patterns of caribou to access nutritious forage during calving, avoid predation, and minimize insect harassment, climate and environmental changes throughout the circumpolar north have direct and indirect implications on the herd (Bergman et al., 2000; GNWT, 2019). Some of these implications include changing plant and insect phenology (Post et al., 2009), vegetation composition and productivity (Myers-Smith et al., 2011; Dearborn & Danby, 2021), extreme weather events (Barrier & Johnson, 2012) and overlap in prey and predator distribution (Vors & Boyce, 2009).

Several studies have examined animal movement characteristics and range changes during the Bathurst herd's decline, as well as identifying the typical seasonal ranges of the herd (Virgl et al., 2017; Mennell, 2021). However, what remains unknown is where the most extensive changes have occurred within the range. Identifying these regions can be challenging due to the widespread distribution of the herd, especially on the winter range due to varied herd structure during that time of year. However, during the summer months, from calving to fall migration, caribou typically band together to avoid insect harassment and predation (Pruitt, 1960; Hughes et al., 2009; Klaczek et al., 2016). Identifying regions of change, specifically increased utilization, during the growing season is crucial, as the intake of nutritious forage during this time is vital for lactating females and newborn calves (Bergerud, 1972). Identifying consistently utilized regions is the first step in facilitating the protection and management of areas critical to the herd's survival.

Moreover, numerous environmental and habitat-related changes at high latitudes influence the seasonal range extent and distribution of the herd. While many studies (e.g., Boulanger et al., 2012; Barrier & Johnson, 2012) have investigated specific environmental factors potentially affecting caribou range use (i.e. transportation corridors), they often overlook the broader combination of influences potentially impacting caribou. These studies include investigations into forest fire occurrence on the winter range (Barrier & Johnson, 2012), shrub proliferation (Bonta et al., 2023), increased vegetation productivity (Dearborn & Danby, 2021), infrastructure avoidance (Boulanger et al., 2012), predator distribution (Klaczek, 2015) and trophic mismatches (Chen et al., 2018). However, there is a gap in the research assessing how seasonal ranges have changed during population decline and how these changes coincide with recent climate and environmental changes. Understanding this combined influence is essential as environmental changes continue, helping to predict how the herd will adjust its distribution in response. Additionally, assessing changes in habitat utilization in response to environmental factors will reveal which habitat preferences are most important to the herd, thereby guiding conservation and management efforts.

1.3 Goals and Objectives

The overarching goal of this research is to understand how the distribution and extent of the Bathurst caribou herd's summer range have changed during a period of rapid population decline (1997 to 2017), and to relate these changes to environmental and habitat characteristics during the same period. GPS collar data collected from select individuals since 1997 is used to identify trends in annual relative summer range distribution and a suite of environmental data, sourced from raw satellite imagery and modeled environmental characteristics, is employed to assess the relationships between changes in range use and habitat. Ground-truthing was conducted to validate the satellite imagery used in this study. The specific objectives are:

1) Analyze collar telemetry data from 1997 to 2017 to determine if and how the relative summer range distribution and extent of the Bathurst herd have changed during its recent population decline.
2) Use results of a recent remote sensing analysis as well as synoptic climatic data to ascertain if there are relationships between environmental changes and regions of increasing or decreasing caribou utilization.

Identifying a relationship between relative summer range distribution and habitat changes would support the hypothesis that climate and/or environmental changes have contributed to the decline of the Bathurst caribou herd.

1.4 Thesis Outline

This thesis is structured in manuscript form and comprises four chapters that address the objectives of this study.

- Chapter 1 (this chapter): Provides the context and rationale for this study, emphasizing the significance of this research for the conservation and management of barren-ground caribou, and identifies the specific objectives of this study.
- Chapter 2: Reviews the current literature related to climate change, barren-ground caribou, and the modeling approaches commonly employed in ecological studies.
- Chapter 3: Details the methods, presents the results, and discusses the findings in relation to other studies addressing similar topics.
- Chapter 4: Concludes the research, highlighting the new findings and discussing their implications for future research and sustainable management practices.

Chapter 2: Literature Review

2.1 Global Change in the North

Arctic and sub-Arctic regions are particularly vulnerable to environmental changes due to polar amplification, where warming at high latitudes exceeds the global average (Rantanen et al., 2022). This is primarily driven by anthropogenic greenhouse gases and is further exacerbated by positive feedback mechanisms such as reduced sea ice cover (Kumar et al., 2010). As ice melts, the loss of highly reflective surfaces allows for increased solar absorption, leading to further warming (Serreze et al., 2009). Temperature warming in the 20th century is likely the greatest in the past millennium and is expected to continue at twice the global average (Folland et al., 2001; Rantanen et al., 2022; IPCC, 2021). Changes in temperature result in environmental changes that can have profound impacts on ecosystems (Parmesan & Yohe, 2003), including more frequent and intense wildfire events (Flannigan et al., 2009), altered precipitation and snow cover dynamics (Zhang et al., 2019) and shifting plant phenology and productivity regimes (Dearborn & Danby, 2021).

2.1.1 Forest Fires

Forest fires are common disturbances, with 2.5 million hectares burned annually in Canada since 1990 (Natural Resources Canada, 2024). Within Canada, boreal forest fires are prevalent (Kasischke & Turetsky, 2006), and although tundra fires are less common, they can cause vegetation composition shifts (Bret-Harte et al., 2013) dominated by deciduous shrubs or grasses (Racine et al., 2004). Climate change, accompanied by increased temperatures and a longer growing season, is expected to increase the frequency, area burned, duration, and severity of forest fires in this century (Flannigan & Van Wagner, 1991; Ali et al., 2012). Given the variability in landscapes and weather across Canada, there will be significant spatial and temporal variations, with a possible decrease in fire severity in some regions (Lewis et al., 2019), making it difficult to predict how forest fires will affect various ecosystems. Severe fires in forested regions eliminate soil organic layers, melt permafrost, and can shift ecosystems from spruce-dominated to deciduous-dominated (Chapin et al., 2010), which will have trickle down effects on the entire ecosystem. Weber & Flannigan (1997) argued that a changing forest fire regime may even be more significant than the direct effects of climate change (e.g. warmer temperatures) in altering the distribution, migration and even the extinction of species that rely on the ecosystem.

2.1.2 Precipitation and Snow Patterns

Circumpolar Arctic precipitation is expected to increase by over 50% by 2100, leading to shifts from snow to rain events (Zhang et al., 2019). Significant reductions in snow cover have been observed by both satellite and in-situ observations, and are projected to continue in the 21st century (IPCC, 2021). Snow depth can vary at the local scale due to variability of vegetation and topography that influences snow properties and the amount of snow accumulating on the ground (Neumann et al., 2006). The phenology of snow cover can also have a prominent influence on plant phenology and growth (Semenchuk et al., 2016), where in spring, the start of new vegetation growth follows snow melt (John et al., 2020) and in autumn, snow accumulation is critical to soil thermal regimes (Lafreniere et al., 2013). In addition, permafrost degradation has occurred in most regions around the world since the early 1880s and is projected to decrease further (IPCC, 2021). This can result in increased water cover (Derksen et al., 2019) and create a muddy sludge that cannot support the weight of soil or vegetation, possibly leading to decreases in the number of plant species and loss of sensitive populations (Huntington et al., 2023; IPCC, 2021).

2.1.3 Vegetation Dynamics

The effects of climate change are largely evident through altered structure, productivity, composition and phenology of vegetation (Price et al., 2013). Dominant biomes in the circumpolar north include boreal forest and tundra, each with unique vegetation dynamics associated with abiotic (e.g., temperature, precipitation, soil moisture) and biotic (e.g., herbivory) factors (Woodward, 2009). Vegetation changes vary significantly across time and space according to terrain, landscape, natural vegetation and varying climates (Danby & Hik, 2007). These changes can occur gradually, due to warming temperatures (Myers-Smith et al., 2015), reduction in sea ice (Bhatt et al., 2010), precipitation changes (Lotsch et al., 2005), and permafrost degradation (Nauta et al., 2015), or abruptly, through events such as forest fires (Heim et al., 2019) and industrial development (Walker et al., 2011).

One of the most widespread changes to vegetation are increased vegetative productivity, or ‘greening’, on the tundra biome and this has been observed through remote sensing (Bonney, Danby & Treitz, 2018), experimental warming (Zamin et al, 2017), and repeat photography (Fraser et al., 2014). However, this is not homogenous across space (Bonney, Danby & Treitz, 2018), where the boreal forest biome has exhibited decreased productivity, termed ‘browning’, due to drought stress (Lotsch et al., 2005) and forest fires (Dearborn & Danby, 2021). Remote sensing indices, such as Normalized Difference Vegetation Index (NDVI) and Enhanced Vegetation Index (EVI), are highly correlated with above ground vegetation biomass, making them useful for quantifying widespread changes in vegetation productivity (Huete et al. 2002). NDVI and EVI are both based on the difference between near-infrared and red light reflectance, while EVI differs by correcting for the influence of bare soil and aerosol (Tucker, 1979; Huete et al., 2002). Researchers using both indices of plant productivity in Arctic and sub-Arctic regions have noted significant increases in productivity trends on the tundra biome, while the boreal forest exhibited decreases or slight increases (Bonney, Danby & Treitz, 2018; Fraser et al., 2014; Dearborn & Danby, 2021). Productivity increases were well-correlated with increases in temperature (Bonney, Danby & Treitz, 2018; Myers-Smith et al., 2011), landcover, proximity to drainage systems and lower elevations (Bonney, Danby & Treitz, 2018).

There are various explanations for these changes in vegetation productivity. Shrubification is the expansion of erect deciduous shrubs, such as alder (*Alnus* spp.), dwarf birch (*Betula* spp.) and willow (*Salix* spp.), and has been observed to be strongly correlated with warming temperatures (Forbes et al., 2010), increased soil moisture, increased nutrient (Myers-Smith et al., 2011; Mekonnen et al., 2021), growing season length (Neigh et al., 2008) and the reduction of non-vascular plants (Zamin et al., 2017). However, some studies point to a decline in vegetation trampling and browsing by herbivores, such as caribou, as a reason for shrub expansion, indicating this may not be a result of climate change directly (Andruko et al., 2020). Additionally, birch shrubs produce resin, which is a toxic compound that disrupts the citric acid cycle needed for metabolism and to generate energy (McLean et al., 2009). This acts to deter herbivory and may contribute to shrub expansion by providing a competitive advantage over non-resin forage favored by caribou and muskoxen (Bryant et al., 2013)

Another explanation for increases in plant productivity is due to altered vegetation phenology, which has been observed across much of the Arctic tundra biome (Post et al., 2009). Specifically, an increase in the length of the growing season, given by an earlier start of season (SOS) and a shorter winter, is a common explanation for increases in vegetation productivity (Goetz et al., 2005; Post et al., 2009; Dearborn & Danby, 2021). However, these changes are very inconsistent across space, and green-up is often constrained by snow cover in warm, moist, coastal and mountainous regions or by temperature in cold, dry inland regions (John et al., 2020; Fauchald et al., 2017). However, warming is not a conclusive explanation for an earlier SOS, as increased precipitation in the fall and winter may cause a later SOS (Pouliot et al., 2009).

2.2 Global Caribou Ecology

Caribou, or reindeer (*Rangifer tarandus*), are a large, herbivorous species found throughout northern North America, Europe, and Asia (Hummel & Ray, 2008). In North America, caribou are divided into subspecies based on movement characteristics, location, and ecotypes (Festa-Bianchet et al., 2011). In Canada, the recognized subspecies include mountain/woodland caribou (*Rangifer tarandus caribou*), peary caribou (*Rangifer tarandus pearyi*), and barren-ground caribou (*Rangifer tarandus groenlandicus*; Festa-Bianchet et al., 2011). For the purpose of conservation management, aggregations of caribou are further divided into herds based on fidelity to calving grounds and congregation on seasonal ranges (Miller, 1982; Bergerud, 2000).

On a global scale, caribou have experienced significant population declines in recent decades; however, the precise cause remains unexplained and is likely the result of cumulative impacts (Vors & Boyce, 2009; Festa-Bianchet et al., 2011). Vors & Boyce (2009) reviewed population estimates for 58 herds to assess mechanisms by which climate change and anthropogenic land-use change have influenced their population. This study found that 34 herds were declining, 16 had no population data and eight were increasing. On average, declines were 57% from a known population maximum and were influenced by various mechanisms including phenology changes, including forage availability and insect harassment; spatial-temporal changes in species overlap, including increased predation access and immigration/emigration between different herds; increased frequency and intensity of extreme weather events, such as freezing rain; and overall warming temperatures. In addition, Indigenous caribou hunters have observed that changes in weather patterns, forest fires and industrial developments have negatively impacted range distribution and caribou body condition at northern latitudes (Kendrick et al., 2005).

2.2.1 Barren-Ground Caribou

For the remainder of this literature review, I will be specifically focusing on the barren-ground caribou subspecies in North America. Barren-ground caribou are a migratory subspecies found from Alaska to Greenland, with continuous presence across northern Canada (COSEWIC, 2016). Known for their extensive migrations, barren-ground caribou travel between boreal forests in winter and Arctic tundra in summer, demonstrating the longest terrestrial migrations of any wingless vertebrate (Bergerud, 2000; Bergman et al., 2000). Similar to other migratory ungulates, such as wildebeest (Berger, 2004), migratory caribou herds demonstrate vast spatial and temporal shifts driven by forage availability and predation (Hebblewhite & Merrill, 2009).

In North America, there are many different herds (*Figure 1*) that are consistently recognized through fidelity to calving areas in the Arctic tundra; however, the exact number of herds at any one time is unknown due to constantly evolving information (COSEWIC, 2016). For example, the Fortymile herd in Alaska is often not considered a barren-ground caribou herd, as its seasonal migration to the calving grounds are not north of the treeline, but rather to higher altitudes (COSEWIC, 2016). As a result of vast range extents and seasonal migrations by caribou, conservation strategies can be incredibly difficult (Berger, 2004) as they are exposed to many different habitat conditions.

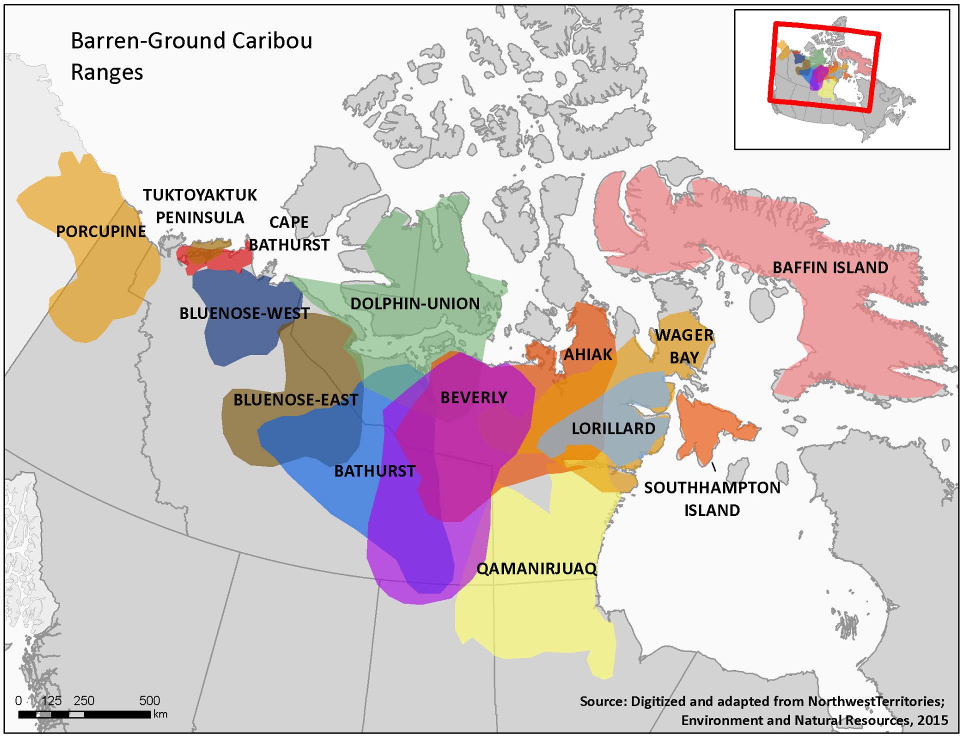

Figure 2.1: The range extents and distribution for all barren-ground caribou herds in central North America (Canada & Alaska) (Image Source: WWF Canada, 2021)

2.2.2 Barren-Ground Caribou Significance

Given widespread distribution and abundance, barren-ground caribou play a crucial role in the regulation and ecology of northern boreal and Arctic regions. Caribou influence plant communities by actively contributing to the nutrient-limited boreal soils through grazing, defecation and trampling (Post & Klein, 1996; Olofsson et al., 2004), which enhances plant growth and affects vegetation composition (Post & Klein, 1996). In contrast, abundant caribou populations can suppress vegetation growth, biomass, production, and reproduction through persistent grazing (Kumpula et al., 2011). Caribou are also key prey for predators such as wolves, grizzly bears and golden eagles, playing a significant role in predator-prey dynamics (Frame et al., 2008). For example, reduced caribou populations were seen to be accompanied by decreases wolf reproduction and pup recruitment, suggesting a positive relationship between wolf and caribou demographics (Klaczek et al., 2016).

Culturally and socio-economically, caribou are vital to northern Indigenous and non-Indigenous communities. Indigenous people, including Dene and Inuit, have relied on caribou for sustenance, cultural practices, and economic value for thousands of years (Gordon, 2005). There is an understanding of the importance of respectful, reciprocal relationships between animals and people, whereby caribou are relied on for sustenance, economic reliance and traditional livelihoods, and communities are responsible for protecting their habitats and treating the animals with dignity and respect (Kendrick et al., 2005). Economically, the annual harvest of caribou holds significant value, such as the Beverly and Qamanirjuaq herds in Canada that was valued at 17.5 million CAD (Beverly and Qaminirjuaq Caribou Management Board, 2008). Northern communities have noticed an "unprecedented shift" in the distribution and abundance of caribou herds (Kendrick et al., 2005), and they are likely being impacted by significant climate variations that will have devastating impacts on hunting, economic reliability and traditional ways of life (Sharma et al., 2009; Gunn et al., 2011a).

2.2.3 Population Demography

Barren-ground caribou herds exhibit natural, large-scale population fluctuations over decades (Zalatan et al., 2006; Gunn et al., 2011b), influenced by weather, forage availability, and Arctic Oscillations (White, 1983; Zalatan et al., 2006). These cycles can span 20-80 years, with high abundance correlated with positive phases of the Arctic Oscillation, which coincides with increased atmospheric pressure, temperature, and precipitation (Zalatan et al., 2006). Population fluctuations are somewhat synchronized across herds (Vors & Boyce, 2009).

The survivorship of demographic classes, including calves (0 to 8 months), yearlings (1 to 2 years), cows (females over two years old) and bulls (males over two years old), are heavily associated with caribou abundance (Gaillard et al., 1998; Boulanger et al., 2011). Similar to other large herbivores, population demography is extremely sensitive to changes in cow survival (Gaillard et al., 1998; Adamczewski et al., 2022), with a survival rate estimated to be between 84% and 87% for a stable population (Boulanger et al., 2011). Furthermore, emigration and immigration between herds both influence and is influenced by population demography. This likely impacted the Red Wine Mountains boreal caribou herd, where five of 36 radio-collared caribou emigrated to the George River herd in the 1990s (Schaefer et al., 1999). However, an analysis of movement of the Bathurst herd in Canada found that rates of emigration and immigration by individuals were relatively equal (<5%) (Boulanger et al., 2011), indicating that movement between herds is not the primary explanation for declines in the Bathurst herd.

The history of population fluctuations for barren-ground caribou are relatively uniform across North America (Vors & Boyce, 2009). For example, during the 1950s to the 1970s, major caribou herds (Porcupine, Cape Bathurst, Bluenose-West, Bluenose-East, Bathurst, Qamanirjuaq, Southhampton, George River and Leaf River) in Canada exhibited low population numbers and then began steadily increasing after 1970 to an estimated two million individuals by 1995 (Gunn et al., 2011b). However, this was followed by dramatic declines of most caribou herds, resulting in an estimated 800,000 individuals by 2015 (COSEWIC, 2016). Specifically, population estimates from 1984 to 2015 of seven herds in Canada were recently reviewed, and it was determined four herds (Cape Bathurst, Bluenose-East, Bluenose-West and Bathurst) had declined by >80%, one herd (Qamanirjuaq) declined by 39.7%, while two (Porcupine and Southhampton) were seen to be increasing (COSEWIC, 2016). Low population densities and abundance make caribou particularly vulnerable to anthropogenic factors and extreme weather events, which can exacerbate declines and potentially lead to local extinctions (Mallory & Boyce, 2018).

2.2.4 Reproduction

Barren-ground caribou have evolved flexible reproductive strategies, based on energy expenditures and forage availability, to cope with harsh northern environments (COSEWIC, 2016). Female caribou make trade-offs between their own survival and that of their offspring based on body weight and forage availability (Barboza & Parker, 2009). High body weight in females correlates with increased reproduction rates, and cows can give birth as early as two years old, although many reproduce for the first time at age three (Gerhart et al., 1997; Thomas & Kiliaan, 1998). Synchronous rutting activity that occurs in autumn of each year results in mostly synchronous calving in June, with up to 50% of a herd giving birth on the same day (Chen et al., 2018), to ensure calves and lactating cows have access to nutritious forage (Bergerud et al., 2008).

Mature bulls become antagonistic towards each other before joining cow groups to mate during the fall rut (Pruitt, 1960; Lent, 1965). Reproductive lifespans are typically around 12 years, with cows giving birth to one calf each year (Thomas & Killiaan, 1998). Calf recruitment and cow survivorship are closely linked, with environmental factors such as nutritious forage availability playing a crucial role in calf and cow survival (Bergerud, 2000). Inadequate nutritious forage has been seen to result in lower pregnancy rates (Rettie & Messier, 1998), poor cow body conditions (Couturier et al., 1990) and smaller calf body mass at birth (Couturier et al., 2009).

2.2.5 Seasonal Migration and Herd Structure

Barren-ground caribou exhibit large-scale migratory behavior and require extensive spatial extents, ranging from 5,700 km² (Coats Island Herd) to 460,000 km² (Qamanirjuaq Herd) (Bergman et al., 2000; GNWT, 2019). Migratory behaviour and herd structure are key adaptations allowing individuals to access various habitat conditions, influenced by weather, forage preferences, land-use and predation (Hebblewhite & Merrill, 2009). Movement between different seasonal grounds are highly synchronized and driven amongst females (Maier & White, 1998). Furthermore, herd structure has evolved to escape predation from wolves, specifically by forming groups ranging from 10 to over 50,000 when caribou are highly vulnerable to predators (such as calving; Pruitt, 1960). This has been shown to increase survivorship, especially of calves as they are less likely to be targeted by wolves (Bergerud, 1972; Gunn et al., 2012; Hansen et al., 2013).

There are several distinct seasonal ranges (winter range, calving grounds, summer range and fall range) and migratory routes (spring migration, post-calving movements, and fall migration) (Bergman et al., 2000; Nagy, 2011; Mennell, 2021). Most herds of barren-ground caribou migrate between the Arctic tundra for calving, summer and fall rut and the boreal forest for winter (GNWT, 2019). The distribution and extent of seasonal ranges for different herds vary. However, during winter, there is significant overlap in the distribution of both migratory and sedentary herds (Boulet et al., 2007). The winter range, typically from November to April, is considerably larger than other seasonal ranges, and the use of specific habitat can vary depending on forest fires, forage accessibility and snow depth (Nagy, 2011; Joly et al., 2010). Winter ranges are typically below the treeline, where caribou form small groups (<10 individuals) and remain relatively sedentary to optimize their fitness and access forage (Pruitt, 1960; Mennell, 2021).

As snow depth decreases and days become longer, in March or April, pregnant cows and yearlings lead the long spring migration to calving grounds (Calef, 1981; Parker, 1972; Miller, 2003), characterized by rapid movement (ranging from 7 to 24 km per day) towards the Arctic coast (Parker, 1972; Miller, 2003). During this migration, cows and yearlings aggregate to form small to medium sized groups (10 to 100 individuals; Pruitt, 1960). In early June, pregnant cows arrive at the calving grounds, while males and non-pregnant cows do not migrate as far north, but rather stop within the summer range (GNWT, 2019). Arrival at the calving grounds and subsequent calving is highly synchronous with the emergence of nutritious forage at the start of the growing season (Post, 2019). At this time, there is a critical need for nutritious forage after a long spring migration, to support cow lactation and calf sustenance (Nagy, 2011; Bergerud, 1972). The calving grounds are characterized as a remote tundra landscape with sparsely vegetated land that caribou access to avoid predation (Heard and Williams, 1992; Bergerud et al., 2008) and escape insect harassment (Morschel & Klein, 1997). During this time, caribou congregate, with greater than 50,000 individuals for some herds, and exhibit extremely low movement rates until calving is completed (Pruitt, 1960; Adamczewski et al., 2022).

In late June and early July (Nagy, 2011; GNWT, 2019; Mennell, 2021), individuals begin to disperse, moving south to their summer range (Calef, 1981). This occurs when continual movement is necessary to escape insect harassment and the ability to nurse, forage and rest is limited as a result (Toupin et al., 1996). In late summer, from July to early September, cows and bulls arrive to the summer grounds (Pruitt, 1960; Mennell, 2021). This range is an extremely important time for calves to access abundant, high-quality forage (Couturier, 2007) and disperse to regions of reduced insect densities (Morschel & Klein, 1997). At this time, new mothers and calves form small to medium sized groups (10 to 100 individuals) to maximize the survivorship of calves (Pruitt, 1960; Lent, 1965), while bulls generally form smaller groups that are separated from cow aggregations (Pruitt, 1960).

After summer, typically in early-September, caribou begin their southern movement to the fall rutting grounds for mating season (Lent, 1965; Miller, 2003; Nagy, 2011). At this time insect harassment has subsided, and caribou reduce their movement rates and form small groups to focus on gaining fat necessary for winter survival (Bergerud, 1972). The fall rut typically takes place in October and is highly synchronous within each herd (Dauphine & McClure, 1974). The final movement of the year, after fall rut, is the fall migration to the winter range in the boreal forest (Mennell, 2021), where softer snow cover allows for higher forage accessibility than on the tundra (Festa-Bianchet et al., 2011). The timing of this migration is less predictable than that of the other seasonal ranges, as it is largely based on environmental conditions, such as the timing of snow fall (McNeil et al., 2005).

Recent scientific literature and Indigenous Knowledge have documented recent shifts in migratory movements and the use of seasonal ranges (Kendrick et al., 2005; SARC, 2016; Virgl et al., 2017). Specifically, there is evidence of significant range contractions of southern margins of various caribou herds' ranges (SARC, 2016; Virgl et al., 2017). This northern shift may be to space themselves out from predators (Heard & Williams, 1992) or due to declines in population abundance (Parmesan & Yohe, 2003). Population fluctuations are often associated with shifts in range distribution (Hinkes et al., 2005), where high abundance is related to larger annual range extents and low abundance is related to small annual range extents (Messier et al., 1988). Additionally, a combination of factors including a variety of natural, anthropogenic and ideological causes have been suggested in both science (Festa-Bianchet et al., 2011) and through accounts from Indigenous elders (Kendrick et al., 2005).

2.2.6 Forage Preferences

As grazers and browsers, caribou diets vary seasonally depending on forage availability and nutritional needs (Bergerud, 1972). On an annual basis, caribou are adapted to a wide variety of plant species, including lichens, herbs, fungi, shrubs and graminoids (Bergerud, 1972). This mixed diet allows caribou to thrive in harsh conditions, as feeding on multiple species is more nutritious than relying on one (Bergerud, 1972). In winter, terricolous lichen mats form the majority of a caribou's diet, with graminoids and moss making up the remainder of the diet (Bergerud, 1972; Joly et al., 2010). Lichens are high in digestible energy, but low in nitrogen and water content, meaning that consumption of lichens acts primarily to reduce weight loss until SOS, rather than to gain weight or nutrients (Adamczewski et al., 1988). During winter months, forage availability is limited by snow/ice and fire history. Specifically, the formation of ice and deep snow layers increases the amount of energy required to excavate a feeding crater, decreasing the nutritional gain from foraging lichens (Fancy & White, 1985). Similarly, mature boreal stands and lichens that have not been exposed to fire are favoured by caribou (Thomas, Barry & Alaie, 1996), likely due to more ideal snow conditions (Thomas, Kiliaan & Trottier, 1998) and high percentages of lichen cover and biomass (Barrier & Johnson, 2012).

In early spring, caribou forage on lichen, sedges and over-wintering berries until a new vegetation cycle begins at SOS (Bergerud, 1972; Adamczewski, 1988). However, dietary replacement of lichen and graminoid species with shrubs and forbs generally occurs as the new cycle of vegetation begins, as lichens and graminoids do not have high protein or fat content, needed during this time (Cebrian et al., 2008; Joly et al., 2010). Energy and protein stores must be replenished early in the growing season to provide energy and nutrition for lactating cows after a long migration and for newborn calves (Adams, 2003). In early phenological stages of plant growth, vegetation is high in nitrogen and digestible energy, while low in anti-browsing defense mechanisms, such as resin (McLean et al., 2009). During this time, key forage includes new leaves, catkins and buds of bog bilberry (*Vaccinium uliginosum*), dwarf birch (*Betula pumila* and *B. glandulosa*), crow berry (*Empetrum nigrum*), willow (*Salix* spp.) and deergrass

(*Scirpus cespitosus*) (Bergerud, 1972). Willow species are especially important to caribou during this time and have been seen to comprise nearly 50% of caribou diets during the calving and summer months (Boertje, 1984).

Given the high density of caribou on the calving grounds, the vegetation is soon depleted, leading caribou to move further south to more abundant green vegetation (Nagy, 2011). During this time and continuing into late summer, insect harassment is increasing, leading to a reduction in the ability to feed, nurse and rest due to increased movement rates (Toupin et al., 1996). Plant quality, typically given by the amount of nitrogen content which reflects protein in plant tissue, is extremely important during the early to mid-summer (Van der Wal et al., 2000; Boertje, 1984). Their diet primarily consists of vascular plants, such as willow and dwarf birch, but also feed on fungi, mushrooms and some reindeer lichen (Bergerud, 1972; Boertje, 1984). As summer progresses, there is a switch from prioritizing plant quality to plant quantity to ensure the body fat reserves have accumulated for the fall rut and long winter months (White & Trudell, 1980). Caribou body size accumulation is extremely important for the fall rut and long winter months and is seen to be associated with the length of time that caribou have access to high quality forage in summer (Langvatn & Albon, 1986). In fact, a decrease in quality and quantity of foraging plants can result in decreased growth of calves, and therefore survival (Messier et al., 1988; White, 1983; Chen et al., 2014).

During movement to the rutting grounds, insect harassment has died off and caribou can continue to gain fat reserves for the winter season. The forage during this movement and while on the rutting grounds is highly varied but often consists of forbs, mushrooms, graminoids, mosses, evergreen shrubs and berries (Bergerud, 1972; Boertje, 1984). After rutting, similar foraging patterns occur, but as temperatures get colder and deciduous leaves become less accessible due to leaf fall, there is a switch to reindeer lichens and evergreen shrubs as they are more readily accessible (Bergerud, 1972).

The congregation of caribou on calving and summer ranges can result in caribou at extremely high densities depending on the population status of the herd (Newton et al., 2014). Intense grazing and trampling pressure can result in habitat degradation through the suppression of vegetation growth, biomass production and reproduction, and the complete removal of above ground forage (Zamin & Grogan, 2013). This range degradation is proposed as a possible reason for caribou population declines, such as the George River herd (Campeau et al., 2019; Manseau et al., 1996); however, it is important not to generalize the impact of intense grazing and trampling as the effects will vary depending on herd size, type of vegetation, and the range being studied (Rickbeil et al., 2015).

2.2.7 Mortality

Caribou mortality results from many causes including predation, human development, and harvests. Tundra wolves (*Canis lupus*) primarily prey on caribou, with wolf diets on the Bathurst herd's summer range consisting of up to 71% caribou (Klaczek, 2015). However, calves or weak caribou are also prey on by golden eagles (*Aquila chrysaetos*), grizzly bears (*Ursus arctos*), lynx (*Lynx canadensis*) and wolverines (*Gulo gulo*) (Calef, 1981; Gau et al., 2002). Tundra wolves are assumed to follow the migratory patterns of caribou, but they typically do not follow caribou more north than the calving grounds and have the most prevalence on the summer ranges (Walton et al., 2001; Mattson et al., 2009; Klaczek, 2015). The extent to which predation can affect population dynamics of caribou is relatively unknown, but it is thought that wolves are likely to have a greater long-term influence when caribou densities are low or when wolf densities are high (Bergerud, 1996; Messier et al., 1988).

Both science and Indigenous Knowledge indicate that caribou may be particularly sensitive to human development, especially during calving and summer (Kendrick et al., 2005; Festa-Bianchet et al., 2011; Bergerud et al., 2008). Specifically, caribou have been seen to avoid areas within 1 to 14 km of a human development, including mines, roads, settlements and pipelines (Vistnes et al., 2008; Polfus et al., 2011; Boulanger et al., 2012, 2021). Industrial development can directly influence caribou through increased energy expenditures, reduced forage intake, increased vulnerability to predation and hunters, reducing migration pace and distributional shifts away from preferred seasonal habitat, resulting in reduced body conditions and fecundity (Boulanger et al., 2021; Cameron et al., 2005; Plante et al., 2018). In addition, when caribou populations are in decline, hunting can have exacerbating effects enhancing the decline and prolonging recovery (Beaulieu, 2012). On the ranges of six herd's (Baffin, Cape Bathurst, Bluenose-West, Bluenose East, Bathurst and Southampton), different management plans were implemented between 2007 and 2015 to limit the harvest of caribou by both Indigenous and non-Indigenous hunters in efforts to slow the population decline (COSEWIC, 2016). The management plans are different between the herds, but overall, most include plans to harvest mostly bulls, as cows are more influential to population dynamics (WRRB 2010, 2016). Predation, human development, and hunting can have major effects on population dynamics, but this is likely a result of accelerating decline rather than being the sole cause for the decline of caribou populations (Bergerud et al., 2008; Gunn et al., 2019).

2.2.8 Influence of Habitat Change on Caribou

Climate change poses a continuing threat to barren-ground caribou, exacerbating the complexities of environmental changes (Sharma et al., 2009). Habitat alterations throughout the circumpolar north, including forest fire dynamics, snow and ice depth reductions, vegetation productivity increases and longer growing seasons, have direct and indirect implications on caribou. These changes have already prompted northward range shifts and contractions in caribou (Mennell, 2021), as well as in white-tailed deer (Veitch, 2001).

The increasing severity, frequency, area burned and duration of forest fires in the boreal forest is expected to have profound implications on forage availability for caribou (Lewis et al., 2019). Specifically, areas that have not been exposed to forest fires are shown to have approximately four times the lichen cover than areas that have not been exposed to fires in the last 50 years (Joly et al., 2010). This is of concern for caribou because lichen growth is extremely slow and does not have sufficient biomass for forage until more than 50 years after a fire (Barrier & Johnson, 2012). Furthermore, caribou have been observed to avoid recently burned areas, especially interior regions, by switching foraging sites (Kendrick et al., 2005; Joly et al., 2003). These changing forest fire dynamics have the potential to reduce the quantity of sufficient forage and alter the distribution of both lichen resources and caribou on the winter range, which has the potential to affect population growth (Barrier & Johnson, 2012; Joly et al., 2003).

Earlier snowmelt and altered snowfall patterns in Arctic and sub-Arctic regions can affect spring migration timing, movement ease and energy expenditures (Le Corre et al., 2017; Fancy & White, 1987; Nicholson et al., 2016). Caribou migrations are influenced by local climatic conditions, with mild winters and early snow melt prompting earlier departures from winter ranges (Le Corre et al., 2017; Gurarie et al., 2019). Adverse snow and ice conditions, including thick and soft snow, ice crusts and slush, along migratory routes require more energy and time to travel through (Fancy & White, 1987; Nicholson et al., 2016). Furthermore, ice crusts from rain-on-snow events injure caribou legs and prevent caribou from accessing forage along spring migration routes (Johnson et al., 2001; Aanes et al., 2002). As a result of increased energy expenditures and reduction in forage availability from altered snow and ice dynamics, the health of pregnant cows is significantly reduced, resulting in low birth weight of calves, as well as reduced postnatal growth and development, leading to poor calf survival (Forchhammer et al., 2002; Adams, 2005; Tveraa et al., 2007).

The circumpolar tundra has experienced significant vegetation changes and particularly, the expansion of erect deciduous shrubs is a common explanation for the increases in productivity. This expansion can result in a shift from graminoid and lichen-dominated landscapes to shrub dominated landscapes, serving as a potential threat to forage availability and quality during important spring and summer months (Christie et al., 2015; Fauchald et al., 2017; Legagneux et al., 2014). Specifically, this expansion is associated with increases of dwarf birch and green alder (Fraser et al., 2014) and decreases in lichen and cotton-grass species (Jandt et al., 2008; Fraser et al., 2014). Shrubs species are not the preferred forage species for caribou (Christie et al., 2015) as they have less available protein and contain anti-browsing properties that deter foraging (Zamin et al., 2017). These vegetation changes are critical during spring and summer, to access nutritious forage for gestation, lactation and during winter months (Cebrian et al., 2008; Fraser et al., 2014; Ronnegard et al., 2002). An inability to access adequate forage at this time directly affects fertility and survival of all age and sex classes (Reimers, 1977; Colman et al., 2003).

Warming temperatures have extended growing seasons, resulting from earlier spring growth and later snow onset in fall (Dearborn & Danby, 2021). These phenological changes in vegetation can create mismatches with reproductive strategies for animals that rely on forage resources in late gestation and for lactation (Plard et al., 2014). Migratory species, especially those in harsh environments, are even more susceptible to negative consequences of altered phenology than non-migratory species (Both et al., 2009). Specifically, trophic mismatch is a phenomenon where the time of calving and the time of food availability are no longer coordinated because migratory patterns have not shifted with vegetative changes (Post & Forchhammer, 2008). This was examined for the Bathurst and Kangerlussuaq herds and it was determined that warming, leading to an advancement in SOS resulted in an overall decrease in calf production and increases in calf mortality (Post & Forchhammer, 2008; Chen et al., 2018). This is likely due to inadequate forage resources, because after SOS, vegetation quickly deteriorates into less nutritious food, so it must be consumed before this occurs (Post & Forchhammer, 2008; Tveraa et al., 2013). However, this is not consistent within the literature, where Post et al. (2003) and Mallory et al. (2020) found no evidence of trophic mismatch, but rather, found advancements in migrations and calving due to cues from environmental conditions on the winter range, such as earlier snow melt and warmer winter conditions.

2.3 Theories in Relative Habitat Utilization

2.3.1 Habitat

In the simplest terms, a habitat refers to the place where an organism lives (Morrison et al., 1998). However, habitat also encompasses the resources and environmental conditions necessary for a species' survival and successful reproduction (Block & Brennan, 1993). These conditions can include forage, vegetation cover, water access, and migration corridors (Thomas, 1979). Dispersal and seasonal movement patterns add complexity to the concept of habitat, especially in northern ecosystems where climate change exacerbates this complexity (Caughly, 1977). For example, Davidson et al. (2020) reviewed over 200 animal tracking studies from 1991 to 2020 and found many northern animals, including black bears, moose, gray wolves and barren-ground caribou, had shifted their ranges northward due to habitat changes. Understanding how a species uses its habitat is fundamental to comprehending species distribution, interactions, and population structure (Morris, 1987; Wells & Richmond, 1995).

While linked, habitat utilization and habitat selection are distinct concepts. Habitat utilization refers to how an animal uses its range and the proportion of time spent in different habitats (Hooten et al., 2013; Johnson, 1980). It provides a descriptive measure of where animals are found and how they use different parts of their habitat. Analytical methods, such as generating probability surfaces, are used to visualize habitat use across the landscape, helping to identify core areas, seasonal ranges and habitat characteristics. This would be useful when using collar telemetry data to understand how a subset of the population utilizes its range and to compare habitat characteristics in regions of core use. In contrast, habitat selection involves choosing habitat based on specific resources (Johnson, 1980; Litvaitis et al., 1996). Analytical methods for assessing habitat selection typically compare available habitats that are used to those that are not. This is beneficial for studies assessing individual selection or population-level assessments where locations are determined using aerial surveys. This distinction is crucial because habitat utilization can be incidental or a result of habitat availability, rather than active selection (Beyer et al., 2010). For instance, habitat use is considered selective if a habitat is used disproportionately compared to its availability (Johnson, 1980).

Using resource selection methods to infer habitat selection from a subset of the population, such as animals fitted with GPS collars, can introduce biases and inaccuracies (Aarts et al., 2008). The absence of use by collared individuals does not necessarily mean that uncollared individuals do not use those areas. It is relatively easy to determine locations where the animal is present, but almost impossible to determine absence of a population on a landscape. Studies have shown that assessing resource selection from a sample of the population can lead to inaccurate results because species may be declared absent from the landscape if they are not detected due to sampling methods (Mackenzie, 2005; Aarts et al., 2008). Due to inaccuracies associated with habitat selection methods for telemetry collar data, the remainder of this literature review will focus on habitat utilization.

2.3.2 Modelling Home Range Distribution

Understanding the concept of home range is crucial to assess a species' habitat. Home range is typically defined as “the area traversed by an individual in the course of its normal activities of food gathering, mating and caring for young” (Burt, 1943). High-resolution tracking technologies, such as satellites and Global Positioning System (GPS) collars, have significantly improved the ease of modelling home range distribution (Hebblewhite & Haydon, 2010). Various methods to estimate home range can be divided into two categories: geometric and probabilistic estimators (Fleming et al., 2015). A review of home range estimators is provided in *Table 1*.

Geometric methods, including minimum convex polygons (MCP; Mohr, 1947) and local convex hulls (LoCoH; Getz & Wilmers, 2004), are widely applied due to their simplicity and ease of implementation (Laver & Kelly, 2008). They construct a two-dimensional shape that encloses a specified number of points to delineate an animal’s home range. In contrast, probabilistic methods, including kernel density estimation (KDE; Worton, 1989) and auto-correlated KDE (AKDE; Fleming et al., 2015), use utilization distributions (UD) to provide a probabilistic measure of an animal’s presence to evaluate intensity of use within the home range (Van Winkle, 1975). Commonly, the 95% UD and the 50% UD are used to delineate home range and core areas of activity, respectively (Worton et al., 1987). These techniques have become the preferred and most reliable method for quantifying home ranges due to the incorporation of UDs (Worton et al., 1987).

Table 2.1: Review of common home range estimators of animal distribution. The four methods include minimum convex polygon (MCP), local convex hull, kernel density estimation (KDE), and auto-correlated KDE (AKDE).

Method	Type	How it Works	Application	Example
Minimum Convex Polygon (MCP)	Geometric	The simplest method to calculate the extent of a home range. It uses locational data to draw the smallest polygon around all points, with interior angles less than 180 degrees. All points within the polygon are accessible from any other point without leaving the boundary.	This process is useful for estimating the overall area used by an animal during a specified period.	Simcharoen et al. (2014) used 100% and 95% MCP isopleths to estimate the range size of tigers in the Huai Kha Khaeng Wildlife Sanctuary, Thailand. 95% isopleth was preferred as the 100% isopleth incorporated outliers.
Local Convex Hull (LoCoH)	Geometric	LoCoH works by constructing many MCPs, based on a user defined parameter that allows for the exclusion of areas with hard boundaries or the exclusion of a percentage of location points.	This process can determine the extent of a species home range, while reducing the impact of outlier points and natural occurring barriers, such as lakes or cliffs.	Ryan et al. (2006) used the LoCoH method to calculate home ranges of buffalo of the Limpopo Province, South Africa. This method allowed for the exclusion of unused areas by the buffalo.
Kernel Density Estimation (KDE)	Probabilistic	KDE uses statistical methods to estimate the probability density function based on clustering and frequency of data points. It generates a UD that shows the probability of the animal being in a particular area.	KDE is used to delineate cores areas and intensity of space use. This can be helpful in conservation planning and understanding habitat preferences.	Rosenbaum et al. (2023) used KDE to assess home range sizes and seasonal use of space of collared snow leopards in the Mongolian Altai Mountain range.
Auto-Correlated Kernel Density Estimation (AKDE)	Probabilistic	AKDE produces a density estimate similar to that of traditional KDE. However, it considers movement patterns to account for temporal autocorrelation in tracking data.	Improves home range estimation by incorporating movement paths and temporal data. Useful for data with highly auto-correlated movement patterns, such as long-range migrations.	Moreira-Ramirez et al. (2019) used AKDE to estimate the spatial requirements of white-lipped peccary groups in the Maya Forest of Guatemala and Mexico.

MCPs are the oldest and simplest method of estimating home range boundaries (Mohr, 1947). Despite newly developed methods, they are still widely used in ecological studies as a generic approximation of home range extents (Laver & Kelly, 2008). However, MCPs are often criticized for their simplicity (Borger et al., 2006). They are extremely sensitive to sample size and outliers, often leading to an over-estimation of range size when sample sizes are large, or under-estimation when sample sizes are small (Seaman et al., 1999; Blundell et al., 2001). MCPs can also perform poorly with concave or linear underlying data distribution, incorporating areas that an animal has not used or visits infrequently (Burgman & Fox, 2003).

LoCoH builds on MCPs by creating localized convex polygons around data points to ensure unused areas are not included in the home range estimate (Getz & Wilmers, 2004). This method typically produces smaller estimates than MCPs by allowing for the exclusion of certain areas (Chirima & Owen-Smith, 2017) and performs particularly well in regions that have topographic or geographic barriers to animals. However, differences can arise when specifying how the localized convex hulls are configured (Getz et al., 2007). With small sample sizes, LoCoH tends to underestimate range extent and produce exaggerated gaps in range, excluding areas where animals were not recorded (Chirma and Owen-Smith, 2017).

KDE generates a smooth surface that reflects the probability density of an animal's locations, highlighting areas of concentrated activity. However, KDE is highly sensitive to the smoothing parameter, bandwidth, which determines the search radius of the kernel over local observations (Silverman, 1986). Improper bandwidth selection can lead to either over-smoothing or under-smoothing, distorting the utilization distribution (Silverman, 1986; Gitzen et al., 2006). Additionally, KDE assumes all data points are independent, an assumption often violated in collar telemetry data due to temporal auto-correlation (Hebblewhite & Haydon, 2010). This makes KDE prone to autocorrelation, which can underestimate the density distribution if not properly accounted for, especially with small sample sizes (Fleming et al., 2014; Noonan et al., 2019). Autocorrelation in KDE can be mitigated by increasing the study duration or decreasing the sampling rate (Fieberg, 2007).

AKDE builds on KDE by addressing temporal autocorrelation. It does this by conditioning home range estimates on a movement model to account for uncorrelated and correlated positions and movement velocities (Fleming et al., 2014). While AKDE can improve accuracy, it requires more complex computations and detailed movement data, which may not always be available. A miss-specified movement model can arise when animals portray migratory behaviors due to the stationary movement models being leveraged (Silva et al., 2021).

2.3.3 Modelling Important Habitat Characteristics

Understanding complex interactions between species and various habitat conditions over time and space can be extremely difficult given the complexities of ecological systems (Dungan et al., 2002; Austin, 2007). However, the ability to model relationships is crucial to understand complex relationships that are often unobservable (Dungan et al., 2002). Researchers often use statistical models to relate distribution and movement data to various environmental conditions, including climate (e.g. precipitation; Deguines et al., 2017), resources (e.g., vegetation; Valeix et al., 2011), energy constraints (e.g. Sells et al., 2022), predation risk (e.g., canopy cover; Godvik et al., 2009), and human influence (e.g. mines; Boulanger et al., 2021). These statistical approaches can provide insight into ecological factors, such as home range distribution, habitat preference, behaviors, and critical habitats (Morales et al., 2010; Nathan et al., 2022).

Historically, linear modeling methods, such as multiple linear regression (MLR), have been employed to understand the influence of environmental characteristics on species populations (Maes et al., 2005; Knudby et al., 2010). MLR models the relationship between a dependent variable and multiple independent variables using a simple linear equation. However, advancements in statistical techniques have introduced models capable of addressing non-linear relationships, such as generalized linear models, generalized additive models, and multivariate adaptive regression splines (Guisan et al., 2002). These traditional modeling approaches are simple, easily interpreted, and effective on small datasets, but their flexibility is often limited and work best when relationships between variables are already understood (Austin, 2007). They can have strict assumptions about the data, including normality and data independence, which are not always true in ecological contexts (Guisan et al., 2002; Elith & Graham, 2009). The computational ability to handle high-dimensional data and data with noise or gaps is another drawback of these approaches (Austin, 2007).

In recent decades, technological advancements have propelled machine learning into a prominent role within ecological modeling. Machine learning techniques learn from the data provided, with no prior assumptions, allowing for useful application in complex and non-linear systems. By learning patterns from large datasets and generating predictive mathematical functions (Hastie et al., 2009), these methods offer enhanced predictive power and flexibility compared to traditional linear models (Christin et al., 2019). In addition, these approaches are capable of dealing with predictor variables that are correlated in nature (Hochacka et al., 2007). There are three common approaches to machine learning in ecology that use various methods to assess relationships (Scowen et al., 2021), including classification and regression trees (CARTs; Breiman et al., 1984), artificial neural networks (ANNs; McCulloch & Pitts, 1943), and clustering algorithms (Jain, 2010).

CARTs are the most widely used approach for predictive modeling in ecology (Scowen et al., 2021). Two common types in the literature include boosted regression trees (BRT) and random forest modeling. BRTs combine multiple single regression trees built sequentially to allow each tree to correct for residuals (errors) prevalent in the previous tree. This results in a strong model that reduces bias and variance (Friedman, 2001). In contrast, random forest models create multiple bootstrapped regression trees that are averaged to gain the most accurate result and reduce over-fitting (Breiman et al., 1984). Both models have their strengths and provide reasonable results; however, random forest models are often preferred because they are easier to train and robust to over-fitting (Alnahit et al., 2022; Park & Kim, 2019). Specifically, random forest models are less prone to over-fitting versus BRTs because they use bagging and random variable selection, which adds diversity to the trees and reduces the overfitting of the training data (Breiman et al., 1984; Hastie et al., 2009). Furthermore, random forest models are robust to noise and outliers in the data due to the ensemble nature and randomization process during model construction (Breiman et al., 1984).

Table 2.2: Review of machine learning techniques, including classification and regression trees (CARTs), artificial neutral networks (ANNs) and clustering algorithms.

Machine Learning Technique	How it Works	Applications	Methods	Examples
Classification and Regression Trees (CARTS)	Supervised decision tree-based models that repeatedly split data into subsets based on specific rules until a certain strength is achieved. Each split is based on the feature that provides the most significant information gains.	Widely used in ecological predictive modeling to assess relationships from remote sensing, terrain and climatic data. Used to assess variable importance of predictor variables.	- Boosted regression trees - Random forest	Adhikari et al. (2019) used regression trees and soil survey data to predict soil carbon stocks under climate change and land use change scenarios.
Artificial Neural Networks (ANNS)	Mimics the human brain with interconnected nodes (neurons) in multiple layers. Each neuron applies a function to input data and passes the result to the next layer. Trained using backpropagation to minimize the error between predicted and actual outputs.	Used in descriptive image analysis models and predictive regression modeling.	- Multilayer perception - Convolutional neural networks - MaxEnt	Khosla et al. (2020) used ANN to predict crop yields from monsoon rainfall data.
Clustering Algorithms	Clustering algorithms are typically unsupervised and function by grouping data into many collections based on similarities of specific characteristics (Jain, 2010).	Clustering is often used as a methodological step in predictive modelling to identify the structure of data without theoretical assumptions, in data sorting and hypothesis generation.	- K-means - Hierarchical cluster analysis - ANN clustering	Majumdar et al. (2017) used clustering prior to predicting agricultural crop yield using regression. This was done to group and assess data based on districts that have similar temperature, rain fall and soil type.

ANNs are commonly referred to as a "black-box," where the internal structure, design, and implementation are not known by the researcher, and there is no theoretical basis that describes how to build these networks (Fielding, 1999). The number of nodes, which are hidden from the researcher, can highly alter the prediction results; if the number of nodes is too small, a minimum training duration will not be completed, and if there are too many nodes, it can lead to over-fitting of data (Srivastava et al., 2014). Furthermore, ANNs have significant dependence on sample quality and quantity, requiring substantial time to learn and capture patterns.

Clustering algorithms often have many drawbacks in ecology. Non-hierarchical clustering methods require the number of clusters to be defined beforehand, which can lead to incorrect parameter choices and misinterpret ecological data (Legendre & Legendre, 2012). Although hierarchical methods do not require a pre-determined number of clusters, they are often highly sensitive to computational complexity, limiting applicability to small datasets (Wu et al., 2022). Additionally, understanding the ecological significance of each cluster and how different variables contribute to cluster formations can be challenging, especially with more complex algorithms such as density or graph theory (Xu & Tian, 2015). Outliers or noisy data can significantly impact clustering methods, resulting in fragmented clusters or merged clusters that do not accurately reflect the underlying ecological conditions (Milligan, 1980).

Chapter 3: Bathurst Caribou Range Use Patterns Through Time

3.1 Introduction

Barren-ground caribou (*Rangifer tarandus groenlandicus*) are one of the most iconic and ecologically significant species in the Arctic. However, they are facing severe population declines across their vast circumpolar range, raising concerns about the health of northern ecosystems (Vors & Boyce, 2009). These declines serve as an indicator of the broader impacts of rapid environmental changes in the Arctic, where caribou play a crucial role in regulating ecosystems by influencing plant and predator communities through their activities, such as trampling, defecation and grazing (Musiani et al., 2007; Sharma et al., 2009; Post & Klein, 1996). Moreover, for thousands of years, northern Indigenous communities have relied on these animals for sustenance, cultural practices and economic value (Kendrick et al., 2005; Gordon, 2005). The widespread distribution and historical abundance of barren-ground caribou make their decline particularly alarming.

Climate change poses a significant threat to barren-ground caribou, especially as Arctic and sub-Arctic regions experience unprecedented warming. The effects of climate change are evident in a range of environmental changes, including more frequent and intense wildfires, altered precipitation patterns, and shifting snow cover dynamics (IPCC, 2021; Zhang et al., 2019; Flannigan et al., 2009). These changes are altering the ecosystems that caribou depend on, leading to shifts in vegetation productivity, phenology and composition (Dearborn & Danby, 2021; Post et al., 2009). For example, the increase in forest fires, due to longer growing seasons and higher temperatures, is expected to transform ecosystems from spruce-dominated to deciduous-dominated landscapes (Chapin et al., 2010). Additionally, circumpolar precipitation patterns are shifting, with more rain and less snow, although local variations due to vegetation and topography will influence snow accumulation and melt characteristics (Zhang et al., 2019). Overall, tundra vegetation productivity is increasing, while boreal vegetation productivity may be decreasing due to drought stress and fires (Bonney, Danby & Treitz, 2018; Dearborn & Danby, 2021; Lotsch et al., 2005). Additionally, altered vegetation phenology, specifically an increase in the length of the growing season due to an earlier start of season (SOS) and a shorter winter, is a common explanation for observed increases in plant productivity (Goetz et al., 2005; Post et al., 2009; Dearborn & Danby, 2021).

The direct and indirect impacts of climate change have important implications for barren-ground caribou, particularly through altered habitat conditions. Caribou rely on seasonal migrations to access forage, avoid predators, and minimize insect harassment, all of which are influenced by environmental cues (Hebblewhite & Merrill, 2009). As temperatures rise, the timing and availability of critical resources, such as forage during the growing season, are shifting (Linderholm, 2006). A disruption in the synchrony between caribou and their food sources, especially during calving and summer, when females and calves have the highest nutritional needs, poses a significant challenge. For instance, an earlier onset of vegetation growth may create a mismatch between caribou reproductive strategies and vegetation phenology, as they rely on specific forage during late gestation and lactation (Plard et al., 2014). Additionally, the expansion of shrubs in tundra regions can reduce the availability of nutritious forage during summer, as shrubs are less palatable and have anti-browsing properties (Zamin et al., 2017; Cebrian et al., 2008; Fraser et al., 2014). Furthermore, warmer summers lead to increased insect harassment, exacerbating stress on caribou and altering their movement patterns as they seek to avoid these pests (Weladji et al., 2003). Ice and snow dynamics, such as the timing of ice break-up, snowmelt, and snow depth, also influence the ease of movement and energy expenditures during spring migration, affecting the health and survival of caribou (Reimers, 1977; Colman et al., 2003).

The focus of this research is the Bathurst caribou herd, located in the Northwest Territories and Nunavut, Canada, which has experienced a drastic population decline over the past four decades. The herd has decreased by 98%, from approximately 470,000 in the mid-1980s to just 6,240 individuals in 2021 (GNWT, 2019). Although caribou populations are known to naturally fluctuate, this decline is considered unprecedented (Kendrick et al., 2005). The low herd numbers increase the vulnerability to climatic and anthropogenic changes, potentially leading to local extinctions (Mallory & Boyce, 2018). Observations from both Indigenous elders and scientific research indicate changes in the herd’s range use patterns, underscoring the need for understanding these shifts to ensure effective conservation (Virgl et al., 2017; Kendrick et al., 2005).

Environmental changes in the circumpolar north are likely having cumulative effects on the Bathurst herd. The stark population decline has prompted research investigating the potential causes, such as changes in vegetation productivity and phenology. Although not ubiquitous, the Bathurst herd's tundra range has seen significant increases in plant productivity, which may be linked to an earlier onset of vegetation growth in the spring (Dearborn & Danby, 2021) and the expansion of deciduous shrubs (Post & Forchhammer, 2008), a process known as 'shrubification'. However, the underlying mechanisms driving shrubification remain unclear. For example, Bonta et al. (2023) suggests that increased foliage production or survival might explain trends in plant productivity, while Andruko et al. (2020) proposed that reduced caribou impacts, such as trampling and browsing, may be a significant driver of shrub expansion. Additionally, a trophic mis-match resulting from climate warming has also been hypothesized (Post & Forchhammer, 2008; Chen et al., 2018), and could lead to decreased calf production and increased calf mortality if forage availability does not align with critical periods of nutritional demand (Cebrian et al., 2008; Fraser et al., 2014). For instance, an inability to access nutritious forage during key periods, such as spring and summer, can directly impact the fertility and survival across all age classes within the herd (Reimers, 1977; Colman et al., 2003; White, 1983). While these mechanisms provide plausible explanations, the evidence remains inconclusive, and further research is required to clarify the complex interactions between climate change, vegetation dynamics, and caribou population declines.

The dramatic population decline of the Bathurst herd led to the use of GPS telemetry collars to track the movement patterns of the population. Analysis of the data has documented decreases in the annual range extents of the herd, as well as reductions in seasonal ranges, particularly from calving to late fall (Virgl et al., 2017). Additionally, a general northward shift in range use has been observed, especially during the winter and autumn months (Mennell, 2021). Individual movement studies have also revealed decreases in the duration of spring migration and increases in post-calving movement to the summer range, potentially due to earlier vegetation onset and increased insect harassment (Mennell, 2021).

However, there is limited research on how the herd utilizes its summer range during this period of population decline, despite this being a crucial time for lactating cows and newborn calves to build fat reserves for the fall breeding and winter periods (White & Trudell, 1980).

This research aims to understand how the distribution and extent of the Bathurst caribou herd's summer range have changed during a period of rapid population decline (1997 to 2017), and to relate these changes to trends in environmental and habitat characteristics during the same period. This is completed using telemetry collar data and environmental data from satellite imagery and by developing various modelling analyses. Ground-truthing was also conducted to assess relationships with satellite imagery. The specific objectives are:

1) Analyze collar telemetry data from 1997 to 2017 to determine if and how the relative summer range distribution and extent of the Bathurst herd have changed during its recent population decline.

2) Use results of a remote sensing analysis and synoptic climatic data to ascertain if there are relationships between environmental changes and regions of increasing or decreasing caribou utilization.

Identifying relationships between relative summer range distribution and habitat changes would support the hypothesis that climate and/or environmental changes have contributed to the decline of the Bathurst caribou herd.

3.2 Methods

3.2.1 Study Area

The annual range of migratory Bathurst caribou herd historically spanned ~390,000 km^2 across tundra and taiga biomes of Nunavut and the Northwest Territories, with the herd calving near the Bathurst Inlet in the Kitikmeot Region of Nunavut (*Figure 2*). The herd's range intersects with many Indigenous groups, including Tłı̨chǫ Lutsel K'e, Dene First Nation, Yellowknives Dene First Nation, Metis

Athabasca Denesuline, and Inuit, for which they rely on and understand mutually beneficial relationships (GNWT, 2019). The focus of this research is the summer range of the herd, a time when the herd moves southwest from the calving grounds to the summer grounds. The dates used to identify the summer range were originally defined by Nagy (2011) for the Bathurst herd based on an analysis of movement rates of 52 collared cows from 1996 to 2008. The summer range is the second most sensitive part of the Bathurst herd's range to disturbances, after the calving area (GNWT, 2019). This range is particularly important to the animals that occupy it after calving and before fall breeding and the winter months. At the time of fall breeding, pregnancy success and calf survival are closely tied to the body condition of breeding females (Cameron & Van Hoef, 1994). Disturbances that reduce access to forage during the summer can negatively impact body condition, which may in turn affect the growth of the population (Rettie & Messier, 1998; Griffith et al., 2002). Therefore, it is critical that breeding females and newborn calves can maximize the intake of nutritious forage on the summer grounds to have adequate body condition prior to the fall breeding season (Cameron & Van Hoef, 1994).

The topography of this region is generally characterized as low elevation terrain, with the greatest elevation located in the vicinity of Contwoyto Lake, greater than 500m above sea level (M.A.S.L), in comparison to the rest of the study region, which ranges from approximately 0 to 450 M.A.S.L. The region contains numerous lakes, and a combination of outwash plains and esker complexes from past glacial sediment deposits (Ecosystem Classification Group, 2012). This region's vegetation and permafrost features reflect low Arctic climatic regimes, characterized by short, cool meteorological summers (July and August), and dry long, and very cold winters. For example, at Lupin Gold Mine, located within the herd's summer range in Nunavut, the July mean temperature is 11.5° C, and the January mean temperature is -29.9° C. The average annual total precipitation is 298.5 mm, with the highest month being August (average of 62.5 mm) (Environment Canada 1981-2010 climate normals from Lupin A Weather Station, 65°45' N 111°15'W, 490.10m above sea level). In the coming decades, average air temperature and precipitation are both projected to increase, which may result in shorter periods of snow coverage, longer growing seasons, changing ecotypes, permafrost degradation and an increase in extreme weather (IPCC, 2021).

The annual range of the Bathurst herd has two dominant vegetation types, with treeline marking the boundary between the two. At the southern extent of the herd's range, below the treeline, exists the Boreal Forest ecozone, often characterized by open canopy spruce with lichen groundcover in valleys and sparsely vegetated outcrops on ridges (GNWT, 2019). Beyond treeline, the landscape is void of trees and upland vegetation is dominated by erect, deciduous shrubs; lowlands are comprised of sedge, moss, dry lichen, and dwarf shrubs. Sparsely vegetated outcrops are common throughout the region (Ecosystem Classification Group, 2012).

Most of the Bathurst range is located on the Slave Geologic Province, which has a long history of resource extraction and associated infrastructure (GNWT, 2019). Currently, diamonds are the primary minerals being extracted and five large diamond mines were at various stages of operation as of 2022, including Jericho, Snap Lake, Diavik, Ekati, and Gahcho Kue (*Figure 2*). Many other areas are claimed with active exploration and remediation occurring (GNWT, 2019). A winter ice road runs from Tibbit Lake to Contwoyto Lake in addition to other smaller winter roads that are open to access by the public. Several mineral developments and associated infrastructure have also been proposed, including the Grays Bay Road and Port, the Bathurst Inlet Port and Road, and the Slave Geological Province corridor (GNWT, 2019).

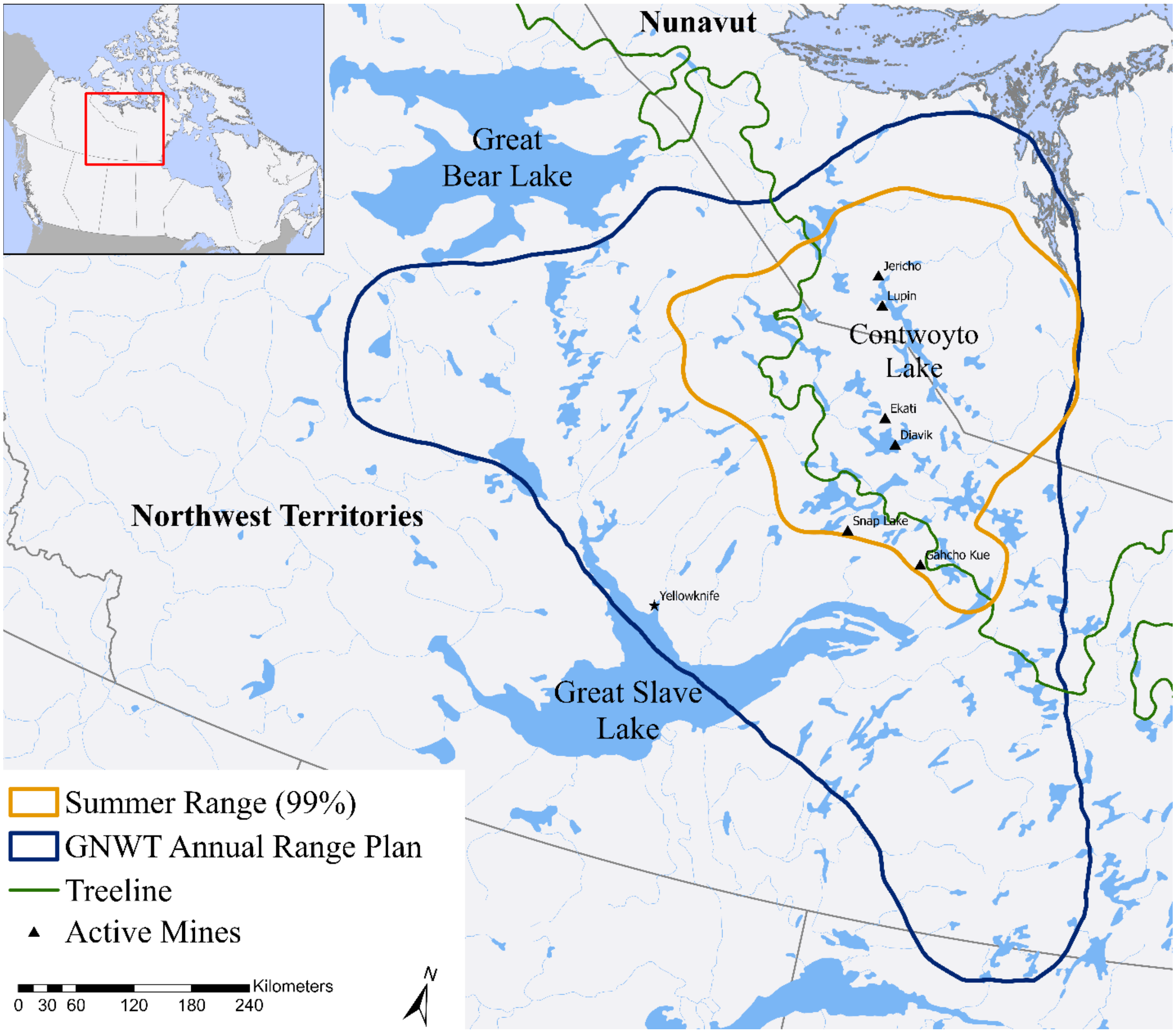

Figure 3.1: The range planning area (blue) for the Bathurst caribou herd in relation to the historical annual summer range (orange) defined. Areas below treeline (green) generally fall within the taiga biome and areas north of treeline fall within the tundra biome. Active mines within the summer range are indicated with a triangle, including Snap Lake, Gahcho Kue, Ekati, Diavik, and Lupin. The annual summer range boundary was calculated using a minimum convex polygon from telemetry data collected between June 29 and September 6, 1997 to 2017 (see Section 3.3.4 for details).

3.2.2 Collar Data Acquisition and Processing

In March of 1996, after consultation and approval by the Tłı̨chǫ Government, the Government of the Northwest Territories (GNWT) deployed a telemetry monitoring program for the Bathurst herd to monitor seasonal distribution and migratory movements. The capture of caribou follows methods outlined in Gunn et al. (2013) and are in compliance with the Canadian Council on Animal Care.

The locational accuracy of global positioning system (GPS) coordinates, the frequency of location coordinates and the number of collars deployed have all increased over time in association with changes in technology and a growing desire to learn about the herd as it continues to decline. Initially, in 1996, 10 collars were deployed on cows, and this grew to 20 collars in 1998 to monitor the influence of the diamond mines on the herd's distribution (Adamczewski & Boulanger, 2016). Until 2015, the number of active collars did not exceed 20, but in 2015 the Tłı̨chǫ Government approved an increase that permitted 50 collars. This was also the first time that bulls were collared, with 30 collared cows and 20 collared bulls that year (Adamczewski & Boulanger, 2016; *Figure 3B*). Overall, the total number of caribou collared between 1996 and 2017 was 221 individuals. Similarly, the number of GPS locations that were collected and stored also increased as technology and data storage improved. Initially, from 1996 to the end of 2004, collars collected and stored a single location coordinate for each animal once every three to five days. However, after 2005, the collection and transfer of coordinates back to the GNWT occurred more than three times daily for each collar (*Figure 3A*). Furthermore, the location accuracy of each observation also increased throughout the study period, as described in Mennell (2021), from approximately +/- 500m early in the monitoring process to +/- 100m later on (see *Appendix A* for more details). The average locational accuracy, calculated by Mennell (2021) from 1996 to 2019, was +/- 275m.

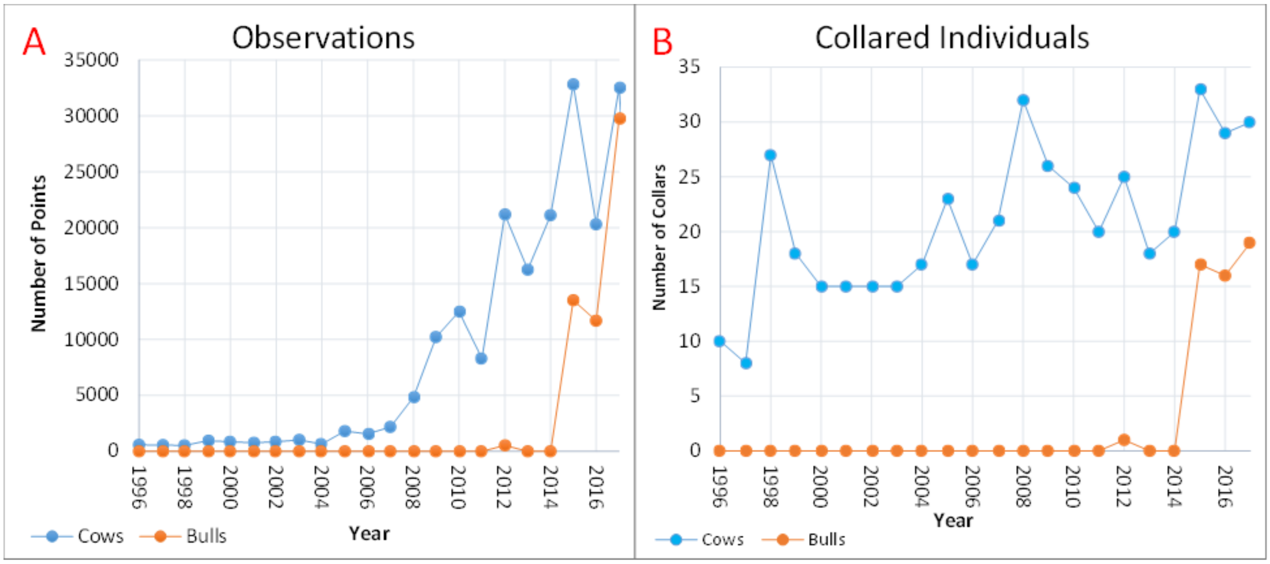

Figure 3.2: The number of observations collected each year (A) and the number of collared individuals each year (B). Bulls were not collared until 2015, and therefore are not included in my analyses.

I obtained the telemetry data under the terms of a data release agreement with the GNWT Environment and Natural Resources. The locations were provided as Longitude and Latitude coordinates and using ArcGIS Pro v3.1.2 (ESRI, Redlands, California), I projected this into the NAD83 datum of the North American Lambert Conformal Conic coordinate system. The raw dataset included multiple duplicate observations and therefore, cleaning of the dataset was completed using the methods outlined by Mennell (2021). This removed all duplicated observations as well data for animals with less than ten observations.

In addition, all observations for bulls were removed, leaving only data pertaining to cows. This was done for three reasons. First, there are significant data limitations when incorporating data from bulls, as they were not collared until 2015 and when they were collared, there were significantly fewer collared bulls than cows (*Figure 3*). Second, from the spring until the rutting period in the fall, bulls do not follow the same movement patterns as pregnant cows; bulls typically begin their spring migration a few weeks after cows (Jakimchuk et al., 1987) and aggregate together on the late summer range only after the cows arrive from the calving grounds (GNWT, 2019). Lastly, the herds' productivity and calf survival have been linked to cow body condition at the time of breeding and it is arguably more crucial to identify regions that are important to cows (Rettie & Messier, 1998; Griffith et al., 2002). After eliminating all bull observations from the dataset, 192,215 total observations from 1997 to 2017 remained.

As previously mentioned, this research focuses on the summer range utilization distribution based on Nagy (2011) movement analyses, who defined the summer range as being from June 29th to September 6th. The last phase of processing the collar data was done to account for variation in the frequency of observations and number of collared individuals, which increased over the study period. Specifically, a mean location was calculated for each collared individual daily. This was done using the function *getDailyMean* from the R package *TuktuTools* v.0.0.0.1 (Gurarie et al., 2019), which calculates an average of location coordinates for each individual on every day of summer. Once completed, the final dataset for the summer range from 1997 to 2017 consisted of 79,819 locations.

3.2.3 Relative Summer Range Distribution

Home range is commonly used to refer to an area used by an animal to fulfill basic needs such as forage gathering, mating, and caring for young (Burt, 1943). Home range in this study is based on collar telemetry data to determine the relative distribution of the Bathurst herd within the summer range. I emphasize the term 'relative' as only a small sample of the total population is collared and therefore, I cannot assume that this distribution aligns with the locations of every individual in the population. Moreover, while individuals may exhibit variation in their specific movements and behaviours, they tend to generally follow similar patterns collectively, reflecting the overall trends of the herd. Therefore, due to the aggregation of females and calves during this time combined with the highly synchronized movement of animals between seasonal ranges, a relatively small sample size can represent the behavior of an entire caribou herd remarkably well (Maier & White 1998; Campbell et al., 2010).

The kernel density estimate (KDE) method was used to estimate the cumulative summer range boundary for the Bathurst herd from 1997 to 2017. Home range, in this sense, is defined as the smallest polygon that contains anywhere from 90% to 99% of datapoints from the original dataset. For the

Bathurst caribou herd's summer range, KDE was used to estimate the extent with 99% of data points to exclude outliers and remove individuals with eccentric movement patterns. Unlike the minimum convex polygon (MCP), which connects the outermost points with straight lines that may overestimate or underestimate range extents, KDE generates a smooth utilization distribution, providing a more nuanced representation of the herd's range use. This approach allows for a more accurate representation of the summer range that is consistently used by the collared individuals, ensuring that the analyses focused on typical range use patterns rather than anomalous behaviors. The resulting 99% cumulative summer range contour line was used to delineate the spatial boundaries within which annual KDEs were calculated and compared, ensuring consistency across all years. This analysis was completed using the R package *adehabitatHR* (Calenge, 2006).

To assess how the herd utilized its summer range each year, telemetry locations were used to generate a continuous surface of probability, called a utilization distribution (UD). This shows the likelihood of where the herd is likely to be found within the summer range. I employed KDE to create this UD. KDE generates a continuous surface that estimates density distribution by calculating density at each intersection of a user defined grid. The kernel function, an algorithm within KDE, determines which locations are included and how they are weighted. This method highlights core-use areas, where the herd's activity is most concentrated, as well as broader home range areas through the use of density isopleths. This approach allows for the exclusion of less densely populated areas, providing a clearer focus on primary regions of herd activity (Worton et al., 1987; Kernohan et al., 2001). For this study, 50% and 95% density isopleths were selected to represent areas of concentrated activity and overall range use, respectively.

Two important variables must be considered in the generation of KDE: bandwidth, or search radius, and cell size. Bandwidth is the smoothing parameter, which controls how much influence each data point has on the estimated density surface. A smaller bandwidth means that each point only affects a small area around it, resulting in a detailed and fine-grained density surface that can highlight small-scale features. In contrast, a larger bandwidth allows each point to influence a wider area, creating a smoother and more generalized density surface that can capture broader trends but may obscure finer details (Silverman, 1986). This parameter is crucial because it balances the trade-off between capturing detailed variations and producing a smooth, interpretable density surface. There is no general consensus on determining what the best bandwidth is for telemetry collar data; however, Silverman's Rule of Thumb and the Plug-in selector are two common methods (Silverman, 1986; Wand & Jones, 1994).

Silverman's Bandwidth calculates how spread out the data points are and aims to minimize the mean square error in estimating the optimal bandwidth. This method is common, but there is evidence that it tends to overestimate the range extent due to its' assumption of normal distribution (Chirima & Owen-Smith, 2017). In comparison, the multivariate Wand & Jones (1994) plug-in method does not assume a Gaussian normal distribution and provides more sophisticated techniques that aim to minimize the asymptotic mean integrated square error of the distribution. Based on the limitations when using Silverman's Rule of Thumb for ecological analyses and the typical movement rates of caribou during this time, I opted to use the Wand & Jones (1994) plug-in method to calculate bandwidth. This was implemented using the function *dpik* from the R package *KernSmooth* v.2.23-22. Bandwidths were calculated separately for each year, and this resulted in an average bandwidth of ~16,351m across all years of analysis.

The second variable to consider when calculating kernel density is cell size and although cell size has less influence on the ability to accurately predict spatial patterns for kernel density analysis, it still influences how the density output appears visually (Chainey, 2013). In simple terms, the smaller the cell size, the more detailed the kernel density output will be and the larger the cell size, the more smoothed the data will appear. To assess the influence that cell size has on the KDE output, I tested cell sizes ranging from 100m to 1000m. Comparison of the results showed only very minor differences in the general, underlying patterns. With this, I opted to use a cell size of 500m to coincide with the lowest locational accuracy of the telemetry data early in the study period. From the daily mean resampled locations, an estimation for a fixed, nonparametric, bivariate kernel was utilized to create 21 KDE maps for the Bathurst herd's summer range, one for each year (1997-2017). This was done using the *getkernelUD* from the R package *adehabitatHR* (Calenge, 2006). The maps were visually inspected and compared to assess how the herd's relative use of space varied from year to year during rapid population decline.

A pixel-wise Theil-Sens regression was performed on the annual KDEs to identify trends in relative range utilization over the duration of the study period. This is a non-parametric rank-based regression technique that uses the median of all regression slopes between unique pairs of observations (Kendall & Stuart, 1967). This illustrates the extent to which pixels have experienced increases or decreases (or no change) of relative caribou utilization over the length of the study period. Theil-Sens regression was performed in R using the *raster.Kendall* package of the *spatialEco* package (Evans & Murphy, 2021). The output of this technique was a slope, p-value, and y-intercept for each 500 m pixel of the study area.

3.2.4 Relative Habitat Utilization Modelling

To assess the influence of changing environmental conditions on the Bathurst herd's summer range, I developed two random forest models. Random forest is a non-parametric ensemble learning method that combines the predictive power of multiple regression trees to assess the impact of multiple predictor variables (e.g., climate data) on a response variable (Breiman et al., 1984). Each regression tree in the forest is trained on a random subset of the data using a technique called bootstrapping, where two-thirds of the data is used to build each tree, and the remaining one-third, known as the out-of-bag (OOB) data, is set aside for validation (Breiman et al., 1984). The OOB data is not used to train the tree, allowing for an unbiased estimation of model performance when the predictions are tested against it. By averaging the errors from these OOB predictions across all trees, the OOB error rate provides a reliable measure of the model's overall accuracy and is commonly used to tune parameters and evaluate model performance.

The core idea behind random forest is to reduce the variance of individual regression trees by averaging their predictions, resulting in a more robust and generalizable model (Hastie et al., 2009). Each regression tree is constructed by splitting the data at various nodes based on criteria that maximizes the separation of the response variable. At each node, the best split is chosen from a random subset of the predictor variables which helps reduce correlation between the trees and prevents over-fitting (Cutler et al., 2007). By aggregating the predictions of all trees, this model produces a final prediction that reflects the average outcome of the individual trees, resulting in an estimate that improves accuracy, avoids overfitting and generalizes well to new data (Breiman et al., 1984). Moreover, random forest models provide insights into the relative importance of each predictor variable through measures such as Gini importance or mean decrease in accuracy, offering an intuitive way to assess which environmental factors most influence the response variable. The modeling process is as follows (Breiman et al., 1984):

1. A specified number of bootstrapped samples (ntrees) are drawn from the dataset. Each sample contains approximately two-thirds of the dataset's observations, with the remaining one-third set aside as OOB data for that sample.
2. For each sample, an unpruned regression tree is grown, with a defined number of variables (mtry) randomly selected at each node.
3. The predictions from the OOB data are averaged across all trees to obtain the OOB error rate, which provides an unbiased assessment of model performance and is used to optimize model parameters such as mtry and ntrees.

Two different categories of variables were used in both models. Static variables refer to environmental variables that have a fixed value through time. This includes variables such as elevation, mean windspeed, and time since the last fire, which do not change or change only marginally overtime. In contrast, dynamic variables refer to environmental variables that represent a trend through time. For example, the maximum summer temperature variable used in our modeling represents the mean annual change in temperature experienced over the entire study period. The first of two random forest models included both static (fixed value) and dynamic (temporal trend) variables, while the second model included only dynamic variables. In real-world scenarios, both dynamic and static variables influence species behavior; hence, the first model integrates both types.

Although incorporating both variable types in the same model can be contentious (Brook et al., 2009), it has been done in various studies, such as vegetation dynamics simulations and medical analysis (e.g., Hole et al., 2009; Choi et al., 2016). Some studies, however, prefer analyzing only one data type to avoid issues related to collinearity and complexity (e.g., Wang et al., 2018). Collinearity arises when predictor variables are highly interrelated, potentially causing model inaccuracies and reduced statistical power (Graham, 2003). This is often mitigated by assessing pairwise correlation coefficients and removing highly correlated variables (Buse & Griebeler, 2011). Consequently, I developed two different models to ensure that issues from combining static and dynamic variables were avoided and to assess that static variables' influence in comparison to the model with only dynamic variables.

Data Acquisition

The response variable in both random forest models was the slope yielded from the Theil-Sen analysis of annual summer range use, while the potential predictor variables included mapped environmental, climatic, and topographic variables. Static variables included elevation (M.A.S.L), time since last fire (years), and the average summer (June, July, August) windspeed (kph). The dynamic variables representing trends through time included change in snow melt timing (days), trend in maximum summer temperature (°C), trend in minimum summer temperature (°C), trend in length of season (LOS; days), trend in SOS (days), trend in end of season (EOS; days), and trends in annual maximum enhanced vegetation index (EVI) and time-integrated enhanced vegetation index (TIE). Dynamic variables with positive values represent an increase overtime, while all negative values represent a decrease overtime. Refer to *Table 3* for further details and *Appendix A* for visual representations of these variables.

Dataset Validation

Vegetation surveys were conducted in July and August 2023 on the Bathurst herd's summer range to validate the accuracy of the Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery used to derive independent predictor variables (max EVI, LOS, SOS, EOS, TIE). This ground-truthing analysis aimed to assess the accuracy of the satellite data in reflecting the actual ground conditions within the herd's summer range.

A total of 31 field sites were selected (*Figure 4*) and surveyed based on dynamic maximum EVI metrics (Dearborn & Danby, 2021) and the relative changes in caribou use (as described in Section 3.3.4). Surveys were conducted during the time spent at the Tundra Ecosystem Research Station on Daring Lake, while accompanying the Tłı̨chǫ-led Boots on the Ground monitoring program, and in various remote locations across the study area. Sites were chosen for their accessibility by foot or boat from campsites, while ensuring a minimum separation of 250m to avoid multiple samplings within one satellite pixel. Each field site measured 50m by 50m with nine 50cm by 50cm quadrats evenly distributed within each site (*Figure 5*). GPS coordinates were recorded using a handheld GPS receiver (Garmin eTrex model) at the corners and center of each site. A modified SLR camera (Canon EOS Rebel T4i) capturing green, blue, and near-infrared (NIR) wavelengths was used to photograph each quadrat twice: one photo included a white balance for ambient light condition standardization, and the second contained only the quadrat. These photos were subsequently analyzed using ENVI and R software.

ENVI (NV5 Geospatial, 2022) was used to identify regions of interest in the photos for analysis. The white balance photo was masked to include only pixels within the white balance, and the quadrat photo was masked to include only pixels located inside the quadrat. These masked photos were exported as TIF files and imported into RStudio (R Core Team, 2023) to calculate a modified NDVI for each quadrat, following methods similar to Freemantle (2020) and Schoenhardt (2023).

NDVI uses NIR and red radiation bands of spectral reflectance to indicate "greenness" or photosynthetic activity. NDVI values range from -1 to +1, with higher scores indicating healthier vegetation biomass (Huete et al., 2002; Goetz et al., 2005). The modified NDVI used in this ground-truthing analysis, known as "green NDVI" (GNDVI), utilizes the green spectral band rather than the red band. GNDVI is more sensitive to chlorophyll concentrations, a key pigment in photosynthesis, than NDVI, and therefore, can provide a more accurate reflection of chlorophyll content, and consequently, photosynthetic activity (Gitelson et al., 1996).

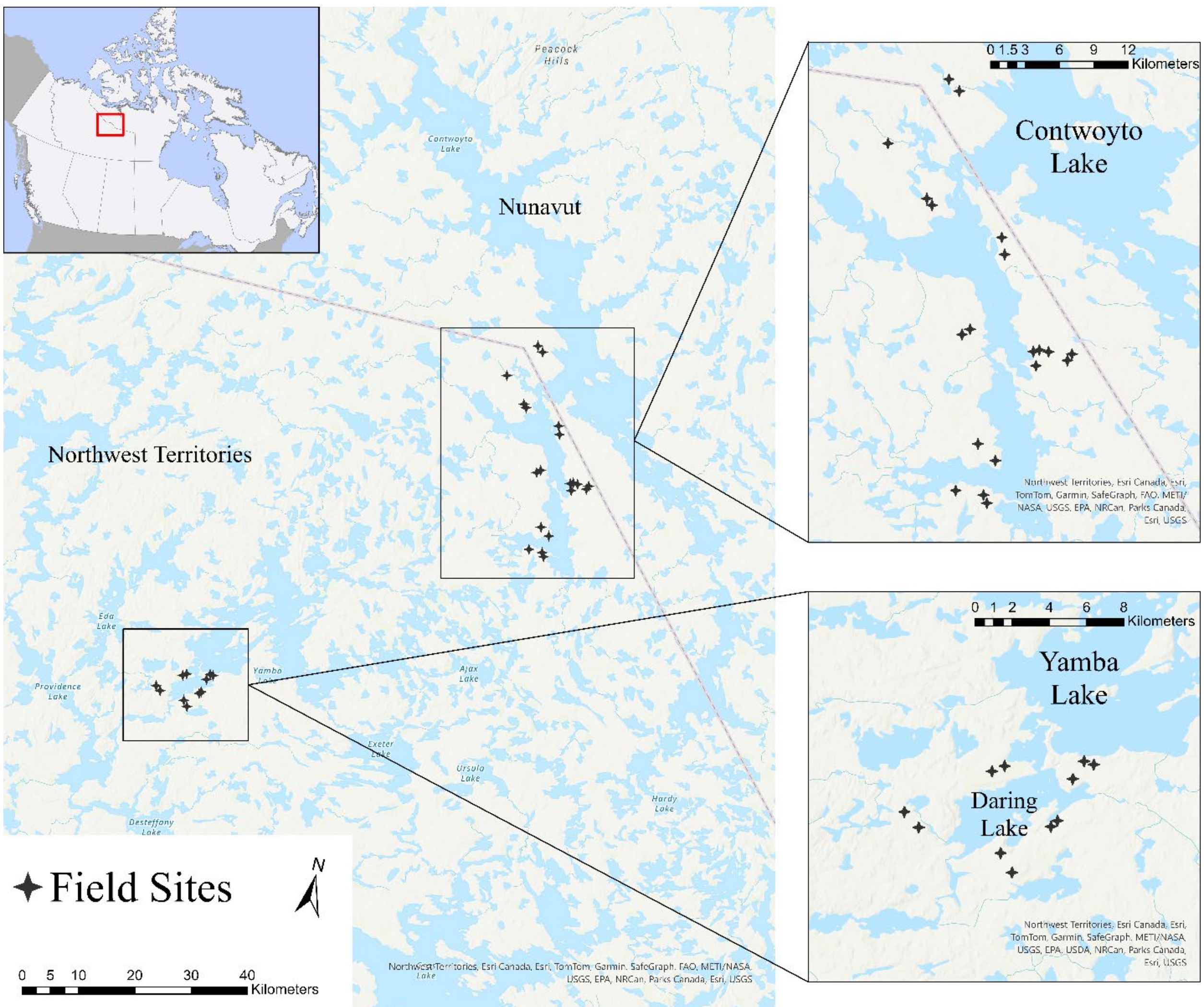

Figure 3.3: The location of all field sites sampled in July and August, 2023. The two regions that were sampled were located around Fry Inlet, the southwestern arm of Contwoyto Lake in NWT and Nunavut, and an area surrounding Daring Lake, NWT.

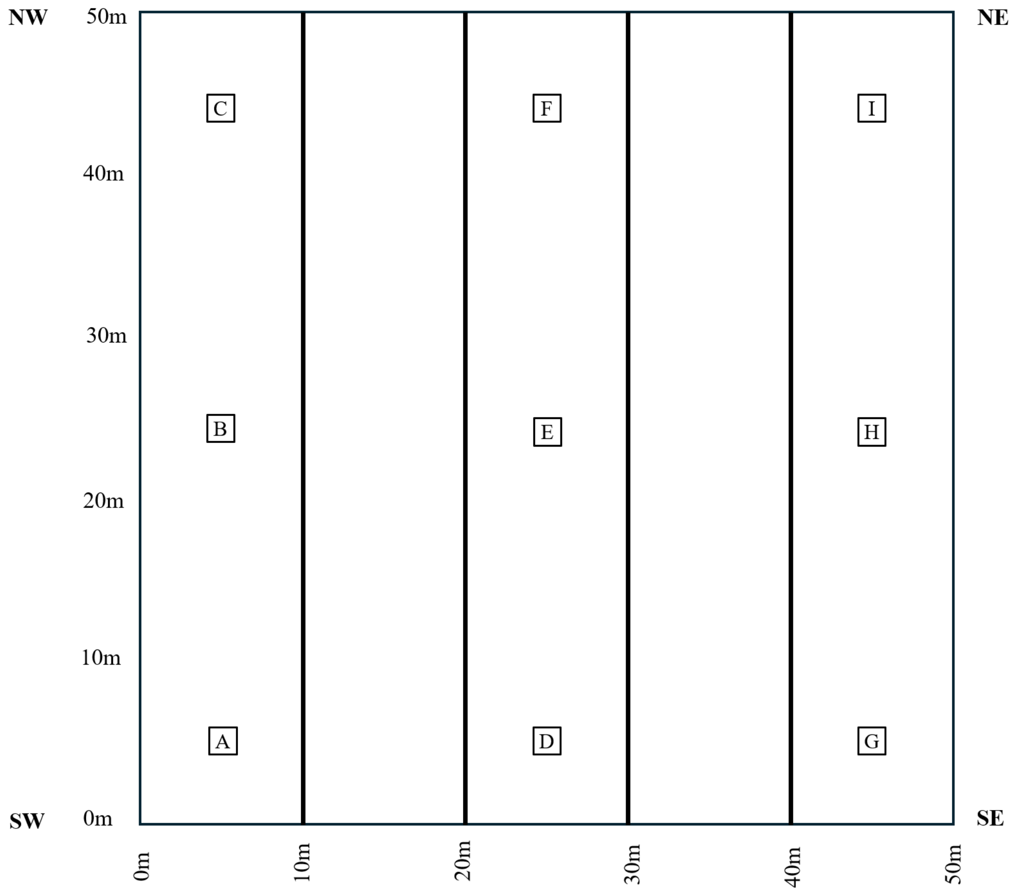

Figure 3.4: Layout of the 50m by 50m field sites established in July and August, 2023. GPS coordinates were obtained at each corner and at the center of the site. NDVI photos were obtained for each quadrat, labelled A-I.

After masking and exporting photos in ENVI, they were imported into RStudio to extract spectral band values and calculate GNDVI. Specifically, GNDVI was calculated for each pixel within the image and then all GNDVI values were averaged to obtain a single GNDVI value for each quadrat. GNDVI for each pixel was calculated using the equation:

$$GNDVI = \frac{\frac{Green_{plot}}{Green_{stan}} - \frac{NIR_{plot}}{NIR_{stan}}}{\frac{Green_{plot}}{Green_{stan}} + \frac{NIR_{plot}}{NIR_{stan}}}$$

where Green represents the spectral reflectance of the green band, and NIR represents the reflectance of the NIR band. The subscript "plot" refers to the quadrat photo, and "stan" refers to the white balance photo. Please see *Figure 6* for process depiction. An average GNDVI value was calculated for each field site from the nine mean quadrat values. GPS locations were then imported into ArcGIS to create points. These points were then connected to create polygons to be used for extracting information from the MODIS satellite data.

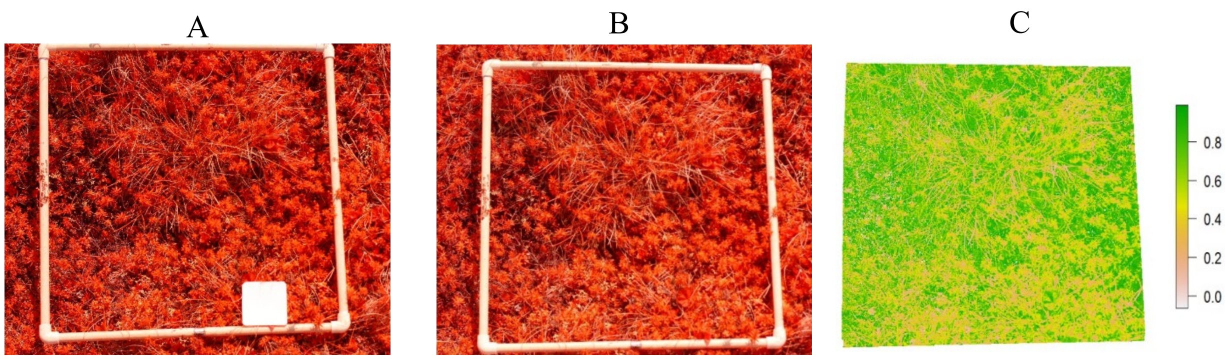

Figure 3.5: Process for calculating GNDVI plot photos with the modified digital SLR camera. A) Sample photo taken for standardization (stan). B) Sample of quadrat photo (plot). C) Result of GNDVI for the quadrat after masking, standardization and calculation.

EVI 16-day composite satellite images were downloaded from MODIS (MOD13Q1 V6.1 product) using Google Earth Engine for dates closest to the field surveys (July 12, 2023, and July 28, 2023). MODIS is an advanced imaging sensor aboard NASA's Terra and Aqua satellites, designed to capture global dynamics in land, ocean and atmospheric processes. In contrast to fine-scale satellite data that collects data at a higher spatial resolution but less frequently, MODIS provides high temporal resolution data, with observations taken every 1-2 days, but the 16-day composite images include the best-quality pixels over a 16-day period, reducing the effects of cloud cover and atmospheric interference. This composite approach provides a more reliable dataset for monitoring vegetation at moderate spatial resolution. EVI values were extracted corresponding to the shape of the field site polygons, averaging values for polygons that overlapped multiple MODIS pixels.

A Spearman's rank correlation coefficient was calculated in R to assess the direction and strength of the relationship between ground-based GNDVI and satellite EVI indices of plant productivity. Unlike Pearson's correlation, which assumes a linear relationship and normally distributed data, Spearman's rank correlation is a non-parametric test that measures the strength and direction of a monotonic relationship between two variables. It works by ranking the values of the variables and assessing how well the rankings correspond, making it more suitable when the data is not normally distributed or have outliers (Zar, 1972). This method was chosen because the distribution of the data violated the assumptions of normality required for Pearson's R. A moderate to high positive correlation would indicate strong similarities between satellite data and plot-level data. This comprehensive approach ensures the reliability of satellite data for monitoring vegetation dynamics.

Model Development

The development and tuning of the two random forest models enabled me to assess the influence of static and dynamic habitat characteristics on caribou use. The models' predictive capabilities were validated using independent datasets, and variable importance was assessed using multiple methods to ensure robustness and reliability in our findings.

Prior to the model development, I calculated Spearman's rank correlation coefficients between all predictor variables and the response variable to assess bivariate relationships and avoid collinearity issues (Graham, 2003). Pearson's correlation coefficient works best when the underlying distribution is normal, data has few outliers, and relationships are linear (Schober, 2018), which was not the case for the biophysical variables assessed in this study. Many predictor variables significant to caribou were included, including static and dynamic variables representing change (*Table 3*). A random noise variable with values ranging from 1 to 100 was also created to ensure accuracy in calculating variable importance. During initial model tuning, variables that performed worse than random noise in relative importance (IncNodePurity) were excluded from the final models. The final models only included variables that were not highly correlated, according to Spearman rank correlation, and significant after model tuning.

Table 3.1: Descriptions of the predictor variables used in the random forest models to assess the influence of the environment on caribou distribution.

Predictor Variable	Cell Size	Value Range	Reason For Inclusion	Raw Data Source	Processing
Elevation	500m	0 to 617.1 M.A.S.L	Areas of high elevation (e.g., eskers) are often used as migratory paths as vegetation is frequently shorter and easier to travel across.	ArcticDEM digital surface model from DigitalGlobe's WorldView-1, WorldView-2 and WorldView-3 satellites (Porter et al., 2023)	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent
Time Since Last Fire	250m	0 to 100 years	Areas exposed to fire influence caribou forage availability.	Fire history shapefile adapted from the Canadian Wildland Fire Information System by Micheal Stefanuk (CFS, 2021)	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent
Average Summer Windspeed	1000m	14.5 to 18.9 km/hour	Higher wind speeds can provide caribou with relief from biting insects, such as mosquitos and black flies.	TerraClimate datasets that combine high resolution climate normals from the WorldClim dataset, with coarser spatial resolution and time-varying data from the Climatic Research Unit Ts4.0 and the Japanese 55-year Reanalysis (Abatzoglou et al., 2018)	Raster files for June, July and August (1997 to 2017) were clipped to the herd's summer range extent and averaged to yield an average windspeed over the study period
Trend In Maximum Summer Temperature (2000-2017)	1000m	0.07 to 0.10 °Celsius/ Year	Higher daytime temperatures influence insect activity and accelerate plant growth, directly affecting forage availability for caribou	TerraClimate datasets combine high resolution climate normals from WorldClim dataset, with coarser resolution data from the Climatic Research Unit Ts4.0 and the Japanese 55-year Reanalysis (Abatzoglou et al., 2018)	Monthly averages for June, July and August from 1997 to 2017 were calculated to obtain annual averages. A Theil-Sen trend analysis was completed in RStudio to yield a single dynamic variable
Trend In Minimum Summer Temperature (2000-2017)	1000m	0.01 to 0.05 °Celsius/ Year	Lower nighttime temperatures affect frost occurrence and soil temperature, which influence seasonal plant productivity	TerraClimate datasets combine high resolution climate normals from WorldClim dataset, with coarser resolution data from the Climatic Research Unit Ts4.0 and the Japanese 55-year Reanalysis (Abatzoglou et al., 2018)	Monthly averages for June, July and August from 1997 to 2017 were calculated to obtain annual averages. A Theil-Sen trend analysis was completed in RStudio to yield a single dynamic variable

Trend In Snow Melt Timing (2001 - 2015)	500m	-3.84 to 2.29 days/year	Early snow melt can cause earlier departure from the winter range and later snow melt might cause later arrivals on calving and summer grounds	Derived from the MODIS (Collection 6) product provided by NASA's Terra satellite. The change in snow melt timing was calculated using an ordinary least-squares linear regression by Micheal Stefanuk. (O'Leary III et al., 2017)	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent
Trend In Sos (2000 - 2017)	250m	-5.0 to 4.94 days/year	SOS has seen to be advancing due to warming temperatures. These changes may result in changes to migratory movements that allow caribou to access preferred forage	Derived from the MODIS (Collection 6) product provided by NASA's Terra satellite (O'Leary III et al., 2017). Dearborn & Danby (2021) calculated annual SOS using curve fitting and modelled EVI values, followed by pixel-wise linear regressions to determine trends.	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent
Trend In Eos (2000-2017)	250m	-5.0 to 5.0 days/year	EOS has seen to be delayed, due to warming temperatures and delayed snowmelt, which influences fall caribou migrations	Derived from the MODIS (Collection 6) product provided by NASA's Terra satellite (O'Leary III et al., 2017). Dearborn & Danby (2021) calculated annual EOS using curve fitting and modelled EVI values, followed by pixel-wise linear regressions to determine trends.	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent
Trend In Maximum EVI (2000 - 2017)	250m	-0.08 to 0.10 EVI/year	Caribou may be influenced by increasing plant productivity and may be correlated with regions that are seeing increased EVI	Derived from the MODIS (Collection 6) product provided by NASA's Terra satellite (O'Leary III et al., 2017). Dearborn & Danby (2021) calculated annual EVI using curve fitting and modelled EVI values, followed by pixel-wise linear regressions to determine trends.	Reprojected to Lambert Conformal Conic using bilinear interpolation and clipped to the herd's summer range extent

The dataset used to generate the random forest models was a random subset of 15% (56,640) of all pixels within the summer range extent. Pixels that fell within 500m of a lake were excluded from sampling to minimize potential interference from lake influence on the data. A minimum of 1% of all pixels is considered acceptable to avoid auto-correlation and processing time in models with large datasets (Genuer et al., 2017). Although independent validation datasets are not necessarily required in random forest modelling, they were used to assess the model's generalization capability (Cutler et al., 2007). For both models, the data was randomly divided into a training dataset (70% of all cases) and a validation dataset (30% of all cases). Training on 70-80% of the dataset is considered optimal for accuracy without overfitting (Gholamy et al., 2018). Each subset contained many observations, each representing a 500mx500m pixel within the summer range extent. I modelled the relationship between changes in relative caribou use and the environmental variables using the training dataset and assessed prediction accuracy using the validation dataset.

I used the "randomForest" package in R to develop each model (Liaw & Weiner, 2022). Model tuning, which is the process of refining model parameters to improve performance, focused on two key parameters. First, the number of trees (ntree) balances processing time and accuracy, with more trees improving accuracy but increasing computation time. Second, the number of variables considered at each split (mtry) was optimized using the *TuneRF* function, which identifies the value that minimizes OOB error rate by testing values around the default (total variables/3; Breiman et al., 1984).

Assessing Variable Importance

The underlying premise of the habitat modeling approach is that certain environmental variables influence caribou summer range distribution more than others. Identifying these key variables is crucial for understanding habitat preferences. The independent variables predicted to be important in each model help to identify which habitat characteristics influence changes in relative caribou summer distribution. The randomForest package has two internal methods, percent increase in mean square error (%IncMSE) and increase in node purity (IncNodePurity), for assessing variable importance (Breiman et al., 1984). Of these two methods, %IncMSE is regarded as a more reliable measure, though it may be less likely to detect relevant variables when correlation between variables increases (Gregoretti et al., 2014). These methods evaluate the model's predictive capability when each independent variable is replaced by random noise. The resulting model deterioration when a variable is removed is a measure of variable importance, with an increase in mean squared error indicative of increased variable importance. More specifically, %IncMSE is the difference between the misclassification rate for the training dataset and the OOB data, averaged over all trees and divided by the standard deviation of the differences (Breiman et al., 1984). This was used in model tuning, and any variable that ranks below random noise was removed from the model.

To account for the influence that independent variables have on each other, I opted to use conditional permutation importance (CPI), which assesses the magnitude of influence for predictor variables on the response variable (Debeer & Strobl, 2020). This method is similar to %IncMSE but considers the interdependence of variables. CPI is assessed by permuting within a conditioning grid, which is a framework that organizes data into groups based on the values of other variables. Essentially, this grid "conditions" the evaluation of a variable's importance by holding the values of other predictor variables constant, ensuring that the importance is measured while accounting for correlations between other predictor variables. This approach gives a more accurate reflection of each variable's unique contribution. The CPI was calculated using the *permimp* R package (Debeer, Hothorn & Strobl, 2022).

I also used partial dependence plots to quantify the effect of each habitat characteristic on relative caribou use. This method is considered one of the most effective means of interpreting random forest model results. Partial dependency plots graphically illustrate the influence of an independent variable on the dependent variable across its range of values while averaging all other variables included in the model (Friedman, 2001). Specifically, the y-axis shows the average of all modelled predictions for a specified value of that predictor variable.

3.3 Results

3.3.1 Summer Range Distribution

A total of 13,737 mean daily locations of cows during the defined summer time-frame (June 29th to September 6th) were calculated from 1997 to 2017. The cumulative 99% contour line encompassed an area of 118,320 km^2, including 13,622 out of the total 13,737 mean daily locations. This refined satellite dataset represents the geographical extent of the Bathurst summer range over the entire study period (*Figure 2*).

These data were subdivided into 21 separate yearly datasets to determine relative summer range distribution on an annual basis. The KDE varied between years, and contour plots of the estimated densities were generated to define areas with especially high concentrations or high relative habitat use. The annual summer range was defined as the 95% UD contour line and the high-use area, or cluster, was defined as the 50% UD contour line to indicate areas of intense habitat utilization. The largest annual summer range was observed in 1999 (1351.4 km^2), covering an area of 11,178 km^2, while the smallest annual summer range was observed in 2016, with an area of 2,377 km^2 (*Table 4*). Similarly, the largest core use area (50% UD) was observed in 1999 (3655.0 km^2) and the smallest annual core use area was observed in 2016 (548.1 km^2) (*Table 4*).

The largest annual core summer ranges (50% UD) occurred in the first decade of the study period (1997-2008), frequently exceeding 1,500 km^2 (*Figure 7*). After 2008, the annual core summer range consistently remained below 1,500 km^2. Following 2007, a northward shift of the summer range becomes noticeable. This shift is particularly evident after 2012, reflecting a significant decrease in relative use of the previously utilized southern expanse of the herd's summer range.

Table 3.2: Results of the annual Kernel Density Estimation of the Bathurst caribou herd's summer range extent. For each year, there is a corresponding column for the number of collars and the total number of mean daily locations calculated and used in the KDE. The right side of the table presents the area for each of the density values, including the core use summer range (50% UD) and the total annual summer range (95% UD).

Year	# of Collared Cows	# of Daily Mean Locations	Area (Km^2)	
			50%	95%
1997	8	78	1635.1	6989.4
1998	10	46	1774.6	6802.9
1999	14	157	3655.0	11178.4
2000	13	150	2060.0	7086.3
2001	13	156	2984.0	11173.9
2002	11	337	1956.2	7219.4
2003	11	420	1998.5	7700.7
2004	5	239	1617.8	6270.3
2005	19	848	2012.9	6970.5
2006	14	632	1351.4	4265.8
2007	19	898	2162.7	6758.3
2008	12	516	1672.7	5930.6
2009	12	814	1460.7	6207.8
2010	19	1043	953.7	3375.0
2011	17	822	1392.5	4744.8
2012	21	1162	1666.5	5403.5
2013	13	723	813.5	2719.9
2014	18	981	914.6	2991.7
2015	31	1790	599.1	2428.8
2016	25	1499	548.1	2377.4
2017	28	1755	708.7	2456.2

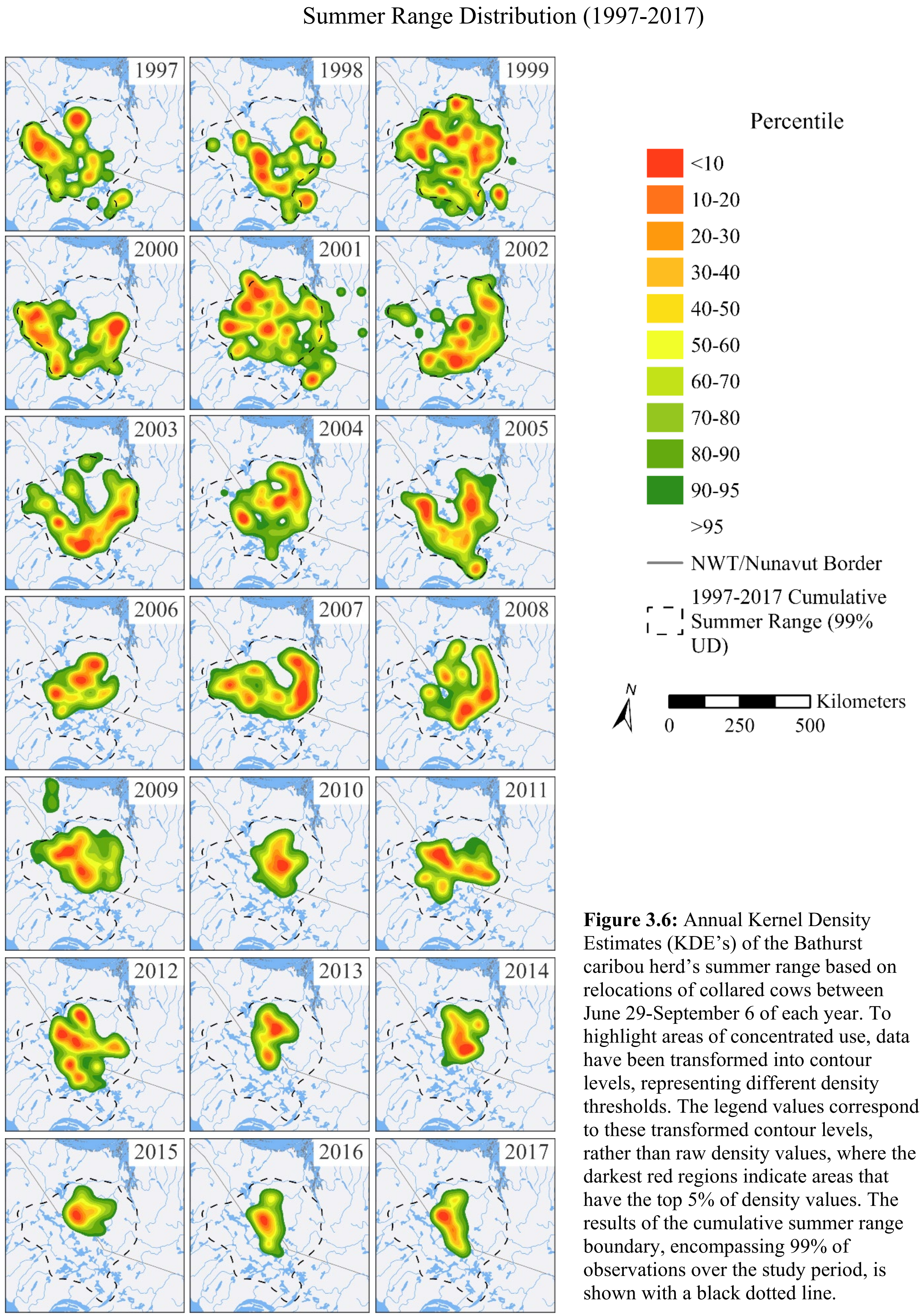

Figure 3.6: Annual Kernel Density Estimates (KDE's) of the Bathurst caribou herd's summer range based on relocations of collared cows between June 29-September 6 of each year. To highlight areas of concentrated use, data have been transformed into contour levels, representing different density thresholds. The legend values correspond to these transformed contour levels, rather than raw density values, where the darkest red regions indicate areas that have the top 5% of density values. The results of the cumulative summer range boundary, encompassing 99% of observations over the study period, is shown with a black dotted line.

3.3.2 Change in Relative Summer Distribution

The trends in relative caribou use over the study period indicated significant contractions of the herd's summer range (*Figure 8*). Statistically significant ($p \leq 0.05$) declines in relative caribou use accounted for 16.8% (19,827 km^2) of the cumulative summer range, while areas with a significant increase in use comprised just 4.9% (5,808 km^2). Areas that exhibited no change in relative caribou use constituted 19.2%, or 22,749 km^2, of the cumulative range. The remaining 59.1% of the region did not experience statistically significant trends. Regions that experienced significant increases in relative habitat use are centered around Contwoyto Lake, southwest of Bathurst Inlet, near the border of the Northwest Territories and Nunavut.

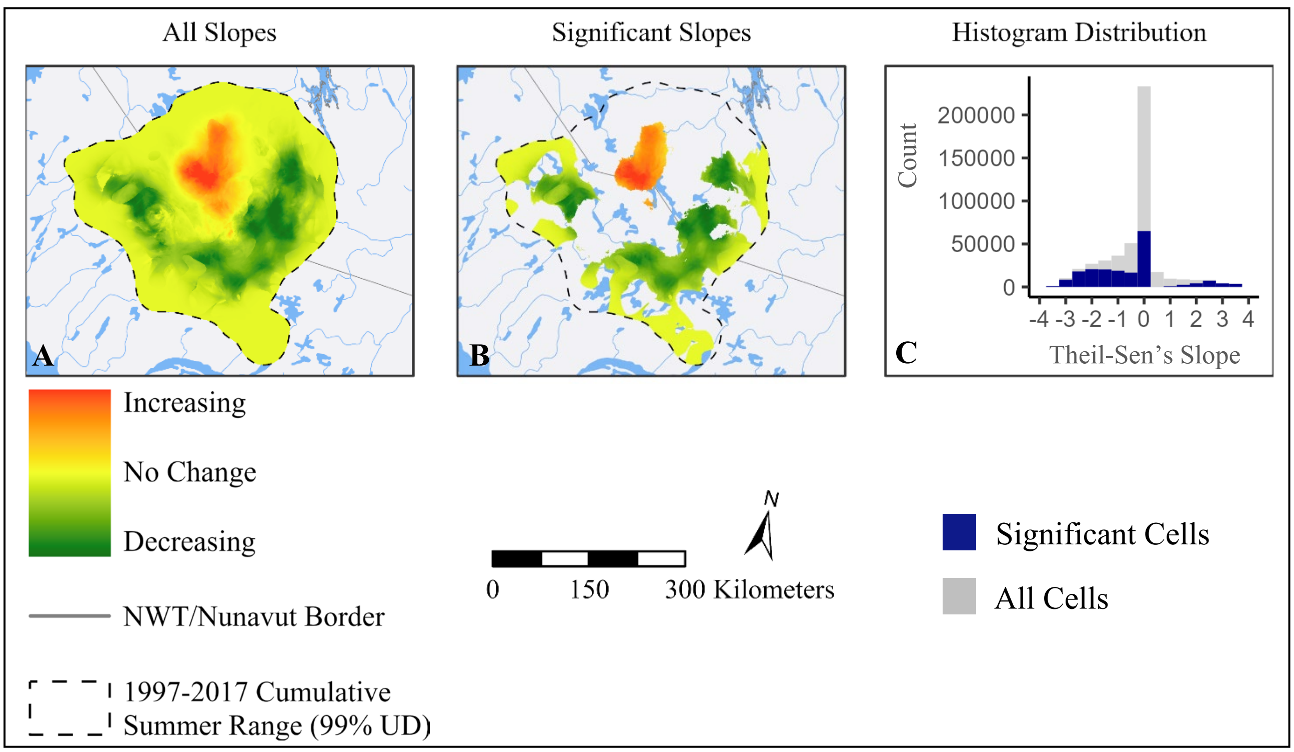

Figure 3.7: The results of the pixel-wise Theil-Sen regression analysis demonstrating the change in relative caribou use from 1997 to 2017. A presents slope values for all areas, while B presents only values that are statistically significant ($p \leq 0.05$). In both maps, red is indicative of areas that experienced an overall increase in annual relative caribou use, green is indicative of areas that experienced a decrease, and yellow represents areas that experienced little or no change. The histogram in C illustrates the distribution of data values based on the number of cells, with grey representing all slope values, and blue representing only those considered statistically significant.

3.3.3 Relative Habitat Use Modelling

As described in section 3.3.5, vegetation surveys were conducted in July and August, 2023 to validate the accuracy of the MODIS satellite imagery used in our random forest modelling. Prior to calculating the relationship between satellite data and plot-level measurements, two data outliers were removed. Specifically, satellite measured EVI values for sites 3 and 16 were clearly inconsistent from the rest of the dataset, possibly due to the influence of cloud cover or other sensor and viewing geometry effects from the 16-day composite images (Holben, 1986; Li & Strahler, 1992). After this was completed, Spearman's rank correlation coefficient between GNDVI from the ground-based measurements and EVI from the MODIS satellite data was calculated as 0.31 ($p=0.10$), indicating a moderate, positive correlation (See *Appendix B* for raw values).

The final dataset used in the random forest modelling was also assessed prior to model development to avoid issues related to collinearity (Graham, 2003). Results from this Spearman's rank analysis indicated no strong correlations ($r>0.70$) between any single predictor variable and changes in relative habitat distribution. However, some predictor variables exhibited high correlation values with each other (*Figure 9*). Specifically, mean summer wind speed (km/hr) correlated with elevation ($r = 0.70$), trends in LOS correlated with trends in the EOS ($r = 0.77$) and SOS ($r=0.71$), and trends in summer maximum temperature correlated with trends in summer minimum temperature ($r = 0.74$). Three of these highly correlated variables (wind speed, LOS, minimum temperature) were consequently removed from the subsequent modelling to mitigate the effects of multicollinearity (Nicodemus & Malley, 2009).

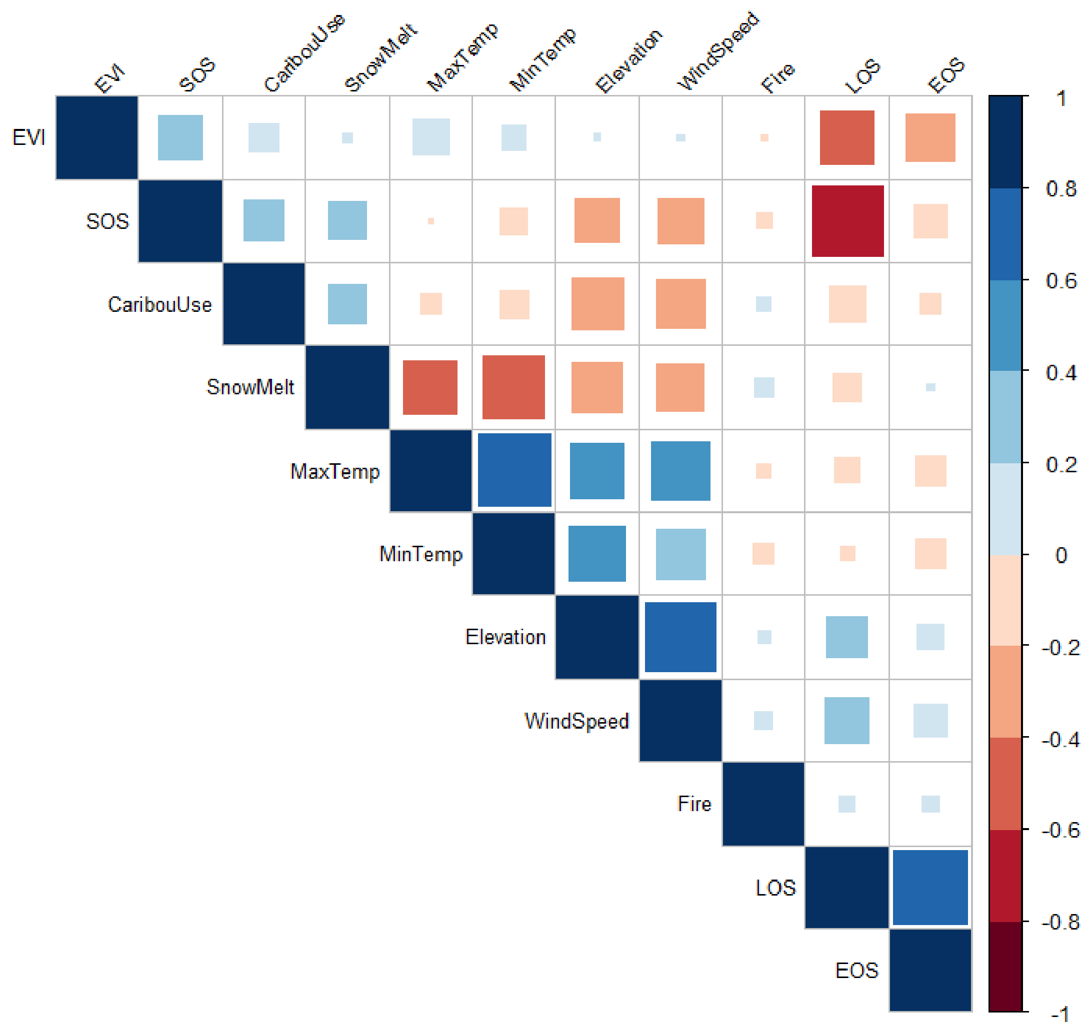

Figure 3.8: A Spearman rank correlation matrix between each of the predictor and the response variables used in the random forest regression models. Darker and larger squares within the matrix represent a strong correlation between the two variables, with dark red indicating a strong negative correlation and dark blue indicating a strong positive correlation.

Prior to model development, tuning was conducted to assess variable importance relative to the random noise variable. Variables ranking below random noise (time since last fire (in years)) were excluded. Additionally, the tuning process involved determining the optimal number of regression trees and the number of variables considered at each split. Initially, ntree was set to the default of 1000, but by assessing the OOB error rate, it was found that 300 trees were sufficient to achieve the lowest error rate without overfitting. The mtry parameter, which controls how many predictor variables are tested at each node split, was optimized using the tuneRF function. The optimal values were four for the model containing both static and dynamic variables, and two for the model containing only dynamic variables. This ensured the model's predictive power while avoiding overfitting or underfitting.

The first random forest model, which included seven variables – both static (elevation) and dynamic (maximum temperature, snow melt timing, maximum EVI, LOS, and SOS) – explained 52.8% of the variation in relative caribou use based on the pseudo-R^2 from the training data. Validation of this model on an independent dataset indicated that these variables explained 50.1% of the variation in relative caribou utilization. The CPI scores highlighted the most significant variables ranked as follows: trend in summer maximum temperature, elevation, trend in snow melt timing, trend in SOS, trend in maximum EVI, and trend in EOS (*Figure 10A*). The importance of these variables was corroborated by the %IncMSE and IncNodePurity metrics. Considering the strong correlation between elevation and average summer wind speed, wind speed could also be inferred as an important factor affecting the change in caribou utilization over time.

The second random forest model, focusing exclusively on six dynamic variables (trend in maximum summer temperature, trend in snow melt timing, trend in maximum EVI, trend in LOS, and trend in SOS), accounted for 34.2% of the variation in relative caribou use. When validated, this model's explanatory power increased to 64.3%. Similar to the first model, the key influencing variables, according to the CPI scores, were trends in maximum summer temperature, snow melt timing, SOS, EOS, and maximum EVI in order of importance (*Figure 10B*).

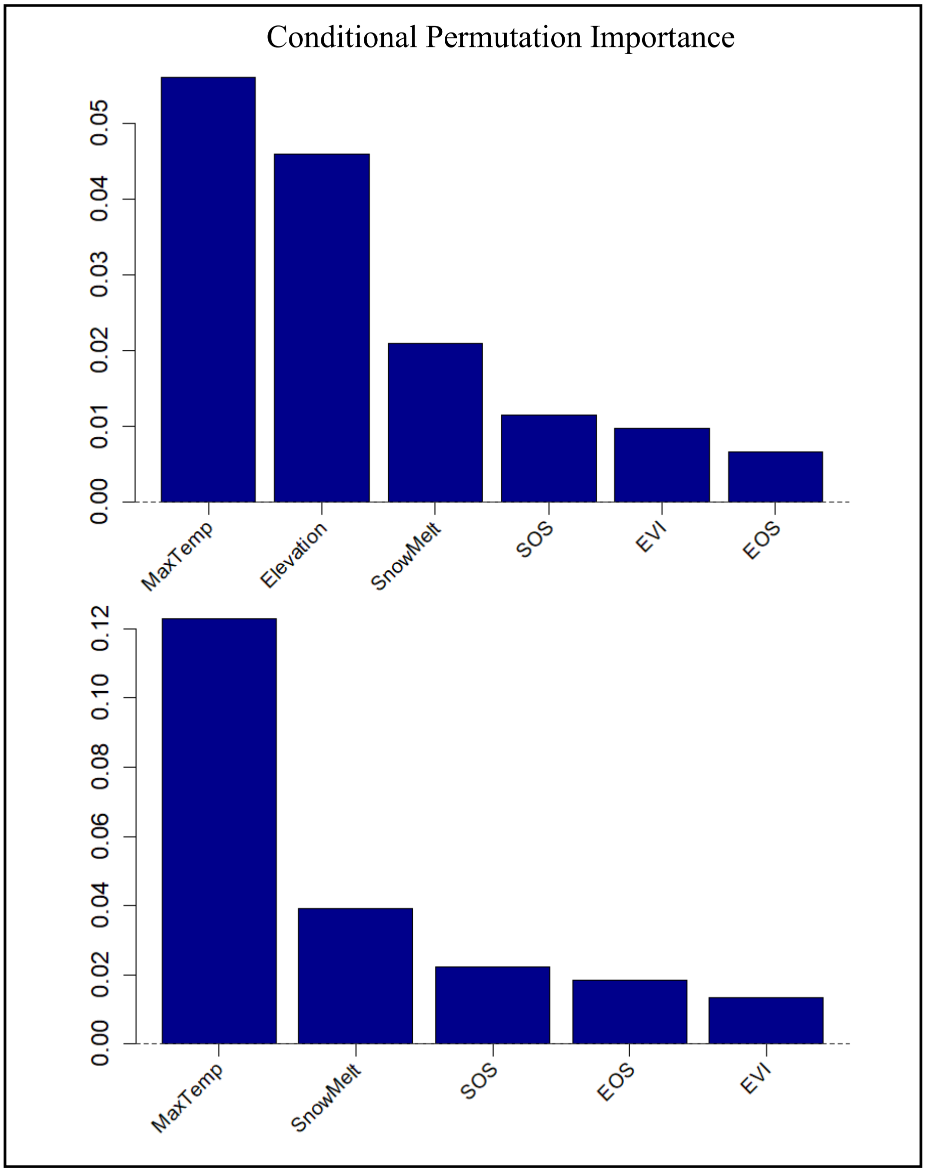

Figure 3.9: Variable importance results, based on CPI scores, highlighting the most significant variables to the random forest models. A – Results when both static and dynamic variables were included. B – Results when only dynamic variables were included. The y-axis is a normalized value, where the values do not need to be considered to assess the model.

Partial dependence plots for both models revealed non-linear relationships between each predictor variable and the response variable (*Figure 11* and *12*). Trends in maximum summer temperature exerted the most substantial impact on both regression models. The partial dependence plots indicated that increases in maximum temperatures above approximately 0.10 degrees Celsius per year were associated with significant decreases in caribou use. The partial dependence plot for elevation indicated that caribou use was greatest in areas of higher elevation, specifically those above approximately 500 meters above sea level (M.A.S.L.).

In both models, the relationships between caribou use with phenological and productivity metrics (e.g., snow melt timing, SOS, EOS, max EVI) were less pronounced, compared to maximum temperature and elevation. Specifically, caribou relative habitat use decreased when the timing of snow melt was delayed by any number of days. Similarly, SOS corresponded with a decrease in caribou use when advanced by one day, increases in maximum EVI corresponded with a decreased relative habitat use, and caribou use was relatively constant, with a slight decrease, no matter the change in EOS. The productivity and phenological metrics were marginally different in the dynamic model, however, slight changes remained in relative caribou use. SOS corresponded to a greater decline in caribou use, in comparison to the static model, when SOS was delayed. When advanced, EOS corresponded to a slight decline in caribou use, but this trended toward no change in caribou use when advanced by one day or delayed by any number of days. Similarly to the static model, maximum EVI increases corresponded to a decrease in caribou use (*Figures 11* and *12*).

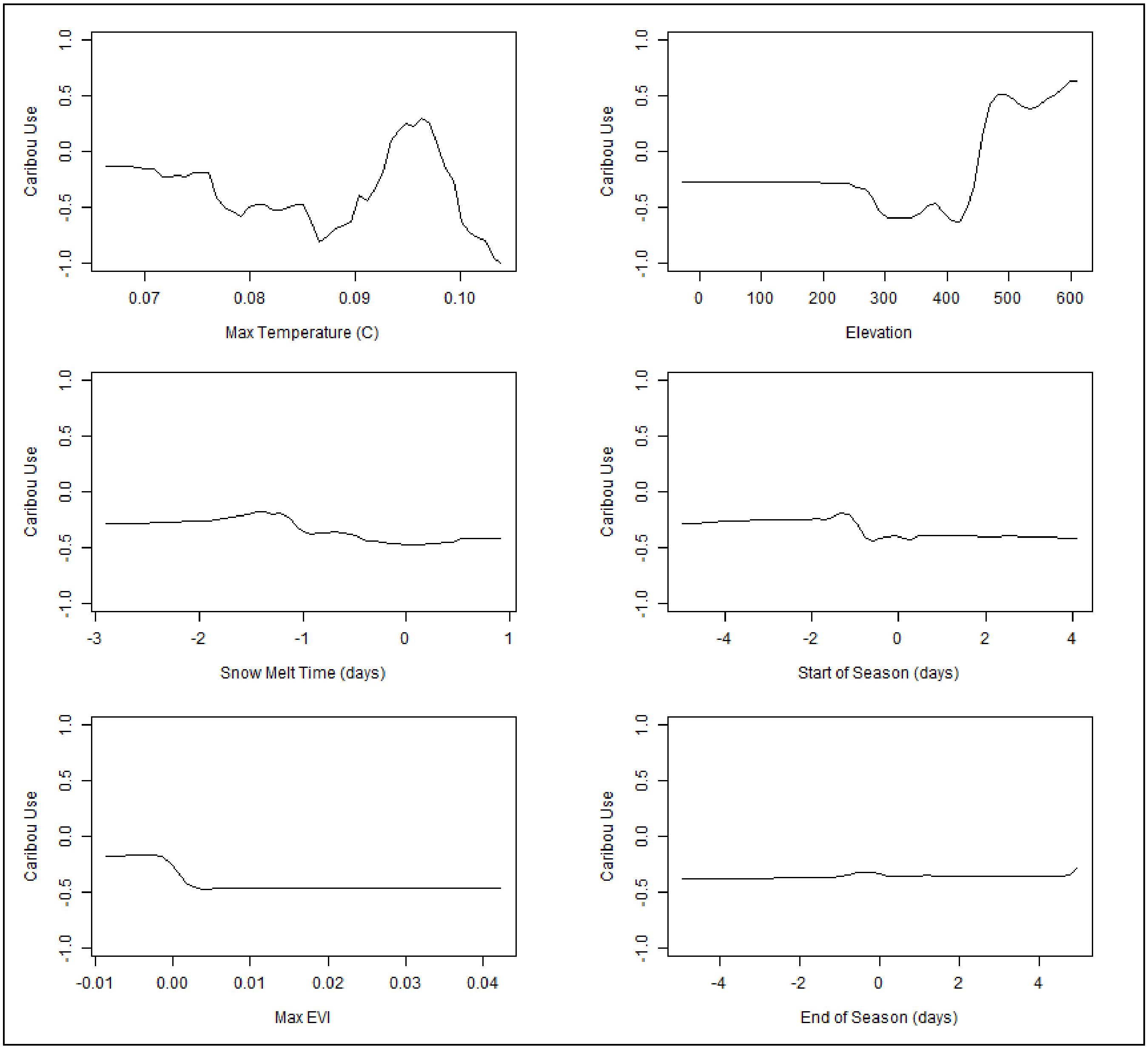

Figure 3.10: Partial dependence plots for the static random forest model, between each predictor variable and results of the Theil-Sen analysis. Values on the x-axis represent the predictor variable units and the y-axis values represent the relative change in caribou use.

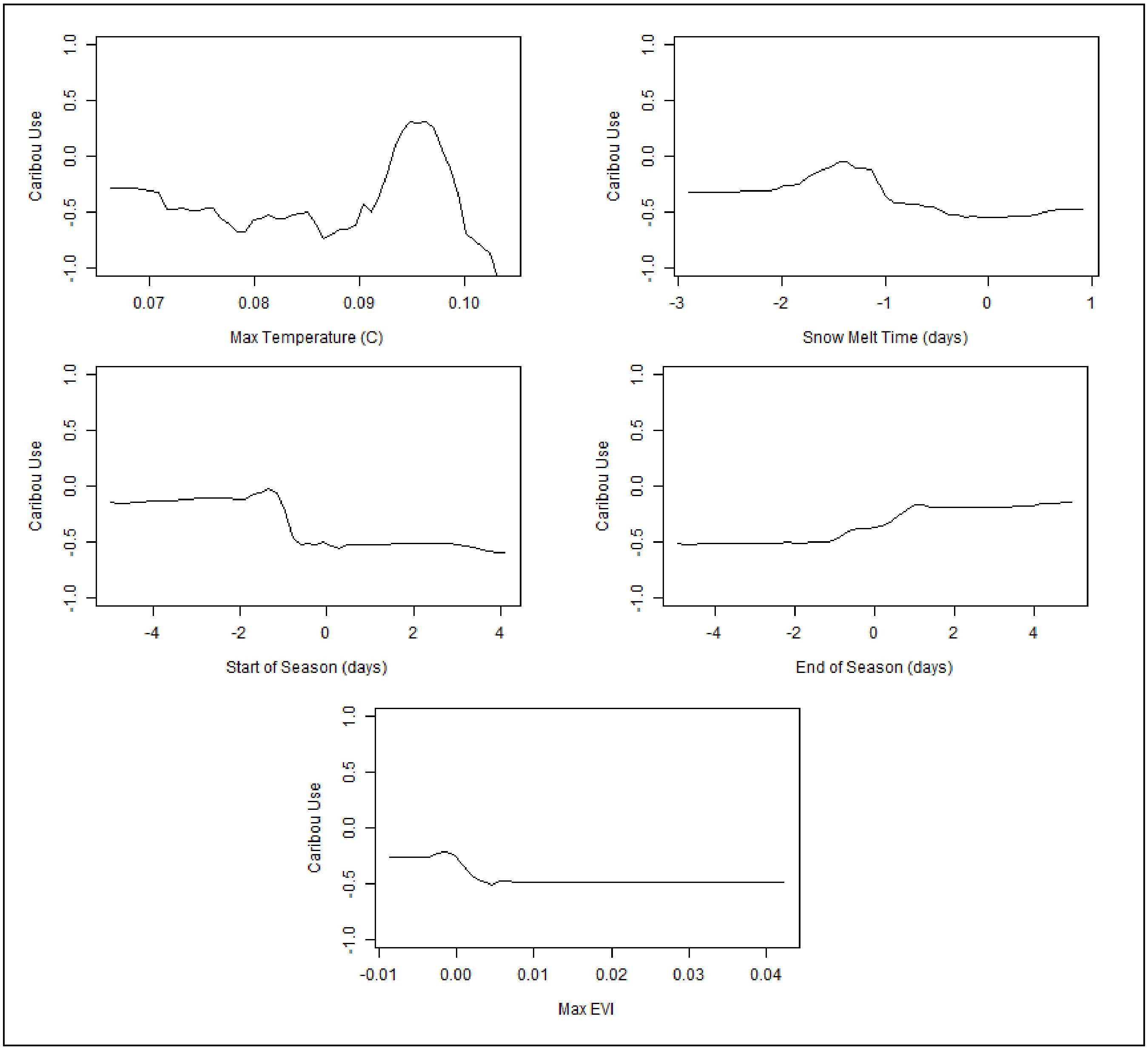

Figure 3.11: Partial dependence plots for the dynamic random forest model, between each predictor variable and the results of the Theil-Sen analysis. Values on the x-axis represent the predictor variable units and the y-axis values represent the relative change in caribou use.

3.4 Discussion

This study leverages 21 years of telemetry and environmental data to investigate shifts in the Bathurst herd's summer range and identify key environmental predictors of these changes amid a significant population decline. The findings reveal marked contractions and a general northward shift of the summer range, with the most notable changes occurring after 2009. The two random forest models, integrating various environmental variables, identified changes in mean summer temperature as the primary predictor influencing these shifts in relative summer range distribution. Following this, elevation, and the trends in snow melt timing, SOS, EVI, and EOS were also significant predictors. The following discussion delves into these results, offering potential explanations and drawing comparisons with existing literature.

3.4.1 Change in Relative Summer Range Distribution

The average summer range, delineated from 99% of daily mean locations, spanned approximately 13,516 km^2, located southwest of the Bathurst Inlet and north of Great Slave Lake. This summer range distribution is similar to the summer home range boundaries reported by the GNWT from 1996 to 2014 (GNWT, 2019), despite the use of different analytical methods. Over the study period, a total of 115 telemetry observations were excluded from this analysis. Two notable individuals (BGCA209 and BCGA126) had many outlier observations and appear to have emigrated into neighbouring herds. Specifically, BGCA126 was found east of the Bathurst Inlet, near the Beverly herd's calving and summering area and BGCA209 was located northwest of the summer range, within the Bluenose-East herd's calving and summer range. Similarly, Boulanger et al. (2011) reported approximately 5% immigration/emigration rates for the Bathurst herd, using a different approach to telemetry data analysis. Other excluded observations were from various individuals located on the periphery of the range, with a cluster between Clinton-Colden Lake, Healy Lake and Moraine Lake.

The analysis of summer range distribution using KDE and Theil-Sen regression techniques revealed substantial change over time in both extent and distribution. Analyses assessing the annual relative summer range distribution indicate a 35% decline in home range and a 37% decline in core-use areas over the study period. The findings of significant contractions witnessed over study period align with the same time period as the herd's population decline, from an estimated 350,000 animals in 1997 to 8,200 animals in 2018 (GNWT, 2019). Specifically, the largest home range was observed in 1999 (11,178 km^2) and the largest core use area was in 2006 (6,970 km^2). The smallest ranges were observed later in the study period, with the smallest home range (2,377 km^2) km and core use area (548 km^2) both occurring in 2016. Notably, the trends in decreasing range sizes were especially evident after 2009, corresponding with the herd's most rapid decline between 2006 and 2009, from an estimated 55,600 to 15,900 individuals (Nishi et al., 2007; Nishi et al., 2014). Specifically, summer home ranges were approximately 42% smaller in areal extent when comparing home ranges before 2006 and after 2009.

The range contraction is not unusual for animals that have recently seen a drastic decline and is an expected outcome of a population low (Hinkes et al., 2005). It is possible that reduced relative summer range extent may be due to density-dependent resource utilization, where the availability of preferred resources increases as population size decreases (McLoughlin et al., 2006). With the significant decline in the Bathurst herd and adjacent herds, caribou may no longer need to spread out widely to access preferred habitat conditions. Similarly, summer mortalities among caribou, which are often due to predation by wolves, may lead the herd to prioritize safety from predators over habitat preferences, a pattern observed in other ungulate populations (Smith et al., 2022). The combination of a population decline allowing for the availability of more resources and the threat of predation are likely responsible for the observed reduction in relative summer range distribution.

The northward shift of the Bathurst herd's summer range mirrors trends already observed in Arctic species, including other barren-ground caribou populations (Chen et al., 2011; Davidson et al., 2020; Veitch, 2001). In this study, the most significant contractions occurred at the south-eastern limits of the summer range, with core-use areas consistently occupying the north-west segment of Contwoyto Lake. Using different analytical techniques, Klaczek (2015), Virgl et al. (2017) and Mennell (2021) also found that the extent of the Bathurst herd's summer range consistently declined and trended northwards towards Contwoyto Lake during the summer period from 1996 to 2012, 1996 to 2013, and 1997 to 2019, respectively. Based on predictive modelling (Sharma et al., 2009), this northward trend is anticipated to continue for barren-ground caribou herds due to several factors, including longer ice-free periods, extreme weather events, altered forest fire regimes and changes to insect and predator distribution. For many migratory species, such as barren-ground caribou, annual movements to summer ranges are largely related to environmental conditions and escaping predation (Bergerud, 2000; Heard & Williams, 1996; Hughes et al., 2009; Post et al., 2003). With warming temperatures and the associated impacts related to nutritious forage, predator distribution and insect prevalence, it is not unreasonable to assume that changing environmental conditions compel caribou to shift their ranges northward to access habitat conditions that were historically found in more southern locations.

It is important to mention that this research did not account for changes in population size, and it was conducted on a small subset (5 to 31 individuals) of the total Bathurst caribou herd. The results represent an index of trends in relative range distribution rather than raw density values, highlighting areas of increasing and decreasing use. The trends in range distribution that do not account for population declines and the small proportion of collared individuals has the potential to influence what is considered the herd's range and core use areas. However, on the summer range, cows and bulls arrive to the summering grounds, where new mothers and calves form small to medium sized groups (10 to 100 individuals), while bulls form smaller groups that are within the vicinity of the cow aggregations (Pruitt, 1960, Mennell, 2021). Therefore, due to these synchronized movement patterns during summer (Gunn & Dragon, 2002; Adamczewski et al., 2020), the results of this research are assumed to generally reflect the summer range distribution and utilization patterns of the entire herd, rather than representing trends in caribou density.

3.4.2 Influence of the Environment on Relative in Range Use

Vegetation Surveys

Correlation analysis between 31 plot-level GNDVI and satellite-derived EVI measurements revealed a significant positive relationship. This was expected, as they are both indices used for measuring plant productivity and other studies have shown that EVI and GNDVI follow similar seasonal profiles (Halabuk et al., 2013; Mezera et al., 2021). The relationship was not as strong as expected, however discrepancies between the two indices have been observed in previous studies (e.g., Spruce et al., 2011) and could be due to differences in spectral bands, spatial resolution, and temporal resolution.

The MODIS EVI and plot-level GNDVI measures use different spectral bands that are sensitive to seasonal and spatial variability (Halabuk et al., 2013). Specifically, GNDVI is calculated using the NIR and green spectral bands, while EVI uses the NIR, blue and red spectral bands. During peak biomass from mid-July to early-August (Berner et al., 2024), the green band, used in GNDVI, is known to be more sensitive at capturing variability in chlorophyll, nitrogen content and leaf area index, while the red band is less sensitive at accounting for these biophysical conditions (Gitelson et al., 1996). In addition, differences in spatial and temporal resolution also likely influenced the relationship between GNDVI and EVI measurements. Specifically, the MODIS EVI data is collected at a much coarser scale ($250m^2$) than our plot-level GNDVI measurements ($0.25m^2$), potentially incorporating different vegetation and land cover types in a single pixel. In addition, MODIS data is a composite of 16 days, and one value is selected to represent the time frame, in comparison to plot level measurements that capture the vegetation state on a specific date (Didan, 2021). In tundra environments, where growing seasons are short, vegetation can vary from the first to the last day of a composite time frame (Guindin-Garcia et al., 2012), which can introduce discrepancies when comparing MODIS data with plot-level measurements (Spruce et al., 2011; Wessels et al., 2009). While spectral bands, spatial and temporal resolution can introduce some limitations, the satellite imagery remains reliable for understanding broad-scale patterns, without introducing significant errors (Aman et al., 1992).

Random Forest Modelling

The random forest models were used to assess the influence environmental conditions had on changes in relative range use by the Bathurst herd. The results indicated that climate, topographic and environmental conditions play a complex role driving changes to habitat distribution overtime. The measures of variable importance in both models were largely consistent with one another. The first model, which included elevation and trends in maximum summer temperature, snow melt timing, maximum EVI, EOS, and SOS as predictor variables, explained a substantial portion of the variation in caribou use in both training (52.8%) and validation (50.1%) datasets. The second model, which focused solely on the trends in maximum summer temperature, snow melt timing, maximum EVI, EOS, and SOS as the predictor variables, accounted for less variation in the training dataset (34.2%), but performed better on the validation dataset (64.3%). This suggests that while topographic features, such as elevation, provide significant explanatory power, trends in environmental factors are crucial for understanding overall range conditions and overall changes in range use.

Trends in maximum summer temperature emerged as the most significant predictor of change in relative caribou utilization in both models. Specifically, partial dependence plots indicated that increases in maximum temperatures above approximately 0.10°C per year were associated with notable decreases in relative caribou use. As Arctic temperatures are rising faster than in any other part of the world (Rantanen et al., 2022), many studies have observed and predicted that this change will impact northern species in various ways, including caribou (Joly et al., 2012; Le Corre et al., 2017; Mallory & Boyce, 2018; Sharma et al., 2009; Witter et al., 2012). There is limited research on the direct effects of extreme heat on caribou, but increased temperatures in general are believed to influence their population dynamics and habitat distribution (Hagemoen & Reimers, 2002; Ion & Kershaw, 1989). For example, Rosenmann & Morrison (1967) concluded that Alaskan reindeer cope well with heat stress when water is available; however, in the absence of water, high temperatures result in elevated heart rates and, in some cases, death. In addition, these impacts are not limited to maximum temperatures, as increases in maximum, minimum and average temperatures can influence caribou habitat distribution and utilization through indirect effects, such as changes in snow conditions, vegetation phenology and productivity, extreme weather events, predator distribution, and insect harassment. For example, rising temperatures are often associated with higher insect prevalence. This has been observed to prompt caribou to seek habitats that provide relief from both insects and heat, such as snow patches or windy areas. However, it is unclear whether this behavior is primarily due to heat stress or to avoid insect harassment (Ion & Kershaw, 1989; Hagemoen & Reimers, 2002).

Elevation was another significant factor related to changes in relative summer range use in the first model. Although the landscape is relatively flat within the summer range, partial dependence plots indicated that higher elevations (>500 M.A.S.L) were associated with increased caribou use, while regions <450 M.A.S.L experienced relatively no change in caribou use. There are several potential mechanisms to help explain this, including cooler temperatures, different vegetation types, or other ecological factors. The strong relationship between wind speed and elevation suggests that wind is also an important factor, likely due to its role in cooling body temperatures and mitigating insect harassment. Warmer temperatures have led to a greater prevalence and distribution of biting insects, such as mosquitos, black flies and oestrid flies, which can indirectly affect caribou by increasing the time spent running from insects, leading to reduced foraging time (Weladji et al., 2003). For example, Couturier et al. (2009) examined variation in calf body size and movement rates for the Rivere-aux-Feuilles herd and the George River herd in Quebec and Labrador, and found that daily movement rates in summer were related to reduced birth weight and lower productivity in fall. As a result, caribou may prefer windy and sparsely vegetated areas, such as esker tops, to avoid insect harassment, thereby increasing their time for optimal foraging conditions (Murphy & Curatolo, 1987; Gunn et al., 2002). These conditions are consistent with those surrounding Contwoyto Lake, which experienced the greatest increase in caribou use in this study. In addition, the Boots on the Ground (Ekwǫ̀ Nàxoèhdee K'è) caribou monitoring initiative by the Tłı̨chǫ government has also noticed high utilization around Contwoyto Lake due to its role as an insect refuge area (Tłı̨chǫ Research & Training Program, 2020).

Trends in phenology and productivity metrics were less prominent, relative to elevation and trends in summer temperatures, in both models. The relationship between snow melt timing and caribou use appears to follow a nonlinear pattern, where decreased caribou use trends were associated with areas of delayed snow melt and increased caribou use trends were associated with regions of earlier snow melt timing. Caribou have been observed to adjust their migrations and foraging habits based on local snow conditions (Gordon, 2005; Mallory et al., 2020). Snow conditions control the time of green-up, which can have important impacts on nutrition and reproductive strategies (Chen et al., 2018; Post & Forchhammer, 2008). Specifically, during calving and early summer, access to high quality vegetation is extremely important and caribou typically seek areas where snow has melted and vegetation is available for consumption (Thorpe et al., 2001). The results of this research indicate that regions around Contwoyto Lake, which experienced an average advancement in snow melt timing of 1.13 days, potentially offered better access to forage, driving higher caribou use during early summer. Relative declines in caribou use in regions with delayed snowmelt suggests that they avoided these areas, as lower population numbers allow them to select more optimal foraging habitats elsewhere. These findings underscore the direct link between snow melt timing and caribou habitat utilization, highlighting the critical role of early snow melt at potentially supporting better foraging conditions and ultimately influencing caribou distribution.

The start of season is also highly influenced by trends in temperature, as well as the timing of snow melt, and therefore, I would expect to see a similar relationship as the timing of snow melt for the start of season. As expected, the partial dependence plots indicated that advancements in SOS were associated with increased caribou use, while delays in SOS resulted in marginal decreases in use. Caribou productivity is highly dependent on access to nutritious forage early in the season, as vegetation quality declines rapidly after SOS (Bergerud, 2000; Post et al., 2003; Post & Forchhammer, 2008; Tveraa et al., 2013). While trophic mismatch is a common concern for migratory species, the positive relationship between trends in SOS and relative caribou use indicate that caribou have adjusted their migrations based on cues from the environment. A similar result indicating no trophic mis-match has also been witnessed in other barren-ground caribou herds in Nunavut (Qamanirjuaq; Mallory et al. 2020) and Alaska (Central Arctic; Gustine et al., 2017), as well as reindeer in Norway and Svalbard (Tverra et al., 2013; Veiberg et al., 2017). As capital breeders, caribou rely on resources gained throughout the summer and fall to support reproduction (Barboza & Parker, 2009; Langvatn & Albon, 1986; Taillon et al., 2013). An early SOS, along with an earlier spring migration, can therefore have positive effects on cow and calf nutrition and survival (Cebrian et al., 2008; Couturier et al., 2009; Tveraa et al., 2013). For example, Paoli et al. (2020) found that in years with an earlier SOS, Finnish reindeer calves were born with higher body weights and had higher survival rates during summer. Therefore, without evidence of trophic mismatch, caribou can benefit from earlier SOS, which provides earlier and longer access to forage resources.

In contrast to SOS, EOS did not demonstrate a strong influence on caribou use in my study, especially in the first random forest model which included both dynamic and static variables. In tundra environments, the extension of the growing season through delayed EOS could theoretically extend the period for accessing late-season forage and help caribou accumulate body reserves before winter. However, many studies have concluded that longer growing seasons are primarily driven by earlier SOS rather than a later EOS (Dearborn & Danby, 2021; Goetz et al., 2005; Post et al., 2009). This is corroborated by my findings, as the relatively weak influence of EOS in comparison to SOS suggests that caribou prioritize early-season forage availability, which is more closely tied to the SOS. This finding could reflect that early summer, rather than late summer, is the critical period for accessing the high-quality forage needed for successful migration and reproduction.

Vegetation productivity has increased across much of the Bathurst herd's summer range, which is generally expected to benefit caribou by enhancing maternal body mass, parturition rates and calf growth (Cameron & Ver Hoef, 1994; Sharma et al., 2009; Taillon et al., 2013). However, despite increased plant biomass and a longer growing season, this has not corresponded with increased caribou populations (Fauchald et al., 2017), suggesting that higher productivity may not be beneficial to caribou foraging patterns. My random forest results further corroborate this, where an increase in plant productivity (given my maximum EVI) was associated with a significant decrease in relative caribou use, revealing a threshold where rising productivity corresponds with a decrease in habitat utilization.

A possible explanation for this is the increase of erect deciduous shrub cover in tundra regions, which is often associated with enhanced plant productivity (Bonta et al., 2023) but leads to a reduction in herbaceous vegetation and lichen, which are critical forage species for caribou (Boertje, 1984; Fauchald et al., 2017). These shrubs have higher concentrations of chemical defenses and lower available protein compared to grasses, forbs and sedges, resulting in a decline in forage quality (Thompson & Barboza, 2014). Insufficient high-quality forage during this time can lead to reductions in critical life-history traits, increased winter mortality and reduced pregnancy rates in the fall (Crete & Huot, 1993; Gerhart et al., 1997). This suggests that increased productivity is not necessarily beneficial if it is associated with poorer forage quality, as seen in the spread of shrub-dominated landscapes. However, changes in forage quality are expected to vary regionally (Hansen et al., 2006; Turunen et al., 2009), and the ability of caribou to mitigate these impacts will be dependent on regional variations and other environmental influences (Cebrian et al., 2008). While productivity was not as important as other variables in the regression models, the nature of the relationship is clear: areas with increased productivity, likely dominated by less preferred vegetation types, experienced decreased caribou use.

There are likely other factors influencing changes in relative caribou use by the Bathurst herd during its population decline that were not included in my analysis. Both scientific research and Indigenous Knowledge suggest that caribou tend to avoid habitats near industrial and developed areas, such as mines, roads and pipelines, with avoidance documented within 1 to 14km of such infrastructure (Bergerud et al., 2008; Festa-Bianchet et al., 2011; Kendrick et al., 2005; Boulanger et al., 2012; Boulanger et al., 2021; Cameron et al., 2005; Polfus et al., 2011). In addition, wolf dynamics may also influence range shifts. Wolves, the primary predator of caribou, tend to follow caribou distribution patterns but prefer to den south of the tree line, closer to structural support of roots and shrubs (Heard & Williams, 1992; Walton et al., 2001). As caribou shift further north, they may be spending more time away from wolf dens, reducing predation risk (Heard & Williams, 1992; Klaczek, 2015). While industrial development and predator dynamics are well-documented influences on caribou habitat use, these factors were not modeled in my study, and therefore I cannot definitively assess their impact on the observed changes in caribou range use, though they are likely to play a contributing role and should be included in future studies assessing the herd’s changes in habitat use.

As a result of climate change, Arctic temperatures are warming, leading to indirect implications related to the phenology of weather (Derksen et al., 2019) and vegetation (Dearborn & Danby, 2021; Goetz et al., 2005; Post et al., 2009), as well as plant productivity (Dearborn & Danby, 2021). Many barren-ground caribou herds worldwide live in Arctic regions and are likely being affected by cumulative impacts related to climate change (Festa-Bianchet et al., 2011; Vors & Boyce, 2009). The results of this study show significant range contractions and northward shifts by the Bathurst herd from 1997 to 2017. Modelling relative habitat use with several environmental variables revealed that warming temperatures above 0.10 degrees Celsius, delayed SOS, delayed snow melt and increased vegetation productivity were all associated with decreased relative habitat use by the Bathurst herd, while elevations above 500 M.A.S.L, advanced snow melt time, and advanced SOS were associated with increased habitat use. The significant contraction indicates that the decline of the Bathurst herd itself is likely associated with changes in distribution, including density-dependent resource utilization, such as escaping predators and insect harassment. The results also highlight the importance of habitat in the regions surrounding Contwoyto Lake, where the largest increases in caribou use have occurred. Overall, the findings emphasize the multifaceted nature of environmental influences on caribou habitat utilization and confirm the notion that changes in range use by the herd are likely influenced by cumulative factors. As climate change continues to alter Arctic and sub-Arctic environments, understanding these complex interactions will be vital for the conservation and management of the Bathurst caribou herd.

Chapter 4: Conclusion

4.1 Summary

The significant decline of the Bathurst caribou herd and its geographical range, likely driven by cumulative impacts related to climate change and increased land-use changes, is a critical concern. Understanding various aspects that are potentially related to the herd's decline, as well as the strategies used to adapt to changes within their range, is essential for developing effective mitigation and conservation strategies. This study utilized 21 years of telemetry, remote sensing and climatic data to assess trends in the herd's summer range distribution and elucidate the potential drivers during a period of decline. The study successfully addressed the following objectives:

Objective One: Analyze collar telemetry data from 1997 to 2017 to determine if and how the summer range distribution and extent of the herd have changed during its recent decline.

The results of the kernel density analysis and Theil-Sen regression indicate significant changes in the herd's summer range distribution and extent over the study period. Notably, there was a pronounced contraction of the herd's range between 1997 and 2017, with the most substantial decreases in area occurring after 2009, coinciding with a stark population decline from 2006 to 2009 (GNWT, 2019). This contraction was centered around Contwoyto Lake, located at the NWT and Nunavut border, and was accompanied by a decreased utilization of south-eastern regions. Contwoyto Lake is situated at the northern edge of the herd's summer range, and is characterized by higher elevations and consistently higher wind speeds. This region may contain crucial habitat conditions and resources for the herd, making it an area of interest for scientists and Indigenous communities intertwined with the herd.

Objective Two: Use remote sensing analyses and climatic data to ascertain if there are relationships between trends in caribou use and environmental changes.

The random forest regression models revealed that trends in relative caribou habitat use correspond to trends in climate, topographic and vegetation indices. Both models, containing static and dynamic variables, showed consistent trends, indicating little variation when elevation was excluded from the dynamic model. Elevation, changes in maximum summer temperature, and changes in snow melt timing emerged as the most influential factors to explain the trends in relative caribou utilization. Vegetation phenology (SOS, EOS) and productivity (maximum EVI) metrics were less influential, but still exhibited significant relationships with changes to relative summer range distribution. This may be because trends in summer temperature and snow melt timing influence vegetation indices and therefore, may be seen as the overall driver of changes to both vegetation, as well as trends in relative caribou use.

4.2 Comments on Methodologies

This study highlights the potential of KDE and Theil-Sen analyses to assess shifts in species distribution overtime. KDE is regarded as the most statistically accurate nonparametric method for estimating density and is supported by vast amounts of statistical literature (Izenman, 1991; Silverman, 1986). The continuous nature of KDE allowed for herd based estimates of summer range distribution, rather than individual based monitoring for a small subset of animals belonging to the herd. This is a common strategy of ecologists to characterize and visual species' home ranges (Laver & Kelly, 2008) and is used in a number of ecological studies to identify species distribution (e.g., Johansson et al., 2018). Additionally, this method allowed for the identification of hotspots, as well as home ranges that will be beneficial for effective management of the herd.

The Theil-Sen regression analysis allowed for the identification of changes in annual summer range use by the Bathurst herd. There are a small number of non-parametric regression techniques in the literature and the Theil-Sen regression method is the most widely acknowledged (Chervenkov & Slavov, 2019). This method is typically chosen over ordinary least squares (OLS) regression techniques as it provides a better line fitting technique when there are outliers present, and it does not require the data to fit a normal distribution (Fernandes & Leblanc, 2005). The results allow for the identification of pixels where relative caribou use increased, decreased, or not changed at all, as well as providing a slope value indicating the magnitude of change. To my knowledge, there is no existing literature that uses these two methods together to examine changes in barren-ground caribou seasonal range use, adding additional knowledge to the literature.

Furthermore, the random forest modelling utilized allows for the examination of many different climatic and environmental variables that are possibly influencing changes in summer range distribution. This type of model has many advantages, including that they are non-parametric in nature, can use many different forms of data, limits overfitting and are capable of handling high dimensional data (Evans et al., 2011). For these reasons, random forests are becoming more popular in ecological studies (Chen et al., 2021; Cutler et al., 2007; Evans & Cushman, 2009). However, researchers using random forest modelling to assess changes in distributions should be cautious when interpreting partial dependence plots. The plots shown in this study represent an average of all pixels included in the model, with a large portion of no change pixels. This makes it difficult to interpret how environmental variables are related to sites of increasing or decreasing use.

4.3 Future Research Directions

Future research on the Bathurst caribou herd should build upon this study's fundings by extending modeling efforts to other seasonal ranges. Specifically, investigating the herd's habitat use during calving, post-calving, and winter seasons will help determine whether the significant environmental and climatic factors identified here – such as maximum summer temperature and snow melt timing – are consistent across the different ranges. This broader modeling approach will provide a more comprehensive understanding of the herd's habitat characteristics and spatial dynamics, and it could also uncover seasonal variations in habitat preferences and environmental stressors.

Moreover, future studies should incorporate additional environmental and climatic variables that may influence caribou distribution and movement patterns. Variables such as vegetation composition, proximity to infrastructure (e.g., roads, mines and ports), and predator distribution are critical to understanding the multifaceted drivers of habitat use. For example, analyzing how vegetation quality and quantity change over time, and how these changes affect caribou foraging behavior will shed light on the nutritional landscape of the range. Similarly, examining the impact of industrial development and associated disturbances will help delineate areas where caribou are most vulnerable to human activity, offering essential data for land-use planning and mitigation strategies.

Additionally, integrating Indigenous Knowledge into research initiatives will be especially valuable. Indigenous communities have observed environmental changes and caribou behavior over generations, offering a longitudinal perspective that complements scientific findings. Collaborations with these communities can highlight cultural and ecologically significant areas, identify critical protection zones, and inform conservation strategies. Protection measures informed by both scientific research and Indigenous Knowledge might include restricting infrastructure development in high-use regions, enforcing stricter harvesting regulations and monitoring until population levels become sustainable, and implementing adaptive management strategies. These long-term strategies are essential for tracking the herd's range use, monitoring habitat conditions, and ensuring that conservation efforts are responsive to ongoing environmental and climatic changes. By taking a holistic and adaptive approach, future initiatives will be better equipped to mitigate the herd's population decline and prevent potential extinction.

4.3 Link to Sustainability

Sustainability is a central tenet of critical environmental studies and therefore, I feel it is necessary to highlight how this study and the Bathurst caribou herd itself are crucial to the environmental, social and economic sustainability of northern ecosystems. Terrestrial ecosystems are heavily intertwined across all trophic levels and have numerous feedback mechanisms that influence their overall health. This means that the decline or absence of the Bathurst herd on the landscape influences not only the specific species, but all facets of the ecosystem from vegetation they feed on to the predators that prey upon them.

This study underscores the importance of ongoing environmental and telemetry data collection to monitor changes within northern ecosystems. Without such data, it would be nearly impossible to understand the herd's shifting range use or the trends in environmental factors related to these changes. A comprehensive understanding of these dynamics, informed by both scientific research and Indigenous Knowledge, is essential. By combining these approaches, conservation efforts can be tailored to meet the unique challenges of the region. For example, Indigenous Knowledge provides critical insights into caribou behavior, migration patterns, and habitat needs, which have been observed and documented over millennia. These insights, when integrated with scientific findings, can inform sustainable management practices that not only conserve the Bathurst caribou herd but also uphold traditional Indigenous ways of life, which are closely tied to the herd's presence and health.

The widespread distribution of the Bathurst herd makes them integral to the functioning of northern ecosystems. Through trampling and defecation, the herd contributes to nutrient cycling of nutrient-limited soils and suppresses vegetation growth (Sharma et al., 2009; Bernes et al., 2015). Caribou also provide sustenance to predators and scavengers, including wolves, grizzly bears and wolverines (Frame et al., 2008). The social implications of the herd's decline are significant, as northern Indigenous communities, such as the Dene and Inuit, have relied on and cared for caribou for thousands of years. The cultural significance and socio-economic value of caribou date back approximately 8000 years, and recent decades have seen unprecedented shifts in both population and movement (Kendrick et al., 2005; Gordon, 2005).

Continued long-term monitoring and assessment of changing habitat characteristics are vital for conserving the herd's habitats, which are crucial for the herd's survival and the sustainability of the region. Effective conservation efforts, such as implementing adaptive management strategies or protecting critical migration corridors, will not only help to prevent extinction but also support the resilience of the broader ecosystem. Additionally, these measures ensure the preservation of Indigenous cultural practices and traditional ways of life, which are intertwined with the land and the caribou. By implementing an integrated approach to research and conservation, we can work toward a sustainable future that maintains the ecological integrity of the northern landscape and honors the cultural heritage of Indigenous communities.

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Appendix

Appendix A – Collar Data Quality

Table A.1: The average GPS receiver error for each year of the study period, from 1997 to 2017. During this time there is an increase in location accuracy, and an average accuracy of 285.13m. Data produced by Mennell (2021).

Year	**Locational Error (meters)**
1997	500.00
1998	592.59
1999	611.11
2000	500.00
2001	483.33
2002	550.00
2003	392.86
2004	294.12
2005	272.73
2006	264.71
2007	261.90
2008	203.13
2009	167.31
2010	113.04
2011	120.00
2012	117.39
2013	123.53
2014	120.00
2015	100.00
2016	100.00
2017	100.00
Average	285.13

Appendix B – Environmental Predictor Variables

Table B.1 – Results of the Spearman rank correlation coefficient analyses between each of the potential predictor variables used in the random forest modelling. Positive values indicate positive relationships, while negative values indicate negative relationships. Values greater than 0.70 or less than -0.70 indicate very strong relationships and were removed prior to modelling.

	TS	*Elevation*	*Fire*	*Wind Speed*	*Max Temp*	*Min Temp*	*Snow Melt*	*EVI*	*LOS*	*SOS*	*EOS*
TS	1.00	0.38	-0.03	0.33	0.06	0.11	-0.21	-0.12	0.18	-0.24	0.05
Elevation	0.38	1.00	0.02	**0.71**	0.42	0.44	-0.36	0.01	0.23	-0.28	0.09
Fire	-0.03	0.02	1.00	0.04	-0.03	-0.06	0.05	-0.01	0.04	-0.03	0.04
Wind Speed	0.33	**0.71**	0.04	1.00	0.47	0.35	-0.32	0.01	0.28	-0.30	0.15
Max Temp	0.06	0.42	-0.03	0.47	1.00	**0.74**	-0.41	0.19	-0.09	0.00	-0.12
Min Temp	0.11	0.44	-0.06	0.35	**0.74**	1.00	-0.55	0.08	-0.03	-0.10	-0.12
Snow Melt	-0.21	-0.36	0.05	-0.32	-0.41	-0.55	1.00	0.01	-0.12	0.21	0.01
EVI	-0.12	0.01	-0.01	0.01	0.19	0.08	0.01	1.00	-0.40	0.29	-0.32
LOS	0.18	0.23	0.04	0.28	-0.09	-0.03	-0.12	-0.40	1.00	**-0.71**	**0.77**
SOS	-0.24	-0.28	-0.03	-0.30	0	-0.10	0.21	0.29	**-0.71**	1.00	-0.16
EOS	0.05	0.09	0.04	0.15	-0.12	-0.12	0.01	-0.32	**0.77**	-0.16	1.00

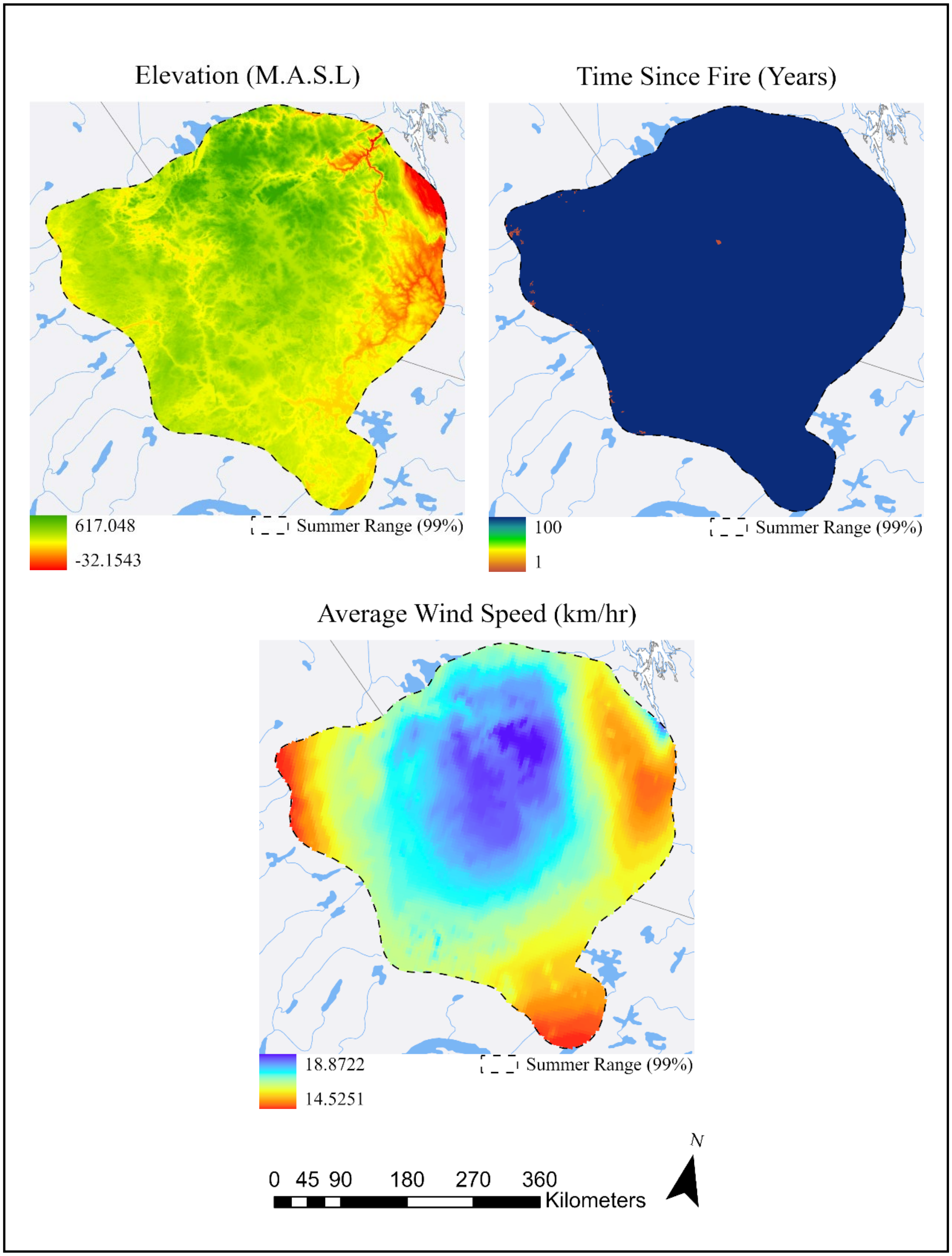

Figure B.1 – The visualization of potential static predictor variables used in the random forest models. Each raster has been clipped to the extent of the 99% summer range extent (MCP).

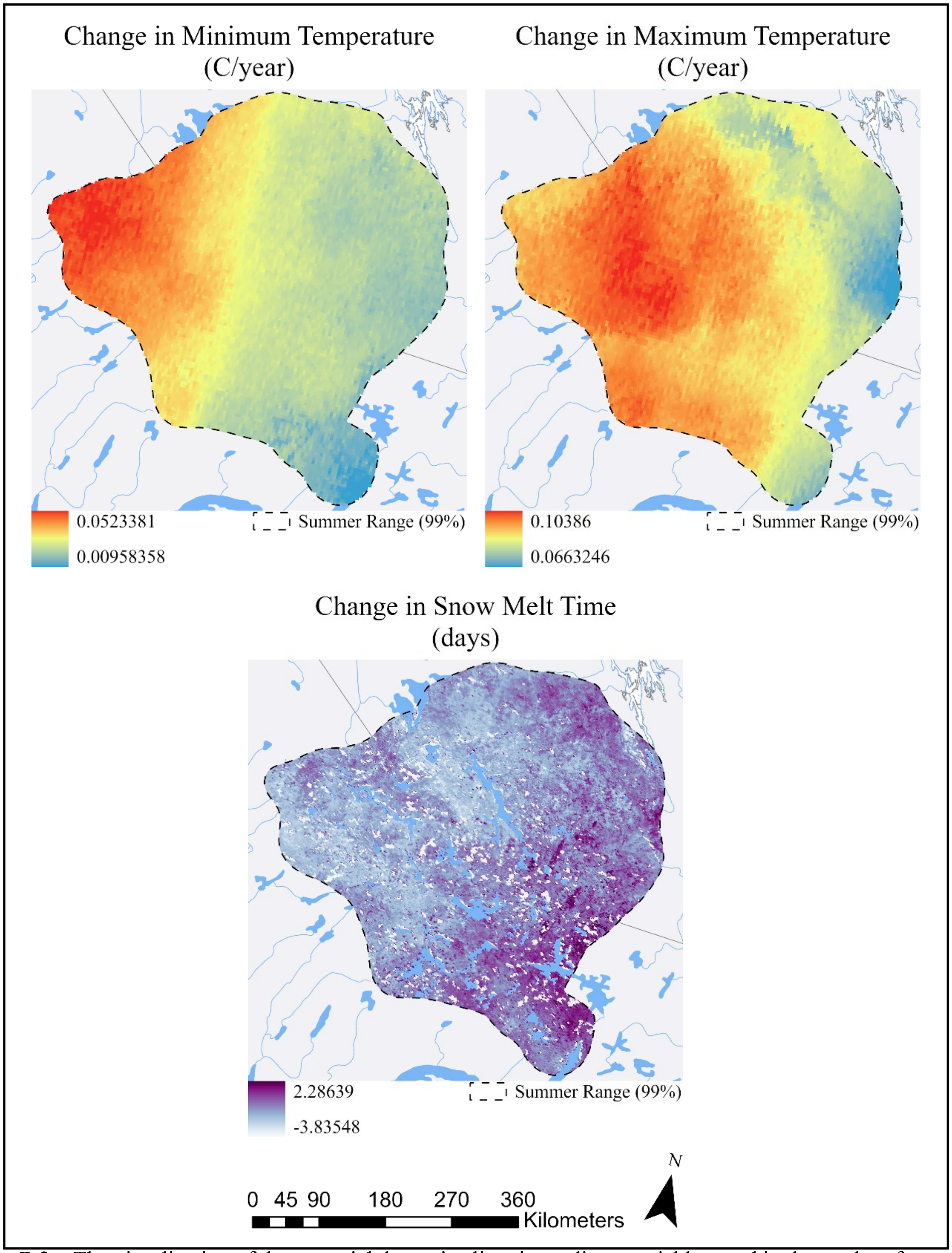

Figure B.2 – The visualization of the potential dynamic climatic predictor variables used in the random forest models. Each raster has been clipped to the extent of the 99% summer range extent (MCP).

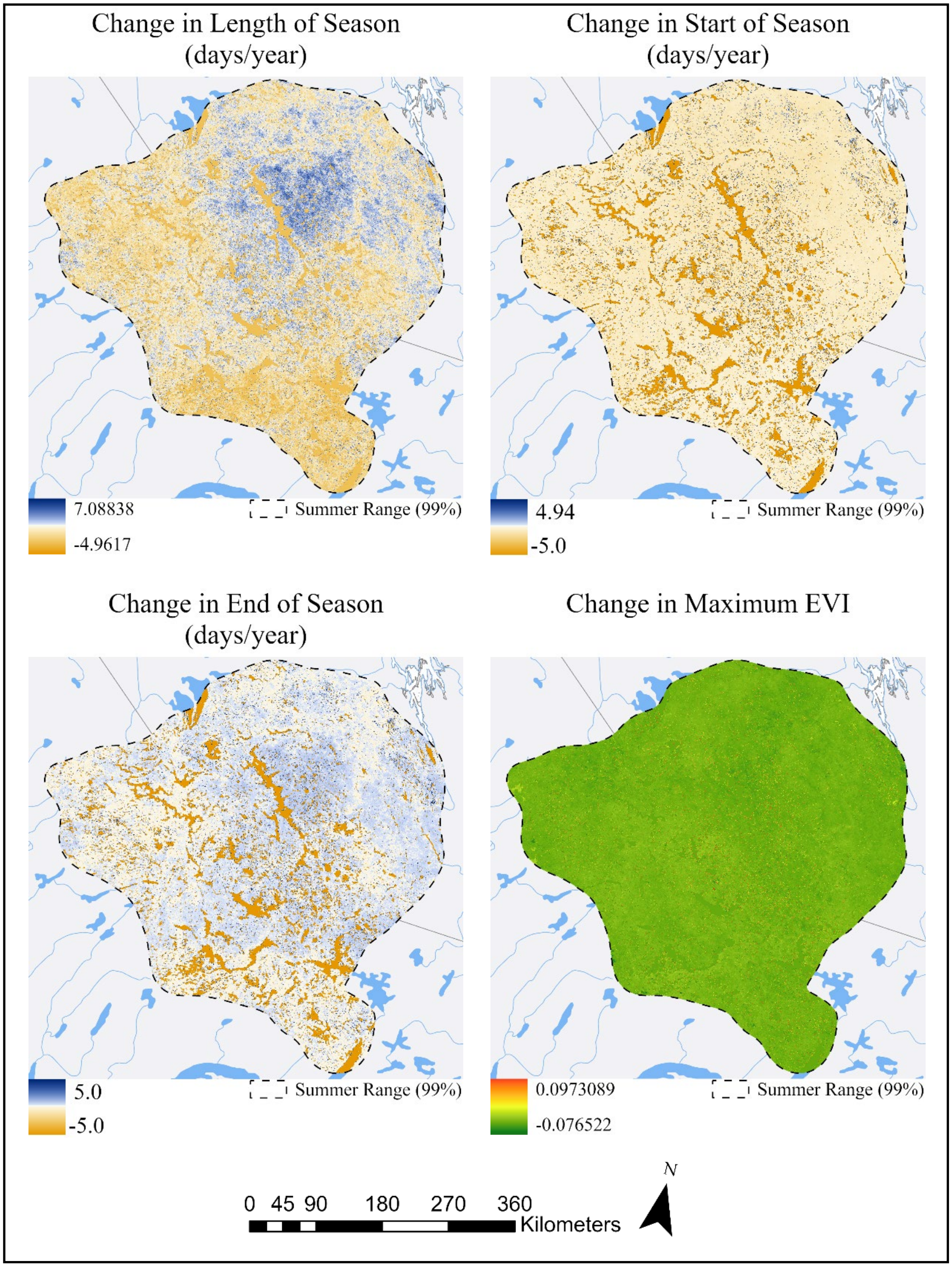

Figure B.3 – The visualization of the potential dynamic predictor variables, related to vegetation phenology and productivity, used in the random forest models. Each raster has been clipped to the extent of the 99% summer range extent (MCP).

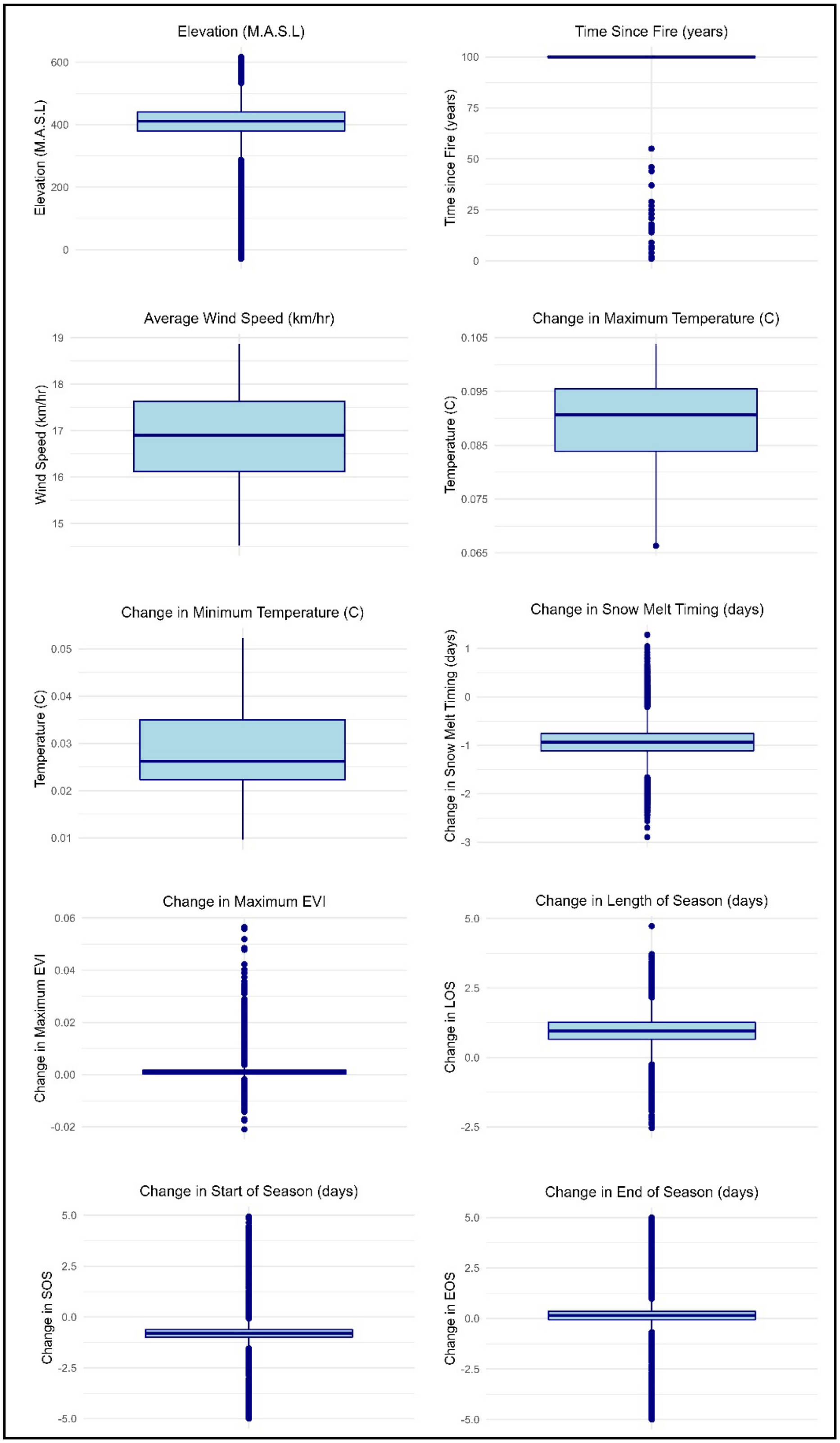

Figure B.4 – The distribution of the potential climatic and environmental variables used in the random forest modelling. Each variable only contains values within the summer range extent (99% MCP).

Appendix C – Ground Truthing Analysis

Table C.1 – Raw data values from the MODIS EVI satellite data and the calculated GNDVI for each of the 31 sites within the Bathurst herd's summer range.

SITE #	GNDVI	EVI
1	0.57	0.3638
2	0.61	0.3555
3	0.6	0.2809
4	0.59	0.3747
5	0.53	0.38295
6	0.54	0.4393
7	0.56	0.4259
8	0.52	0.38465
9	0.55	0.3573
10	0.52	0.3581
11	0.54	0.3317
12	0.52	0.3315
13	0.54	0.3326
14	0.55	0.34565
15	0.54	0.419
16	0.52	0.3525
17	0.52	0.359
18	0.57	0.3371
19	0.55	0.3699
20	0.55	0.3451
21	0.56	0.372
22	0.56	0.3508
23	0.55	0.3608
24	0.51	0.3045
25	0.55	0.3291
26	0.54	0.3541
27	0.54	0.3699
28	0.58	0.3612
29	0.55	0.3591
30	0.5	0.3334
31	0.47	0.3336

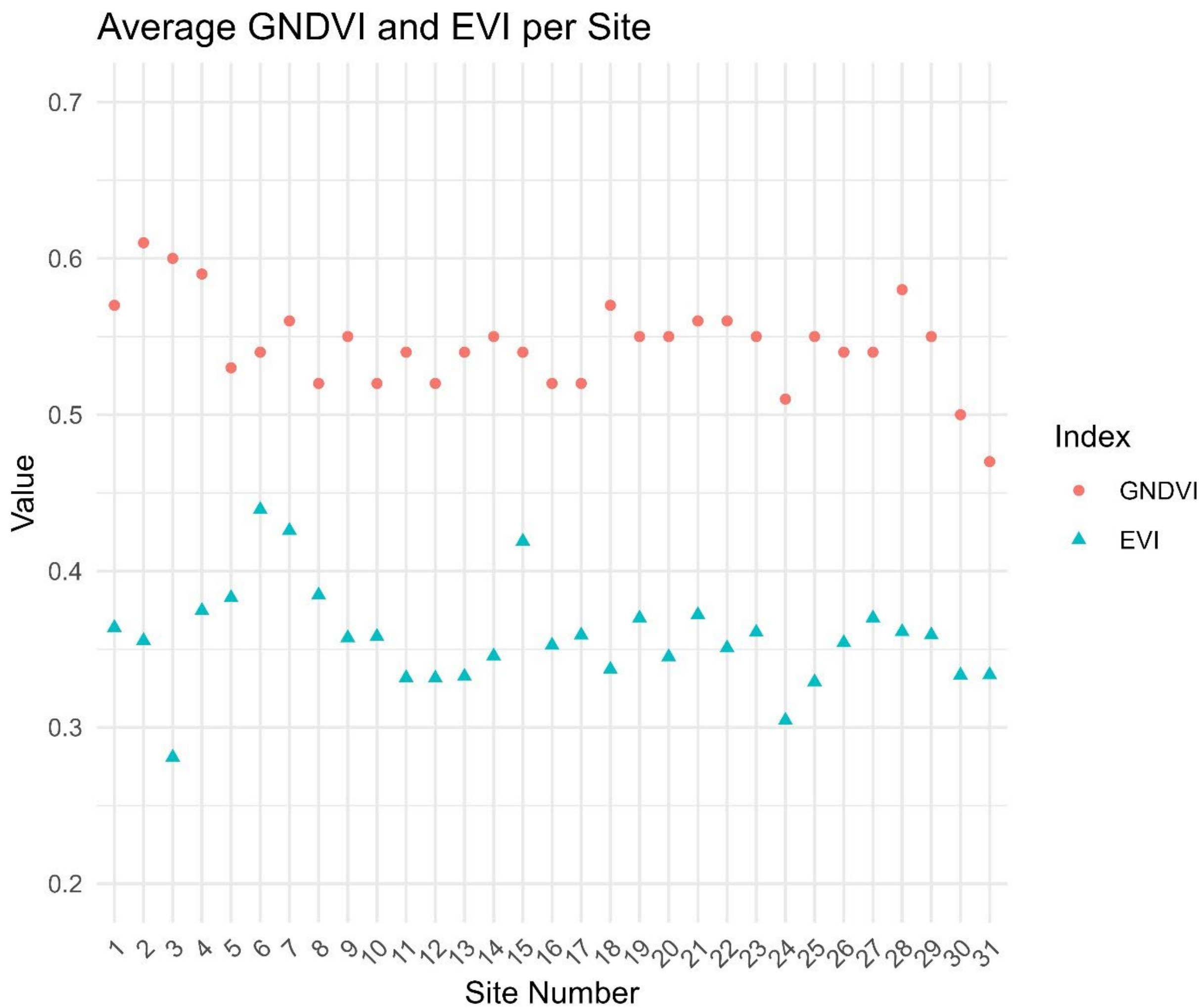

Figure C.2 – Visualization of the trends in the raw data values from the MODIS EVI satellite data and the calculated GNDVI for each of the 31 sites within the Bathurst herd's summer range.

Appendix D – R Code

```
####Installing Packages
library(TuktuTools, dplyr, sf, KernSmooth, adehabitatHR, raster, rgdal,
      zyp, terra, spatialEco, readxl, gridExtra, permimp, readr, randomForest,
      rpart, rfUtilities, rfviz, stats, corrplot, RColorBrewer)
##################################################Dataset Processing
###Import df
caribou <- read.csv("caribou_all.csv")
### Subset df for cows and bulls
caribou_fem <- caribou[caribou$Gender == "F", ]
caribou_male <- caribou[caribou$Gender == "M", ]
###Subset summer months (complete for each year of data)
caribou_fem$location_d <- as.POSIXct(caribou_fem$location_d, format = "%Y-%m-%dT%H:%M:%SZ", tz = "UTC")
start_date97 <- as.POSIXct("1997-06-29", format = "%Y-%m-%d", tz = "UTC")
end_date97 <- as.POSIXct("1997-09-06 23:59:59", format = "%Y-%m-%d %H:%M:%S", tz = "UTC")
Summer_97 <- caribou_fem[caribou_fem$location_d >= start_date97 & caribou_fem$location_d <= end_date97, ]
###Combine all years
Summer_ALL <- rbind(Summer_97, Summer_98, Summer_99, Summer_00, Summer_01, Summer_02, Summer_03,
            Summer_04, Summer_05, Summer_06, Summer_07, Summer_08, Summer_09, Summer_10,
            Summer_11, Summer_12, Summer_13, Summer_14, Summer_15, Summer_16, Summer_17)
###Get daily mean location
dailyMean <- getDailyMean(Summer_ALL, id.col = "animal_ID2", time.col = "location_d")
###Re-name the df
caribou <- dailyMean
###Create annual dfs (complete for each year)
caribou_97 <- caribou[caribou$Year == 1997, ]

##################################################KDE and Theil-Sen Processing
###Calculate annual bandwidths (complete for each year)
bw_97 <- dpik(cbind(caribou_97$X_Axis, caribou_97$Y_axis))
# View the calculated total bandwidth
cat("Total Bandwidth:", bw_97, "\n")
###Calculate MCP
caribousp <- SpatialPoints(coords = caribou[, c("X_Axis", "Y_Axis")])
kde <- kernelUD(caribousp, h = bandwidth, kern = "bivnorm", same4all = FALSE, grid = xy.sp)
proj4string(kde) <- CRS("+init=EPSG:3580")
con99 <- getverticeshr(kde, 99)
###Calculate average of all bandwidths
bandwidth <- mean(bw_97, bw_98, bw_99, bw_00, bw_01, bw_02, bw_03, bw_04, bw_05, bw_06, bw_07,
      bw_08,  bw_09, bw_10, bw_11, bw_12, bw_13, bw_14, bw_15, bw_16, bw_17)
###Make grid
x <- seq(-325000, 470000, by=500)
y <- seq(8200000, 9100000, by= 500)
xy <- expand.grid(x=x, y=y)
xy.sp <- SpatialPoints(xy)
gridded(xy.sp) <- TRUE
###Calculate KDE (complete for all years)
caribousp_97 <- SpatialPoints(coords = caribou_97[, c("X_Axis", "Y_Axis")])
kde97 <- kernelUD(caribousp_97, h = bandwidth, kern = "bivnorm", same4all = FALSE, grid = xy.sp)
proj4string(kde97) <- CRS("+init=EPSG:3580")
###Transform raw values and extract contours
vol97 <- getvolumeUD(kde97)
con95_97 <- getverticeshr(kde97, 95)
con50_97 <- getverticeshr(kde97, 50)
###Calculate TS
volumefiles <- list(vol97_ras, vol98_ras, vol99_ras, vol00_ras, vol01_ras, vol02_ras,
            vol03_ras, vol04_ras, vol05_ras, vol06_ras, vol07_ras, vol08_ras,
            vol09_ras, vol10_ras, vol11_ras, vol12_ras, vol13_ras, vol14_ras,
            vol15_ras, vol16_ras, vol17_ras)
KDE_Stack <- stack(volumefiles)
KDE_spatrast <- rast(KDE_Stack)
TS <- raster.kendall(KDE_spatrast, p.value = TRUE)
```

```
#################################################Vegetation Surveys
###Comparing MODIS EVI and plot-level GNDVI
##Read Data (contains both EVI and GNDVI for each site)
GroundTruthData <- read_excel("GroundTruthData.xlsx")
##Visualize Data
GNDVI_plot <- ggplot(GroundTruthData, aes(x = factor(Site), y = GNDVI, group = 1)) +
 geom_point() +
 geom_line() +
 labs(x = "Site Number", y = "Average GNDVI", title = "Average GNDVI per Site") +
 theme_minimal() +
 theme(axis.text.x = element_text(angle = 45, hjust = 1)) +
 ylim(0.20, 0.70)
EVI_plot <- ggplot(GroundTruthData, aes(x = factor(Site), y = EVI, group = 1)) +
 geom_point() +
 geom_line() +
 labs(x = "Site Number", y = "Average EVI", title = "Average EVI per Site") +
 theme_minimal() +
 theme(axis.text.x = element_text(angle = 90, hjust = 1)) +
 ylim(0.20, 0.70)
grid.arrange(GNDVI_plot, EVI_plot, ncol = 2)
##Calculate Spearman
cor.test(GroundTruthData2$GNDVI, GroundTruthData2$EVI, method = "spearman")

################################################# RF: Predictor Variable Processing
###Max Temperature Processing (complete for each year)
##Import raster files
MaxTemp97_06 <- raster("199706.tif")
MaxTemp97_07 <- raster("199707.tif")
MaxTemp97_08 <- raster("199708.tif")
MaxTemp97_09 <- raster("199709.tif")
MaxTemp97_10 <- raster("199710.tif")
##Scale the data
MaxTemp97_06Scale <- MaxTemp97_06 * 0.10
MaxTemp97_07Scale <- MaxTemp97_07 * 0.10
MaxTemp97_08Scale <- MaxTemp97_08 * 0.10
MaxTemp97_09Scale <- MaxTemp97_09 * 0.10
MaxTemp97_10Scale <- MaxTemp97_10 * 0.10
##Calculate an annual average
MaxTemp97_STACK <- stack(MaxTemp97_06Scale, MaxTemp97_07Scale, MaxTemp97_08Scale,
              MaxTemp97_09Scale, MaxTemp97_10Scale)
MaxTemp97 <- calc(MaxTemp97_STACK, fun = mean)
##Calculate TS
MaxTemp.list <- list(MaxTemp97, MaxTemp98, MaxTemp99, MaxTemp00, MaxTemp01, MaxTemp02, MaxTemp03,
            MaxTemp04, MaxTemp05, MaxTemp06, MaxTemp07, MaxTemp08, MaxTemp09, MaxTemp10,
            MaxTemp11, MaxTemp12, MaxTemp13, MaxTemp14, MaxTemp15, MaxTemp16, MaxTemp17)
MaxTemp_STACK <- stack(MaxTemp.list)
MaxTemp_spatrast <- rast(MaxTemp_STACK)
MaxTemp_TS <- raster.kendall(MaxTemp_spatrast, p.value = TRUE)
###Min Temperature Processing (complete for each year)
##Import raster files
MinTemp97_06 <- raster("199706.tif")
MinTemp97_07 <- raster("199707.tif")
MinTemp97_08 <- raster("199708.tif")
MinTemp97_09 <- raster("199709.tif")
MinTemp97_10 <- raster("199710.tif")
##Scale the data
MinTemp97_06Scale <- MinTemp97_06 * 0.10
MinTemp97_07Scale <- MinTemp97_07 * 0.10
MinTemp97_08Scale <- MinTemp97_08 * 0.10
MinTemp97_09Scale <- MinTemp97_09 * 0.10
MinTemp97_10Scale <- MinTemp97_10 * 0.10
##Calculate an annual average
MinTemp97_STACK <- stack(MinTemp97_06Scale, MinTemp97_07Scale, MinTemp97_08Scale,
              MinTemp97_09Scale, MinTemp97_10Scale)
MinTemp97 <- calc(MinTemp97_STACK, fun = mean)
##Calculate TS
```

```
MinTemp.list <- list(MinTemp97, MinTemp98, MinTemp99, MinTemp00, MinTemp01, MinTemp02, MinTemp03,
              MinTemp04, MinTemp05, MinTemp06, MinTemp07, MinTemp08, MinTemp09, MinTemp10,
              MinTemp11, MinTemp12, MinTemp13, MinTemp14, MinTemp15, MinTemp16, MinTemp17)
MinTemp_STACK <- stack(MinTemp.list)
MinTemp_spatrast <- rast(MinTemp_STACK)
MinTemp_TS <- raster.kendall(MinTemp_spatrast, p.value = TRUE)
###Wind Speed Processing (complete for each year)
##Import raster files
Wind97_06 <- raster("199706.tif")
Wind97_07 <- raster("199707.tif")
Wind97_08 <- raster("199708.tif")
Wind97_09 <- raster("199709.tif")
Wind97_10 <- raster("199710.tif")
##Scale the data
Wind97_06Scale <- Wind97_06 * 0.01
Wind97_07Scale <- Wind97_07 * 0.01
Wind97_08Scale <- Wind97_08 * 0.01
Wind97_09Scale <- Wind97_09 * 0.01
Wind97_10Scale <- Wind97_10 * 0.01
##Transform to km/hr
Wind97_06KM <- Wind97_06Scale * 3.6
Wind97_07KM <- Wind97_07Scale * 3.6
Wind97_08KM <- Wind97_08Scale * 3.6
Wind97_09KM <- Wind97_09Scale * 3.6
Wind97_10KM <- Wind97_10Scale * 3.6
##Calculate an annual average
Wind97_STACK <- stack(Wind97_06KM, Wind97_07KM, Wind97_08KM, Wind97_09KM, Wind97_10KM)
Wind97 <- calc(Wind97_STACK, fun = mean)
##Calculate an average for the study period
Wind.list <- list(Wind97, Wind98, Wind99, Wind00, Wind01, Wind02, Wind03,
            Wind04, Wind05, Wind06, Wind07, Wind08, Wind09, Wind10,
            Wind11, Wind12, Wind13, Wind14, Wind15, Wind16, Wind17)
Wind_AVG <- calc(Wind_STACK, fun = mean)
```

Random Forest Modelling

```
###Random Sampling
AllCells <- st_read("AllCells.shp") #points represent all cells within the MCP
set.seed(3)
sampled_indices <- sample(nrow(AllCells_filter), 56640)
sampled_cells <- AllCells_filter[sampled_indices, ]
sampled_cells <- st_drop_geometry(sampled_cells)

###Calculating Spearman Rank
Spearman_matrix <- cor(sampled_cells, method = "spearman")
corrplot(Spearman_matrix, method = "square", type = "upper", order = "hclust",
      col=brewer.pal(n=10, name = "RdBu"),
      tl.col = "black", tl.srt = 45)
###Static Model
RFREG_static <- sampled_cells
RFREG_static <- RFREG_static[, -which(names(RFREG_static) == "WindSpeed")]
RFREG_static <- RFREG_static[, -which(names(RFREG_static) == "MinTemp")]
RFREG_static <- RFREG_static[, -which(names(RFREG_static) == "Fire")]
RFREG_static <- RFREG_static[, -which(names(RFREG_static) == "LOS")]
set.seed(3)
RFREG_static.test <- RFREG_static %>%
 sample_frac(0.3)
RFREG_static.test
RFREG_static.train <- RFREG_static %>%
 anti_join(RFREG_static.test)
RFREG_static.train
RFREG_static.train <- as.data.frame(RFREG_static.train)
RFREG_static.test <- as.data.frame(RFREG_static.test)
set.seed(3)
RFREG_static_mtry <- tuneRF(x = RFREG_static.train[,-1],
                  y = RFREG_static.train$CaribouUse,
                  stepFactor = 0.5, improve = 0.1, trace=T, plot= T)
```

```
set.seed(3)
RFREG_static_mod <- randomForest(CaribouUse ~ ., data = RFREG_static.train,
                        ntree = 300, mtry = 4,
                        importance = TRUE, keep.forest = TRUE, keep.inbag = TRUE)
set.seed(3)
RF_static_val <- predict(RFREG_static_mod, newdata = RFREG_static.test[,-1])
actual_variance_static <- var(RFREG_static.test$CaribouUse)
predicted_variance_static <- var(RF_static_val)
var_explain_static <- (1 - (predicted_variance_static / actual_variance_static))*100
print(paste("Percent Variance Explained:", var_explain_static))
RFREG_static_cpi <- permimp(RFREG_static_mod, conditional = TRUE,
                     threshold = 0.8, scaled = TRUE)
par(mfrow = c(2, 3))
partialPlot(RFREG_static_mod, RFREG_static.train, MaxTemp, plot = TRUE,
        main = "Max Summer Temperature (°C)", ylim = c(-1, 1))
partialPlot(RFREG_static_mod, RFREG_static.train, Elevation, plot = TRUE,
        main = "Elevation (M.A.S.L)", ylim = c(-1, 1))
partialPlot(RFREG_static_mod, RFREG_static.train, SnowMelt, plot = TRUE,
        main = "Snow Melt Timing (days)", ylim = c(-1, 1))
partialPlot(RFREG_static_mod, RFREG_static.train, SOS, plot = TRUE,
        main = "Start of Season (days)", ylim = c(-1, 1))
partialPlot(RFREG_static_mod, RFREG_static.train, EOS, plot = TRUE,
        main = "End of Season (days)", ylim = c(-1, 1))
partialPlot(RFREG_static_mod, RFREG_static.train, EVI, plot = TRUE,
        main = "Enhanced Vegetation Index", ylim = c(-1, 1))
###Dynamic Plot
RFREG_dynamic <- sampled_cells
set.seed(3)
RFREG_dynamic.test <- RFREG_dynamic %>%
 sample_frac(0.3)
RFREG_dynamic.test
RFREG_dynamic.train <- RFREG_dynamic %>%
 anti_join(RFREG_dynamic.test)
RFREG_dynamic.train
RFREG_dynamic.train <- as.data.frame(RFREG_dynamic.train)
RFREG_dynamic.test <- as.data.frame(RFREG_dynamic.test)
set.seed(3)
RFREG_dynamic_mtry <- tuneRF(x = RFREG_dynamic.train[,-1], y = RFREG_dynamic.train$CaribouUse,
                     stepFactor = 0.5, improve = 0.1, trace=T, plot= T)
set.seed(3)
RFREG_dynamic_mod <- randomForest(CaribouUse ~ ., data = RFREG_dynamic.train,
                        ntree = 300, mtry = 2,
                        importance = TRUE, keep.forest = TRUE, keep.inbag = TRUE)
print(RFREG_dynamic_mod)
plot(RFREG_dynamic_mod)
set.seed(3)
RF_dynamic_val <- predict(RFREG_dynamic_mod, newdata = RFREG_dynamic.test[,-1])
actual_variance_dynamic <- var(RFREG_dynamic.test$CaribouUse)
predicted_variance_dynamic <- var(RF_dynamic_val)
var_explain_dynamic <- (1 - (predicted_variance_dynamic / actual_variance_dynamic))*100
print(paste("Percent Variance Explained:", var_explain_dynamic))
RFREG_dynamic_cpi <- permimp(RFREG_dynamic_mod, conditional = TRUE, threshold = 0.8,
                     scaled = TRUE) #calculating cpi
par(mfrow = c(2, 3))
partialPlot(RFREG_dynamic_mod, RFREG_dynamic.train, MaxTemp, plot = TRUE,
        main = "Max Summer Temperature (°C)", ylim = c(-1, 1))
partialPlot(RFREG_dynamic_mod, RFREG_dynamic.train, SnowMelt, plot = TRUE,
        main = "Snow Melt Timing (days)", ylim = c(-1, 1))
partialPlot(RFREG_dynamic_mod, RFREG_dynamic.train, EOS, plot = TRUE,
        main = "End of Season (days)", ylim = c(-1, 1))
partialPlot(RFREG_dynamic_mod, RFREG_dynamic.train, SOS, plot = TRUE,
        main = "Start of Season (days)", ylim = c(-1, 1))
partialPlot(RFREG_dynamic_mod, RFREG_dynamic.train, EVI, plot = TRUE,
        main = "Enhanced Vegetation Index", ylim = c(-1, 1))
```