University of Alberta

WOODLAND CARIBOU CONSERVATION IN ALBERTA: RANGE DELINEATION AND RESOURCE SELECTION

by

Simon Christopher Slater

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Abstract

Woodland caribou (*Rangifer tarandus caribou*) populations are threatened across Canada. Recovery plans are being implemented to address conservation priorities using the best available knowledge. I used animal location data to evaluate sampling requirements for estimating caribou population ranges in Alberta, and developed resource selection functions to assess caribou response to landscape change for one population over a 13-year period. Both the number of caribou and years sampled influenced population level inferences regarding range size. Data were insufficient to generate stable range estimates for several populations in Alberta. Caribou from the Redrock Prairie Creek population exhibited variable annual winter resource selection, but showed overall avoidance of both natural and anthropogenic disturbances. A shift from the historical core range to a less disturbed winter range occurred over the 13-year sampling period, in conjunction with increased anthropogenic disturbance. I provide guidelines for appropriate use of caribou location data for conservation and management planning in Alberta.

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Chapter 1: Introduction

Development of effective conservation initiatives for rare or threatened species requires detailed knowledge of species ecology. Such information is needed to address current issues related to woodland caribou (*Rangifer tarandus caribou*) conservation in Alberta, Canada. The federal Species at Risk Act (SARA) and the Alberta Wildlife Act currently list woodland caribou as a threatened species in Canada and Alberta, respectively. Habitat loss and fragmentation, predation and climate change, have all been recognized as threats to population persistence (Thomas and Gray 2002, Environment Canada 2008, Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA) 2010).

Woodland caribou distribution is circumpolar. In Canada, woodland caribou occur across most of the boreal forest region and are found in all jurisdictions except Nova Scotia, New Brunswick, Prince Edward Island and Nunavut. There is wide variation in their habitat use, behaviour patterns and local ecological conditions, such that woodland caribou have been classified into twelve Designatable Units (DU), including boreal, forest tundra, southern mountain, central mountain and northern mountain (COSEWIC 2011). Both boreal (DU6) and central mountain (DU8, hereafter called mountain) caribou reside within Alberta. Sixteen populations remain in Alberta - twelve boreal and four mountain populations; a fifth mountain population (Banff National Park) was extirpated in 2009 (ASRD and ACA 2010, Hebblewhite et al. 2010). A recent woodland caribou status report noted that, of the 13 populations for which there are sufficient data, 10 are in population decline and these comprise approximately 70% of the caribou occurring in Alberta (ASRD and ACA 2010).

Boreal caribou populations inhabit peatland complexes interspersed with upland pine forest in northern Alberta. Boreal caribou live year-round in forested habitats and occupy distinct seasonal ranges, however because these ranges tend to overlap they are considered non-migratory (Dzus 2001). Mountain caribou populations exhibit seasonal migratory movements between low elevation forested foothills in the winter to alpine habitat in the Rocky Mountains during summer (Smith et al. 2000). Mountain caribou occur in west central Alberta, and their summer range extends into British Columbia. Caribou from both DUs are typically found at low densities and depend on landscapes where their primary winter food source, lichen, is abundant (COSEWIC 2011). In addition, caribou are known to be sensitive to habitat alteration caused by both natural (wildfire) and anthropogenic disturbances (forest harvesting, oil and gas development; Dyer et al. 2001, Johnson et al. 2004, Schaefer and Mahoney 2007, Vors et al. 2007).

Currently, significant resources are being deployed across the country to determine the status of woodland caribou populations and to identify habitats critical for their long-term survival (O'Brien et al. 2006, Racey and Arsenault 2007, Environment Canada 2011). Essential to this understanding, and often the first step in developing conservation strategies, is delineating the spatiotemporal extent of a species range to identify discrete boundaries for conservation units (Bethke et al. 1996, Johnson et al. 2004). Recent advancements in radio telemetry have led to the widespread use of global positioning system (GPS) collars, which allow researchers to gather detailed data on species distribution, behaviour, movement and resource selection (Cooke 2008). Simultaneously, research methodologies and analytical techniques have evolved to account for the increase in quantity and quality of animal location data. Recent studies have focused on issues surrounding the use of location data in wildlife research, such as defining sampling allocation (Leban et al. 2001, Girard et al. 2006) and addressing issues with spatial and temporal resolution (Boyce 2006, Meyer and Thuiller 2006). It is important to recognize that methodological errors, false assumptions and misguided analytical techniques can greatly reduce the validity of interpretation and misinform intended applications.

I studied how inferences from common analytical techniques can be confounded by how animal location data are acquired and analysed. A relevant example that applies to most telemetry studies is the trade-off between the number of locations collected and number of animals sampled. This decision is common in studies using GPS location data, where generally the number of locations per animal is extensive compared to the number of individuals collared. In contrast, studies using very high frequency (VHF) location data typically collect fewer locations, for a larger number of animals. With few animals sampled intensively over time, it is possible to obtain valuable information on the sampled individuals, however generalizations from local population data may not be representative of larger populations. I provide an assessment of how sampling regime impacts local population range delineation and specify appropriate sample sizes required to make population generalizations.

Resource selection function (RSF) models have become widespread in wildlife literature as a method for quantifying habitat requirements of species by comparing used landscape variables to those available (Manly et al. 2002). RSFs allow development of spatially explicit models to describe animal occurrence, however they are generally static and only provide a snapshot of resource selection and potential population distribution in time (Carroll et al. 2003). This is problematic, since the specific goal of species' recovery plans is to achieve self-sustaining natural populations (Environment Canada 2011). Therefore, identifying resource selection patterns, and understanding how selection of these habitats changes over time with availability, is crucial to conservation planning (McLoughlin et al. 2010, Moreau et al. 2012). Previous research has focused on how availability is defined, and examining resource selection on a smaller temporal scale to account for seasonal variation (Arthur et al. 1996, Alldredge and Griswold 2006, Johnson and Gillingham 2008). However, it is also important to understand how resource selection changes in response to natural and anthropogenic disturbances. I incorporate these landscape changes into RSF models to examine resource selection over time.

1. Thesis Overview

My overall objective in this thesis was to examine the issues related to the use of location data in conservation planning and attempt to reconcile methods for range delineation and account for landscape dynamics in resource selection to inform development of standardized protocols that can be applied and compared across local populations and DUs. The data chapters in this thesis are written in manuscript format, therefore there is some overlap in the introductory and methodological sections between Chapters 2 and 3.

In Chapter 2, I investigate how inferences from range delineation vary with differing sampling regimes and estimation methods. I used caribou GPS location data from eight caribou populations in Alberta and subsampled those data to evaluate the impact of sampling frequency and sample size on range estimation. Finally, I compared the population range sizes from two commonly used range estimation methods, the minimum convex polygon (MCP) and kernel density estimator (KDE).

In Chapter 3, I explore how inferences from resource selection functions change in a dynamic landscape, by addressing how resource selection changes due to natural processes and anthropogenic disturbances. I used GPS location data from the Redrock Prairie Creek caribou population during winter and temporally-matched landscape variables to examine resource selection over time by creating annual resource selection functions (RSFs). Finally, I compared range use in the historic core winter range and examined how it changed over the 13-year sampling period.

Chapter 4 highlights my key findings, provides management recommendations and offers direction for future research.

2. Literature Cited

Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA). 2010. Status of the Woodland Caribou (*Rangifer* *tarandus caribou*) in Alberta: Update 2010., Alberta Sustainable Resource Development, Edmonton, AB.

Alldredge, J. R. and J. Griswold. 2006. Design and analysis of resource selection studies for categorical resource variables. Journal of Wildlife Management 70:337-346.

Arthur, S. M., B. F. J. Manly, L. L. McDonald, and G. W. Garner. 1996. Assessing habitat selection when availability changes. Ecology 77:215-227.

Bethke, R., M. Taylor, S. Amstrup, and F. Messier. 1996. Population delineation of polar bears using satellite collar data. Ecological Applications 6:311-317.

Boyce, M. S. 2006. Scale for resource selection functions. Diversity and Distributions 12:269-276.

Carroll, C., R. E. Noss, P. C. Paquet, and N. H. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications 13:1773-1789.

Cooke, S. J. 2008. Biotelemetry and biologging in endangered species research and animal conservation: relevance to regional, national, and IUCN Red List threat assessments. Endangered Species Research 4:165-185.

COSEWIC. 2011. Designatable units for caribou (Rangifer tarandus) in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa. 88 pp.

Dyer, S. J., J. P. O'Neill, S. M. Wasel, and S. Boutin. 2001. Avoidance of industrial development by woodland caribou. Journal of Wildlife Management 65:531-542.

Dzus, E. 2001. Status of the woodland caribou (*Rangifer tarandus caribou*) in Alberta. . Wildlife Status Report. Alberta Environment, Fisheries and Management Division, and Alberta Conservation Association.

Environment Canada. 2008. Scientific Review for the Identification of Critical Habitat for Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population, in Canada. Environment Canada, Ottawa.

Environment Canada. 2011. Recovery Strategy for the Woodland Caribou, Boreal population (*Rangifer tarandus caribou*) in Canada [Proposed]. *Species at Risk Act* Recovery Strategy Series. Environment Canada, Ottawa.

Girard, I., C. Dussault, J. P. Ouellet, R. Courtois, and A. Caron. 2006. Balancing number of locations with number of individuals in telemetry studies. Journal of Wildlife Management 70:1249-1256.

Hebblewhite, M., C. White, and M. Musiani. 2010. Revisiting Extinction in National Parks: Mountain Caribou in Banff. Conservation Biology 24:341-344.

Johnson, C. J., M. S. Boyce, R. Mulders, A. Gunn, R. J. Gau, H. D. Cluff, and R. L. Case. 2004. Quantifying patch distribution at multiple spatial scales: applications to wildlife-habitat models. Landscape Ecology 19:869-882.

Johnson, C. J. and M. P. Gillingham. 2008. Sensitivity of species-distribution models to error, bias, and model design: An application to resource selection functions for woodland caribou. Ecological Modelling 213:143-155.

Leban, F. A., M. J. Wisdom, E. O. Garton, B. K. Johnson, and J. G. Kie. 2001. Effect of sample size on the performance of resource selection analyses. Pages 291-307 *in* J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, New York, New York, USA.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource selection by animals : statistical design and analysis for field studies. 2nd edition. Kluwer Academic Publishers, Dordrecht ; Boston.

McLoughlin, P. D., D. W. Morris, D. Fortin, E. Vander Wal, and A. L. Contasti. 2010. Considering ecological dynamics in resource selection functions. Journal of Animal Ecology 79:4-12.

Meyer, C. B. and W. Thuiller. 2006. Accuracy of resource selection functions across spatial scales. Diversity and Distributions 12:288-297.

Moreau, G., D. Fortin, S. Couturier, and T. Duchesne. 2012. Multi-level functional responses for wildlife conservation: the case of threatened caribou in managed boreal forests. Journal of Applied Ecology 49:611-620.

O'Brien, D., M. Manseau, A. Fall, and M. J. Fortin. 2006. Testing the importance of spatial configuration of winter habitat for woodland caribou: An application of graph theory. Biological Conservation 130:70-83.

Racey, G. D. and A. A. Arsenault. 2007. In search of a critical habitat concept for woodland caribou, boreal population. Rangifer Special Issue 17:29-37.

Schaefer, J. A. and S. P. Mahoney. 2007. Effects of progressive clearcut logging on Newfoundland Caribou. Journal of Wildlife Management 71:1753-1757.

Smith, K. G., E. J. Ficht, D. Hobson, T. C. Sorensen, and D. Hervieux. 2000. Winter distribution of woodland caribou in relation to clear-cut logging in west-central Alberta. Canadian Journal of Zoology-Revue Canadienne De Zoologie 78:1433-1440.

Thomas, D. C. and D. R. Gray. 2002. Update COSEWIC status report on the woodland caribou Rangifer tarandus caribou in Canada, in COSEWIC assessment and update status report on the woodland caribou Rangifer tarandus caribou in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa.

Vors, L. S., J. A. Schaefer, B. A. Pond, A. R. Rodgers, and B. R. Patterson. 2007. Woodland caribou extirpation and anthropogenic landscape disturbance in Ontario. Journal of Wildlife Management 71:1249-1256.

Chapter 2: The Effect Of Sampling Regime On Woodland Caribou Range Delineation In Alberta

1. Introduction

Population delineation and use of space are critical components of many ecological investigations (Horne et al. 2008, Klaver et al. 2008). Delineating the spatiotemporal extent of a species or population range is also often the first step in developing conservation and wildlife management strategies (Bethke et al. 1996, Johnson et al. 2004). More specifically, home range estimation is fundamental to quantifying space use by animals and investigating animal-habitat relationships (Johnson 1980). Burt (1943) provided the first formal definition of a home range as the 'area traversed by the individual in its normal activities of food gathering, mating and caring for young'. This definition, however, makes no allowance for change with time and it is difficult to define 'normal' objectively (White and Garrott 1990). To address these issues in part, a home range may be described as the area with a specified probability of occurrence of an animal during a specified time period (White and Garrott 1990, Powell 2000, Kernohan et al. 2001). A home range is characterized by its size, shape, structure and location and is important for understanding a species spatial and behavioral ecology (Powell 2000). These characteristics may be affected by predator-prey relationships, competition, location of important resources or social pressures and mating systems (White and Garrott 1990, Powell 2000, Horne et al. 2008). Home range estimation methods have received much attention in the wildlife literature (e.g.

Harris et al. 1990, Powell 2000, Hemson et al. 2005), and some authors have even questioned the existence of a measurable home range (Gautestad and Mysterud 1995).

The main methodological issues associated with home range estimation include spatial and temporal autocorrelation of animal location data (Swihart and Slade 1985, Otis and White 1999, Fieberg 2007), sample size (Girard et al. 2002, Hemson et al. 2005), and differences among home range estimators (Powell 2000, Kenward 2001). Harris et al. (1990) and Laver and Kelly (2008) reviewed home range studies and provided recommendations for improved data collection and analysis. However, how home range estimates vary with sampling regime and how this affects statistical inferences has received little attention (except see Börger et al. 2006), nor have the conservation and management implications of these differences been clearly elaborated. In short, there are still no standardized data requirements for defining a home range (Laver and Kelly 2008). Sampling regimes vary substantially from study to study, depending on the specific research objectives and data collection method.

Radio telemetry involves the remote collection of animal location data, and allows researchers to gather data on species distribution, behaviour, movement and resource selection (Cooke 2008). Recent advancements in radio telemetry have lead to the widespread use of global positioning system (GPS) collars, which are considered superior to their predecessors, very high frequency (VHF) and satellite (Argos) collars, due to ease of data collection, increased number and accuracy of data locations. GPS collars can collect thousands of data locations on an animal over a two to three year period, whereas the number of locations for a VHF collar depends on the search effort of the researcher and varies greatly from daily to monthly to quarterly data locations.

Sample size can affect home range estimates (Girard et al. 2002, Hemson et al. 2005), leading to errors in estimating a species home range. Recent studies have provided suggestions for appropriate sample sizes to accurately determine home range size for several species. Boulet et al. (2007) used bootstrapping simulations of ARGOS location data for migratory caribou (*Rangifer tarandus caribou*) to determine that a minimum number of 12 animals per population per year were necessary to generate nonbiased estimates for their study area. A study on European roe deer (*Capreolus capreolus*) suggested that 10 VHF locations per month were sufficient for accurate estimates of individual home range size, and also that the sampling interval between fixes may influence home range estimates (Börger et al. 2006). However, Girard et al. (2002) indicated that 30 to 100 data locations were needed seasonally to define a home range for an individual moose (*Alces alces*), suggesting that GPS telemetry is better suited than VHF telemetry to estimate home range sizes precisely and accurately. GPS technology provides larger sample sizes, but may also lead to a lack of independence among sub samples. Otis and White (1999) reviewed the implications of spatial autocorrelation in home range and resource selection analyses and recommended the use of individual animals as the sample unit, instead of pooling location data across animals. This is particularly relevant when the objective is to make valid population inferences.

Although home range methods are usually applied at the individual level, defining the range for a population is an extension of these methods, by including location data for animals within the population, not annually or seasonally, but over the whole timeframe of sampling. Population ranges can be defined using a variety of data sources and methods. Jurisdictional range estimates often incorporate long-term observational data, a combination of GPS and VHF telemetry data, and expert opinion (e.g., Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA) 2010). With the inclusion of multiple data sources over a long temporal scale, a jurisdictional range often represents a precautionary approach, resulting in a larger delineated range. Clearly, given the issues described above, care must be taken when delineating a range using location data, as failure to recognize the basic assumptions and limitations of home range estimation methods may lead to biased range delineations and consequentially invalid inferences about animal-habitat relationships.

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and the Alberta Wildlife Act currently list woodland caribou (*Rangifer tarandus caribou*) as a threatened species in Canada and Alberta, respectively. Under the Alberta Wildlife Act and the Canadian Species at Risk Act, development of recovery plans for threatened or endangered species is required. Recovery strategies are detailed plans that outline short-term objectives and long-term goals to restore species identified as threatened or endangered to viable, naturally self-sustaining populations (SARA 2002). Recent woodland caribou recovery plans have identified the importance of delineation of not only the species geographical range but also ranges for local populations or herds (Thomas and Gray 2002, Racey and Arsenault 2007, Environment Canada 2008, Ontario Woodland Caribou Recovery Team 2008). A geographical range refers to the extent of the species occurrence, whether being a national or global occurrence. A local population or herd range (hereafter called population range) is a group of caribou occupying a defined area distinguished spatially from areas occupied by other groups of caribou, and is considered the basic unit for conservation and management of woodland caribou in Canada (Thomas and Gray 2002, Environment Canada 2008).

Alberta is considered one of the leading jurisdictions in Canada with regard to woodland caribou monitoring, with over 20 years of extensive monitoring of populations. Currently, sixteen caribou populations remain in the province of Alberta, representing both boreal (northern Alberta) and central mountain (west central Alberta) Designatable Units (DU6 and DU8 respectively, COSEWIC 2011). Boreal caribou populations inhabit peatland complexes interspersed with upland forests, generally do not occupy distinct seasonal ranges and are considered non-migratory (Dzus 2001), whereas mountain caribou exhibit seasonal migratory movements between spring and summer alpine areas to lower elevation mixed conifer forests in the winter (Edmonds 1988, Saher 2005). Both boreal and mountain caribou populations are reported to be declining in Alberta, a recent status report noted that of the 13 populations with sufficient monitoring data, 10 are declining and these comprise the majority of the caribou occurring in the province (ASRD and ACA 2010).

In this chapter, I explore how sampling regimes effect range delineation for woodland caribou using empirical data collected in Alberta from 1998 to 2010. I apply a subsampling procedure to caribou GPS location data to simulate increasingly less intense sampling regimes commonly found in conventional telemetry studies. My specific objectives for this chapter are to: 1) quantify caribou range delineation in relation to the number of animals sampled (sample size) and the tracking schedule (sampling frequency) used, and 2) compare home range estimates from two commonly used home range estimators, the minimum convex polygon (MCP) and the fixed kernel density estimator (KDE). I predicted that as the sampling frequency decreases, the number of animals sampled will have to increase in order to define a population range that represents the variation present in the local population. I explore these questions for both non-migratory (boreal) and migratory (mountain) caribou local population ranges; and provide recommendations for sampling and analysis protocols to help guide local population delineation for woodland caribou.

2. Study Area

This research was conducted in woodland caribou ranges in Alberta, where GPS location data were available for both boreal and central mountain DUs (Figure 2.1). Data for boreal caribou included the Little Smoky (LSM), Slave Lake (SL), Nipisi (NIP), West Side of the Athabasca (WSAR), and Chinchaga

(CHIN) populations. Data for mountain caribou included the A la Pêche (ALP), Redrock Prairie Creek (RPC) and Narraway (NAR) populations.

Boreal and mountain populations of caribou differ in their seasonal movements and habitat use patterns. Boreal caribou occur in northern Alberta (with the exception of Little Smoky in west-central Alberta), are considered sedentary, and typically show considerable overlap between winter and summer ranges (Dzus 2001). Mountain caribou populations are located in west-central Alberta, and are considered migratory, making seasonal migrations between alpine and subalpine summer ranges (in both Alberta and British Columbia) and lower elevation foothill winter ranges in Alberta (Edmonds and Bloomfield 1984, Brown and Hobson 1998).

Mountain caribou occupy habitats classed into Lower and Upper Foothills, Subalpine, and Alpine Natural Subregions (Natural Regions Committee 2006). The Foothills subregion consists of upland areas characterized primarily by lodgepole pine (*Pinus contorta*) or lodgepole pine/white spruce (*Picea glauca*) forests with small patches of trembling aspen (*Populus tremuloides*) and lowland areas with poor drainage are characterized by black spruce (*P. mariana*) and larch (*Larix laricina*). At higher elevations, the Subalpine subregion is dominated by Engelmann spruce (*P. engelmannii*) and subalpine fir (*Abies lasiocarpa*) forests, while the Alpine subregion has few trees and is characterized by graminoids, sedges (*Carex spp.*) and wind-swept ridges.

Boreal caribou occupy habitats classed into Central and Dry Mixedwood, Lower and Upper Boreal Highlands, Lower and Upper Foothills and Subalpine

Natural Subregions (Natural Regions Committee 2006). The Central and Dry Mixedwood subregion is characterized by white spruce, jack pine (*P. banksiana*) and trembling aspen, whereas the Lower and Upper Boreal Highlands are characterized by low-lying peatlands with black spruce and larch.

In addition to woodland caribou, the study area supports a variety of other ungulate species, including moose (*Alces alces*), elk (*Cervus elaphus*), white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), bighorn sheep (*Ovis canadensis*), mountain goats (*Oreamnos americanus*), and their predators, wolves (*Canus lupus*), grizzly bears (*Ursus arctos*), black bears (*U. americanus*), cougars (*Felis concolor*), lynx (*Lynx canadensis*), wolverine (*Gulo gulo*) and coyotes (*C. latrans*).

Various land use activities occur within caribou ranges across the province, and vary in intensity depending on the area. These include forestry, oil and gas exploration and development, mining, non-motorized outdoor recreation (hiking, horse travel, camping, fishing), off-road vehicle use (snowmobile, all-terrain vehicles), recreational hunting, and commercial trapping (Brown and Hobson 1998, Dyer et al. 2001).

3. Methods

3.1 Caribou location data

During 1998-2010, in collaboration with Weyerhaeuser Company Ltd. Grande Prairie, the University of Alberta, and Alberta Sustainable Resource Development, adult female mountain caribou were captured using helicopter net-gunning techniques and fitted with GPS telemetry collars (Lotek GPS1000, 2000, 2200, 3300, and 4400 models; Lotek Engineering, Newmarket, Ontario, Canada). Capture protocols were approved by the University of Alberta Animal Care Committee (Protocol Number 731910). Between 1998 and 2010, GPS collars were deployed on 146 individual caribou in three mountain caribou populations, resulting in data for 445,912 locations. GPS location data for the boreal caribou populations were obtained through a data sharing agreement with the Alberta Caribou Committee and Alberta Sustainable Resource Development. During 1998-2010, GPS collars were deployed on 111 individual caribou in six boreal caribou populations, resulting in data for 399,786 locations (Table 2.1).

Since the GPS location data were collected over multiple years and under different research and government projects, the GPS collar type varied, and the tracking schedule varied from a location every 15 minutes to every 24 hours, with an average GPS tracking schedule of one location every 4 hours. The two types of GPS collars deployed collected either solved or differential locations, resulting in a positional accuracy of 10-35 and 5-9 meters (95% of the time), respectively. The GPS collars with differential data were differentially corrected using N4win (Lotek Wireless Inc. 1999). All locations with a horizontal dilution of precision (HDOP) greater than 12, indicating erroneous location accuracy, were removed prior to analysis (D'Eon and Delparte 2005). Caribou locations were imported into a geographic information system (GIS; ArcGIS 9.2, Environmental Systems Research Institute, Redlands, CA, USA) and any additional erroneous locations were removed from the dataset, including locations clearly outside the specific study area, and extremely large movements between fixes (e.g. a location 20 km away between fixes).

3.2 Caribou range delineation

I avoided potential bias by estimating population range sizes using only complete sets of annual data collected on caribou, starting from the date of capture to 12 months post capture. Individual caribou were removed from the analysis if they did not have at least 12 consecutive months of location data, as were locations outside the standardized sampling interval of 12 months for each individual.

Population range size was estimated using a two tiered sampling regime, varying both the number of animals (sample size) included from the population, and the frequency (sampling frequency) with which they were sampled (Figure 2.2). First, the number of animals included in the estimate was selected by randomly removing one caribou at a time, without replacement, from the total number of animals available for each population. Second, in order to simulate less intense sampling regimes commonly found in conventional VHF telemetry studies, six tracking schedules were selected based on systematic subsampling of the data locations from this set of animals. The tracking schedules included one location every 1, 4, 7, 14, 30, and 60 days. Start dates for each animal were randomly selected from the first week of data collection, and from this date, a random location was selected from the next sampled day in the tracking schedule. This procedure was replicated ten times for each set of animals, and the random selection of sets of animals was replicated 100 times (Figure 2.2). The subsampling procedure was written in R version 2.10.1 (R Development Core Team 2009). As each population was sampled over various sample sizes (number of animals included), the number of replicates per population varied from 18,000 for the Nipisi population (N = 3) to 216,000 for the Redrock Prairie Creek population (N = 36).

I estimated annual range size for each population using two commonly used home range estimation methods: minimum convex polygon (MPC; Mohr 1947) and fixed kernel density (KDE; Worton 1987; Figure 2.3). All population ranges were calculated in R 2.10.1 with the adehabitat package (Calenge 2006). The MCP method is one of the oldest and most used home range estimators (Harris et al. 1990, Börger et al. 2006) and is simply the minimum area polygon that encompasses all recorded locations for an animal (White and Garrott 1990, Powell 2000). The popularity of the MCP method is mainly due to the ease of use and interpretation. However, authors have noted that the method is highly sensitive to sample sizes, spatial resolution and sampling duration (Hansteen et al. 1997, Powell 2000, Kenward 2001), leading to suggestions that MCPs should not be used as a home range estimator (Börger et al. 2006, Laver and Kelly 2008). Kernel density estimation is a nonparametric, probabilistic method, which calculates the density of space use, with home range boundaries built by joining sites with equal density (Worton 1987). The KDE method is robust to changes in spatial resolution of the data (Hansteen et al. 1997) and can account for multiple centres of activity (Powell 2000, Kernohan et al. 2001). The choice of the smoothing parameter or bandwidth for the KDE method can lead to either over or under estimating an animal's home range. The least squares cross validation method (h_{lscv}) is generally recommended over the reference method (h_{ref}; Seaman and Powell 1996, Seaman et al. 1999) as the appropriate smoothing parameter. However estimates using the h_{lscv} smoothing parameter are highly variable at low sample sizes (Hemson et al. 2005). Börger et al. (2006) found that both smoothing parameters gave comparable results in terms of accuracy and precision and recommend the h_{ref} method as a conservative estimate. I applied both smoothing parameters in order to compare the results. To ensure comparability between the two estimation methods, I used the 100% MCP and the 95% KDE methods because both estimates are based on all the data locations collected (Börger et al. 2006).

3.3 Sample regime implications

Mean population range size and variance were calculated for each estimation method and data set (i.e. sample size and sampling frequency). For each estimation method, accuracy of population range size estimates was assessed by dividing the area obtained using a particular sampling regime by the area obtained using the most intensive sampling regime (baseline condition: all caribou and all locations available). This baseline condition was considered to provide the least biased home range size estimates (Girard et al. 2002). For each sampling schedule, and estimator, I also evaluated the mean and standard deviation (SD) of the number of animals needed (hereafter referred to as the minimum number of animals) to obtain a population range <20% smaller than the one issued from the baseline (most intensive) sampling frequency (Girard et al. 2002).

To examine the influence of each explanatory variable (number of caribou (Ncar), number of locations (Nloc), sampling frequency (SamFreq) and number of years (Nyrs) on the response variable (Range size) I used multiple regression. I first examined the residuals for each variable for normality and homogeneity of variance using univariate regression. In addition, I examined all explanatory variables for collinearity, using a correlation coefficient cut-off of 0.7 to exclude collinear variables from the same model. R version 2.10.1 (R Development Core Team 2009) was used for all analyses.

Mean population range area for each sample size and sampling frequency was plotted against number of caribou. I used the asymptote of this relationship to identify sample size requirements. There is currently no defined method for assessing when an asymptote is approached; however, Laver and Kelly (2008) suggest using the value at which the home range estimate is within a specified percentage (e.g. 5-10%) of the total home range size using all locations for at least five consecutive home range estimates. I examined the effect of sample size and sampling frequency on population range size, by identifying the number of animals needed for the population range area to reach an asymptote defined by the point at which the population range estimate was within 5% of the total population range size for five consecutive home range estimates.

3.4 Population range comparisons

There are various ways to estimate an animal's home range, and local population delineation for management purposes often includes a variety of data sources such as telemetry location data (GPS, VHF and ARGOS), long-term observation data and expert knowledge. I compared the baseline population range sizes for the MCP and KDE with the jurisdictional population range delineated in the most recent status of the woodland caribou in Alberta report (ASRD and ACA 2010). For comparison, I calculated the population range area within Alberta by clipping the inter-provincial baseline population ranges to the Alberta provincial boundary.

4. Results

4.1 Caribou location data

A total of 255,204 locations were retained from 68 boreal caribou ($\bar{x}$ = 3,753; SD = 1,918 per caribou), and 221,883 locations from 84 mountain caribou ($\bar{x}$ = 2,614; SD = 1,487), for which at least 11 months of continuous data were available (Table 2.2; Figure 2.1). Despite having fewer animals, a larger total number of locations were available for boreal caribou, principally because caribou in the WSAR herd had a variable sampling schedule that included a 15-minute sampling interval. Of note in the mountain DU is the substantial reduction in the number of caribou included in the RPC population, from 76 to 36, due to a variety of factors including animal mortality (11 caribou), collar failure (16 caribou) and poor fix rate or unknown factors (13 caribou).

4.2 Caribou range delineation

Population ranges were calculated for each sample size and sampling frequency, and replicated 100 and 10 times, respectively. The number of replicates for the boreal DU varied from 18,000 for NIP to 138,000 for LSM, and for the mountain DU from 126,000 for ALP to 216,000 for RPC (Table 2.3).

MCP population range sizes varied from 226 to 1,181,448 ha for the boreal DU, and 832 to 574,968 ha for the mountain DU (Table 2.3). The h_{lscv} smoothing parameter method for the kernel density estimator failed, which is common with large sample sizes (Hemson et al. 2005). As a result, I used the h_{ref} smoothing parameter for the kernel density estimates. KDE population range sizes varied from 1,897 ha to 3,344,786 ha for the boreal DU and 6,977 ha to 2,400,679 ha for the mountain DU (Table 2.3). Baseline population range size is considered to represent the best approximation of the population ranges since it contains the most intensive tracking schedule including all caribou and all locations. Baseline MCP population range sizes varied from 220,115 to 1,191,010 ha for the boreal DU and 417,300 to 578,703 ha for the mountain DU. The baseline KDE population range sizes varied from 64,743 ha to 777,245 ha for the boreal DU and 302,984 ha to 415,387 ha for the mountain DU (Table 2.4; MCP Ranges: Figure 2.4; KDE Ranges: Figure 2.5).

4.3 Sample regime implications

The variables number of caribou (Ncar), number of locations (Nloc), sampling frequency (SamFreq) and number of years (Nyrs) were not normally distributed, so Spearman Rank correlation was used to examine relationships between the candidate predictor variables. Sampling frequency and number of locations were highly correlated ($r_{SamFreq, Nloc} = -0.82$). Given a primary objective was to examine how different sampling designs affect the home range size estimation, I retained sampling frequency for further analyses. I used univariate regression analyses to explore the relationship of each predictor variable with both the MCP and KDE response variables. However, as none of the variables conformed to assumptions of normality and homogeneity of variance, and transformations did not improve the data distributions, I attempted to fit the data using generalized linear models (GLMs), with a gamma distribution and identity link function. The GLMs did not improve the fit of the data, so I reverted to standard linear models.

I employed a manual stepwise model building approach, with a cut-off p value of 0.10 for variables to be added or removed from the model, and used the RPC range estimates for one sampling regime (i.e. number of animals replicate = 4, sampling frequency replicate = 1) to conduct an exploratory analysis. The final MCP model included all variables (Ncar, Nyrs, SamFreq) and explained 93% (adjusted R^2) of the variation in population range size ($p < 0.001$). The RPC KDE final model included the same variables, but only explained 68% of the variation in population range size ($p < 0.001$). Because the data failed to conform to the assumptions of multiple regression and GLM, I was unable to partition variance among the predictor variables. Nevertheless, a notable outcome of this analysis was the effect of the trial variable (the random selection of animals repeated 100 times) on the model. Across trials, the Ncar variable was significant 85% and 89% of the time for the MCP and KDE range size, respectively, suggesting that individuals, and the order in which they were removed from the sample, affected range size estimation.

To further examine the implication of the number of animals sampled on range size estimation, I used the results from the most intense sampling frequency of one location per animal per day to generate accumulation curves. For the MCP method, no boreal population range estimate approached asymptote (Figure 2.6), suggesting that for these populations, an insufficient number of animals were sampled to confidently define a range using this estimation method. Among the mountain DU, the Narraway and Redrock Prairie Creek populations approached an asymptote at 26 and 18 animals, respectively (Figure 2.6). Conversely, for the KDE method, the Chinchaga and Little Smoky populations in the boreal DU approached an asymptote at 10 and 14 animals, respectively, and the A la Pêche, Narraway and Redrock Prairie Creek populations in the mountain DU reached asymptotes at 10, 7, and 8 animals, respectively (Figure 2.7, Figure 2.8).

To further examine how the estimate of population range varied with less intensive sampling frequency, I used the population with the largest data set - the Redrock Prairie Creek population - to generate accumulation curves for comparison with the baseline generated with all animals and locations. For the MCP method, the estimate of population range decreased with less intensive sampling frequency (Figure 2.9A; Figure 2.10A), whereas the KDE home range was overestimated when sampling was less frequent (Figure 2.9B, Figure 2.10B). The minimum number of animals required to reach asymptote varied from 18 to 23 for the MCP method, and 8 to 24 for the KDE method, for the sampling frequencies of 1 location every day, and 1 location every 60 days, respectively (Table 2.4).

4.4 Population range comparisons

After removing the British Columbia portion of inter-provincial populations, range sizes were 31%, 85%, and 26% smaller than the total population range, for CHIN, NAR and RPC, respectively. All KDE baseline ranges estimates for both boreal and mountain populations were smaller than the MCP estimates and the delineated jurisdictional ranges (Figure 2.11). However, the jurisdictional range was smaller than the MCP baseline range for the LSM, NIP, and SLAVE populations. For the largest jurisdictional range in Alberta, the Chinchaga, the MCP and KDE population areas were 54% and 70% smaller, respectively. For the smallest jurisdictional range, the Narraway, the MCP and KDE population areas were 41% and 66% smaller, respectively.

5. Discussion

Identification of ranges for species or populations is critical to successful integration of conservation and land use management. Addressing complex issues related to delineating ranges requires a clear understanding of the data requirements for sampling animal populations. Most home range methods are applied at the individual animal level, however adequate conservation planning for wide ranging species such as woodland caribou requires delineation of ranges at the local population level. With the emergence of GPS technology, biologists have faced trade-offs in the design of sampling programs to collect animal location data; specifically regarding the number of locations collected for each animal versus the number of animals sampled (Girard et al. 2006). I estimated population range size for eight caribou local populations in Alberta, Canada, compared two different home range estimation methods, and addressed the role of sampling regime when defining population ranges. Finally, I identified how many caribou should be sampled to make general statements regarding range delineation for both boreal and mountain DUs in Alberta.

Population ranges were estimated for five boreal and three mountain local populations, with the number of animals available per range varying from 3 in the Nipisi population to 36 in the Redrock Prairie Creek population. Several range estimates failed to approach asymptotes, indicating sample size was inadequate for those ranges, and thus resultant range sizes may be biased (Harris et al. 1990). Depending on the population and the estimation method, up to 26 caribou with annual data were required for stable population range estimates. The minimum number of animals and sampling frequency needed was 22 animals with a 14 day sampling frequency, or 8 animals with a 1 day sampling frequency, for the MCP and KDE estimates, respectively.

My analysis reinforces the notion that sampling regime strongly influences range size estimates (Powell 2000, Kernohan et al. 2001, Girard et al. 2002). However, to my knowledge, no previous studies have examined how sample size and sampling regime affects the delineation of population ranges, as most studies have focused on how sample size affects the delineation of individual animal home ranges (Girard et al. 2002, Hemson et al. 2005, Börger et al. 2006). One study defined population ranges for investigating genetic connectivity, and suggested that 12 animals per population per year were necessary to define a MCP population range in a non-biased way (Boulet et al. 2007). An important distinction is that the minimum number of animals I report is not for defining the yearly population range, but the total population range for the entire length of sampling. This suggests that the temporal scale over which populations are sampled also affects how robust the defined local population range is.

When examining the sampling frequency required for population range estimation, I found that as the sampling frequency decreased, the number of animals required for population range estimates to asymptote increased. I used data from the Redrock Prairie Creek population to illustrate these trends because it had the largest sample size and was able to provide unbiased population range estimates using KDE and MCP methods. It has been noted previously that fewer numbers of locations are required to define unbiased KDE home ranges compared to MCP ranges (Hemson et al. 2005, Börger et al. 2006). However, the KDE and MCP estimates responded in an opposite manner to decreasing sampling frequency.

The subsampled MCP population range estimates were smaller than the baseline MCP (Figures 2.10A, 2.13A). Even with a large number of animals sampled in the population, a 60-day sampling frequency would significantly underestimate the true population range (30% smaller than the baseline).

In contrast, the areal extent of KDE estimates were greater with decreasing sampling frequency (Figures 2.10B, 2.13B), a finding similar to other studies (Girard et al. 2002, Joly 2005). The KDE population estimates produced population ranges that were 20% larger than the baseline KDE for all sampling frequencies except the 1-day sampling regime. Seaman et al. (1999) reported that the overestimation of home range sizes with kernels calculated from small sample sizes occurs because the bandwidth parameter increases when the sample size decreases, thus estimating a large population range area. In my study, the KDE method began to overestimate the kernel range size at the 4-day sampling frequency (31% larger than the baseline), and if increased to 60 days, the kernel home range size would be overestimated by 64%. This result contrasts with the findings of Börger et al. (2006), who suggested that an accurate estimate of individual home range size could be achieved with a 4-day sampling regime (<25% bias for 95% KDE).

Not surprisingly, the MCP population ranges were larger in areal extent than the KDE population ranges. While both methods include all data locations, the 100% MCP method draws a convex polygon around all locations, whereas the 95% KDE method draws a 95% density isopleth. Sample size was sufficient for kernel estimates to asymptote for all populations except for the Nipisi, Slave Lake and West Side of the Athabasca River populations, whereas the MCP estimates only asymptoted for the Narraway and Redrock Prairie Creek populations. Very small numbers of animals were sampled in the NIP and SLAVE populations (n = 3 and n = 5 respectively), but 21 caribou were sampled from the WSAR population. WSAR data were collected over a two-year sampling window, with collaring concentrated in the southern portion of the range, and apparently did not adequately sample the full distribution of the population.

This illustrates the temporal constraints in using GPS collars to define population ranges, where collecting a large amount of data over a short time fails to capture the variability present (Kie et al. 2010). In the WSAR case, this is exacerbated by a bias in the animals sampled. In other words, the data were both temporally and spatially biased. This also highlights the need to consider proportional sampling strategies, whereby the number of animals required to represent the population could vary with population size. Overall, MCP estimation based on available data failed to adequately define population ranges in most areas of the province. Exploration of an extensively sampled mountain population suggested that 18-26 animals sampled up to ten years may be necessary for this purpose.

Some studies have recommended that the MCP method should not be used for range estimation due to potential biases (Börger et al. 2006, Laver and Kelly 2008). However, I suggest that if an adequate sample size is available, the MCP method is appropriate for defining population ranges for species at risk. The MCP method has been favoured over the KDE method because it allows for a precautionary approach towards population range delineation (Environment Canada 2008). KDE estimates are substantially smaller than MCP estimates, and also more fragmented (Figure 2.5). Areas with lower concentrations of locations, such as migration routes, are typically excluded from KDE ranges. Burt (1943) suggested that dispersal and occasional sallies outside the normal area should not be considered as part of an animal's home range. However, these movements may be critical for maintaining connectivity within and across caribou ranges. For example, the fragmented WSAR KDE population range is split into several polygons, and the same caribou have been recorded in multiple polygons, yet the area between the polygons is not represented (Figure 2.5). Representation of areas of concentrated use may be appropriate for certain management applications, such as defining core areas, but is not sufficient for comprehensive conservation planning.

Defining population ranges for conservation and management requires use of the best available data, but sometimes these data may not be adequate to delineate a range with confidence. Jurisdictional ranges in Alberta were delineated using up to 35 years of caribou observations, radio telemetry data (VHF and GPS), local knowledge, and biophysical analyses (ASRD and ACA 2010). Depending on the caribou population, the jurisdictional area was either smaller or larger than the ranges delineated with MCP and KDE methods. The existence of larger jurisdictional ranges is not surprising, as a longer time frame and broader suite of information was included in the jurisdictional delineations. In contrast, smaller jurisdictional range delineations present a conservation dilemma, particularly where sampling was considered inadequate for MCP estimation. In these cases, which include the LSM, NIP and SLAVE populations of the boreal DU, the area over which current conservation and management planning is occurring may underestimate the needs for effective conservation of local caribou populations.

The jurisdictional areas also do not include data locations outside of the province (Figure 2.1); therefore, the population boundaries for the Chinchaga, Narraway, and Redrock Prairie Creek were smaller than their actual range area, although this was accounted for in comparisons. These inter-jurisdictional caribou populations, however, underscore illustrate the importance of coordinated management of caribou populations. Delineation of population ranges should be based on all available data from all jurisdictions in which individuals occur. Finally, given that some woodland caribou populations may occupy distinct seasonal ranges, sampling regimes should account for temporal variability in occupancy when delineating local populations.

In general, my results suggest that boreal caribou populations occupy larger ranges than mountain populations. More detailed comparisons are difficult, because too few animals were sampled to provide unbiased range sizes for boreal populations. Only the CHIN and LSM range estimates reached asymptotes for the KDE estimates and no MCP estimates asymptoted for the boreal populations. Although boreal populations have limited GPS location data, they have been sampled for a number of years with VHF collars, at a sampling frequency of 4-5 locations per year. However, my analysis indicates this sampling frequency is insufficient for estimating unbiased population range sizes. More broadly, a report on boreal populations in Canada indicated that over half of the ranges had insufficient data to delineate local population ranges (Environment Canada 2011). This assessment also found that ranges with a higher percentage of anthropogenic disturbance were smaller in size, suggesting that delays in delineating ranges could compromise conservation efforts by failing to accurately identify the spatial requirements of populations. My results provide guidance for sample size requirements for delineating population ranges and can inform sampling designs for jurisdictions with insufficient data.

6. Conclusion

Accurately delineating a species or population range depends on both the sample size and sampling frequency of location data available. Since range delineation is often the first step in conservation and management planning, it is important to identify whether the sample size available is sufficient to make population level inferences. In this study, both the number of caribou and years sampled influenced population level inferences regarding range size. In addition, as sampling frequency decreased, the number of caribou required for population ranges to asymptote increased. Trade-offs in sampling regime varied depending on the range estimator, which also has implications for the type of location data collected. Whereas the KDE method appears best suited to GPS data, the MCP method was robust to lower sampling frequencies, and thus reliable populations delineations could be derived from VHF data collected biweekly.

MCP estimation of population ranges represents a precautionary approach to range delineation, and is appropriate for identifying ranges for species at risk, for use in broad-scale conservation and management. However, KDE of core areas within a population range can augment understanding of range use. Current VHF sampling frequencies in many boreal populations in Alberta (4-5 locations per year) are likely to provide biased population range estimates, therefore increased sampling frequencies are recommended. In general, a critical examination of available data prior to analysis is essential if the resulting range delineations are to reflect the distribution of the local populations and provide appropriate guidance for conservation and management decisions.

Table 2.1: Available GPS location data for boreal and mountain caribou DUs in Alberta.

Ecotype	Population	No. Animals	No. Locations	Date Range	Estimated Population Size[a]
Boreal	Chinchaga	16	92,942	2007 - 2010	250 - 300
	Little Smoky	41	95,135	1999 - 2009	80
	Nipisi	3	24,588	2006 - 2010	60 - 70
	Richardson	11	7,210	2008 - 2009	Unknown
	Slave Lake	5	65,261	2006 - 2008	75
	WSAR	35	114,650	1998 - 2000	300 - 400
Mountain	A la Peche	26	111,726	2001 - 2010	150
	Narraway	44	116,222	2000 - 2009	100
	Redrock Prairie Creek	76	217,964	1998 - 2009	325

([a] Source: ASRD & ACA 2010)

Table 2.2: GPS location data included in the range delineation analysis for boreal and mountain caribou DUs in Alberta. Annual data from animals with 11 consecutive months of location data were included, and any locations after 12 months of data were removed.

Ecotype	Population	No. Animals	No. Locations	Date Range
Boreal	Chinchaga	16	74,736	2007 - 2010
	Little Smoky	23	55,708	1999 - 2009
	Nipisi	3	17,278	2006 - 2010
	Richardson	0	0	0
	Slave Lake	5	43,676	2006 - 2008
	WSAR	21	63,806	1998 - 2000
Mountain	A la Peche	21	83,140	2002 - 2009
	Narraway	27	54,715	2000 - 2008
	Redrock Prairie Creek	36	84,028	1998 - 2008

Table 2.3: Summary of Alberta woodland caribou population range size estimates. Number of replicates increases with the sample available for each population. Minimum and maximum population range sizes for each home range method are based on the subsampled data set. Baseline population home range size is based on all animals and all data locations for each population.

		Number of	MCP Area (Ha)			KDE Area (Ha)		
Ecotype	Population	Replicates	Min	Max	Baseline	Min	Max	Baseline
Boreal	CHIN	96,000	5,078	1,181,448	1,191,010	70,458	3,344,786	777,245
	LSM	138,000	447	333,287	335,440	6,268	771,121	125,209
	NIP	18,000	297	311,960	318,347	4,025	2,414,718	157,106
	SLAVE	30,000	226	212,093	220,115	1,898	1,351,036	64,743
	WSAR	126,000	236	1,147,461	1,149,370	3,787	2,691,782	330,200
Mountain	ALP	126,000	1,151	574,968	578,704	21,928	2,100,549	327,449
	NAR	162,000	832	414,458	417,301	6,977	1,736,226	302,985
	RPC	216,000	1,951	550,590	554,598	15,581	2,400,679	415,387

Table 2.4: Minimum number of animals required to reach asymptote and the corresponding ratio of estimated population range size compared to the baseline value for each sampling frequency.

	Number of Animals		Ratio to Baseline	
Sampling Frequency	MCP	KDE	MCP	KDE
1	19	8	0.91	1.11
4	21	14	0.88	1.31
7	21	16	0.85	1.36
14	22	19	0.81	1.46
30	23	21	0.78	1.58
60	23	24	0.70	1.64

Figure 2.1: Study area for population range delineation analysis. Alberta woodland caribou population boundaries are defined based on up to 35 years of caribou sightings, telemetry data (VHF and GPS), local knowledge and biophysical analyses (Source: Alberta Caribou Committee 2010). GPS data were used for boreal DUs: CHIN, LSM, NIP, SLAVE, WSAR and mountain DUs: ALP, NAR, RPC. Populations coloured in green were not included in the analysis.

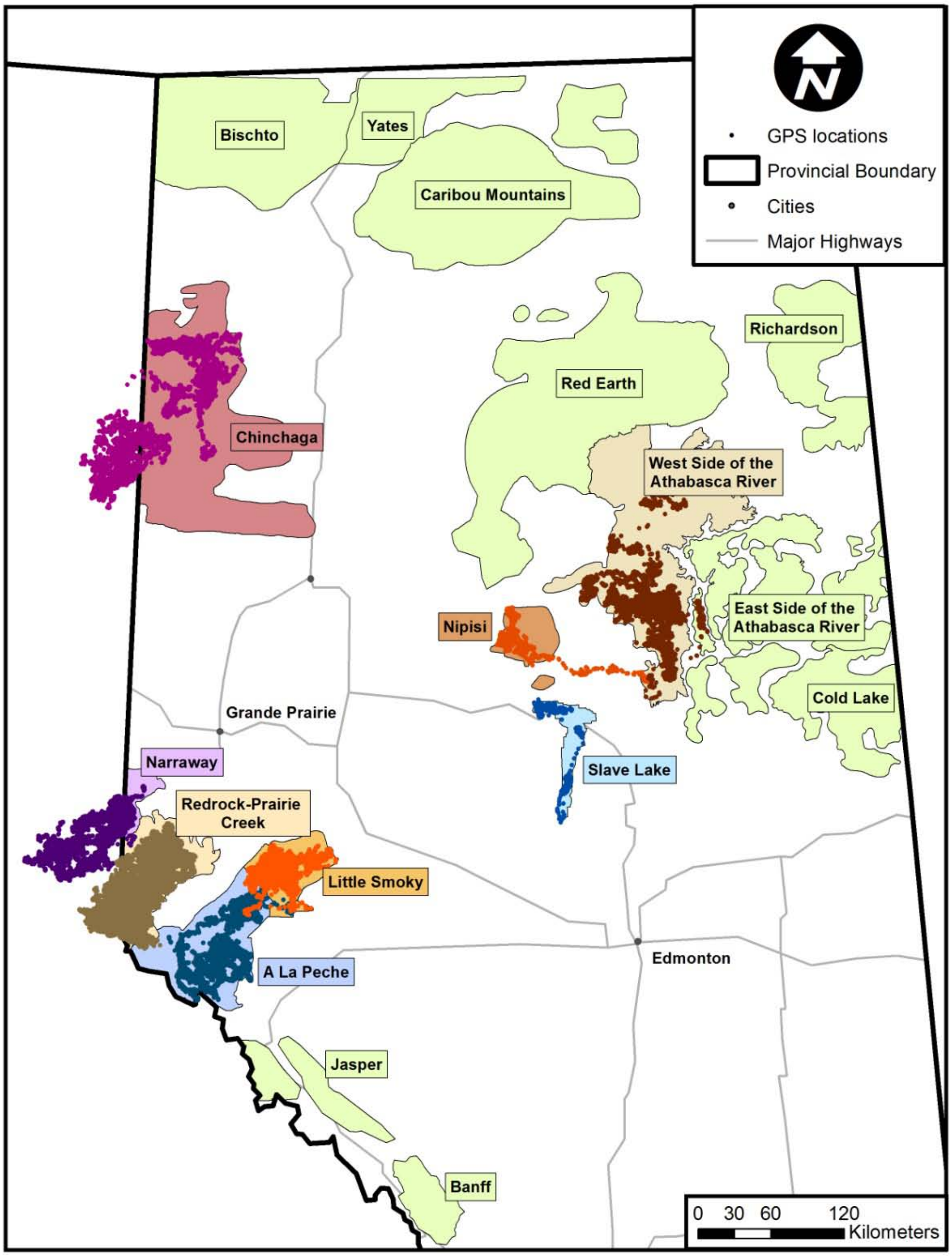


Figure 2.2: Flowchart of subsampling procedure for population range delineation analysis. Sampling regime includes varying the number of animals by randomly removing one individual at a time and varying the sampling frequency by 1 data location every 1, 4, 7, 14, 30, and 60 days. Each sampling frequency were replicated 10 times for each sample of caribou. The number of animals sequence was replicated 100 times.

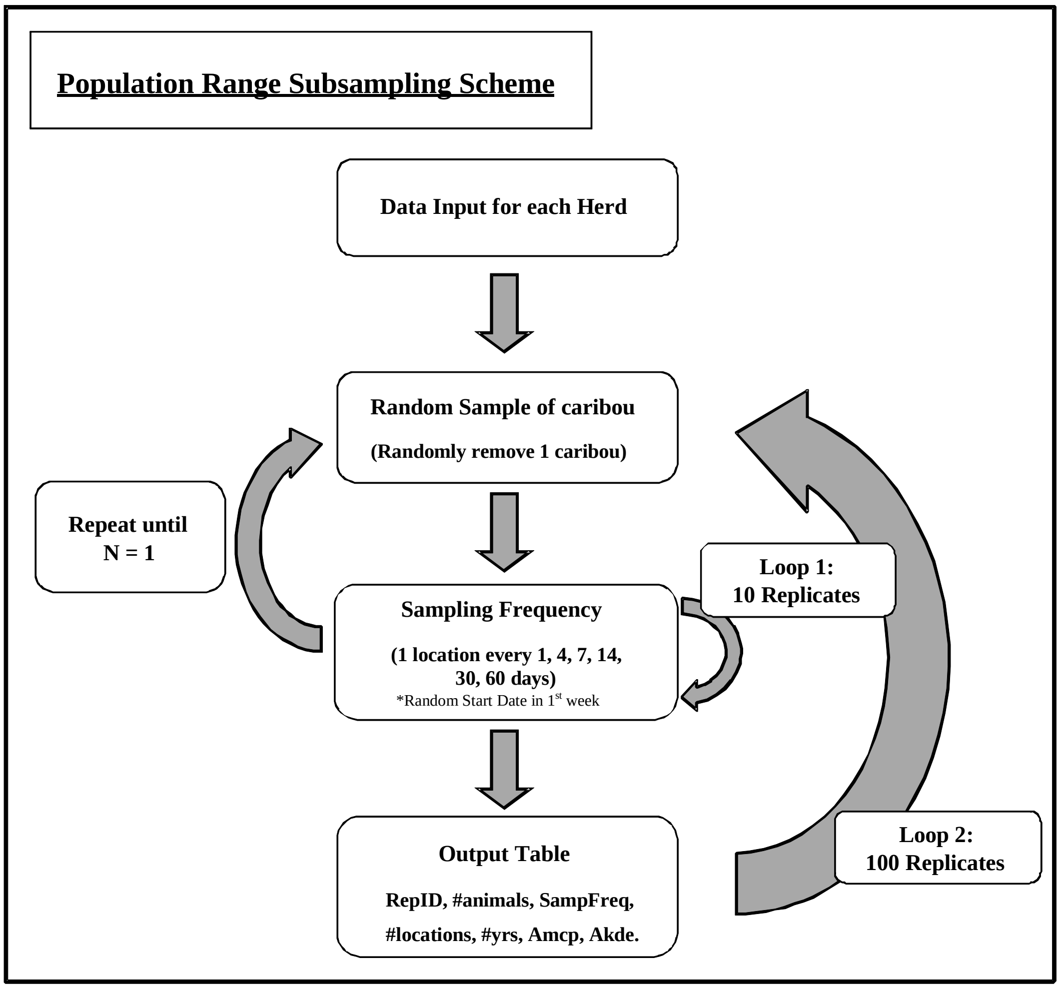


Figure 2.3: Comparison of home range estimators used in the population range delineation analysis. A. 100% Minimum Convex Polygon method. B. 95% Kernel Density Estimator method.

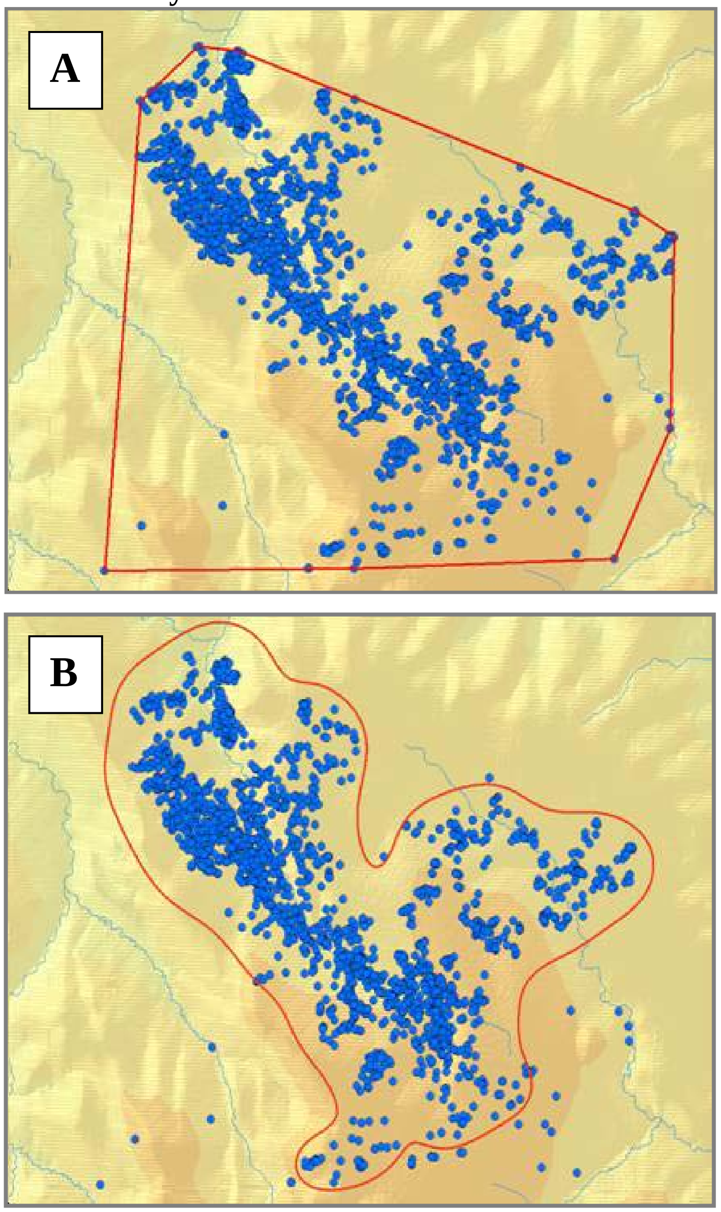


Figure 2.4: 100% Minimum Convex Polygon (MCP) population ranges for both the boreal and mountain DUs. MCP areas represent the baseline condition for each population including all available animals and all available locations. Populations coloured in green were not included in the analysis.

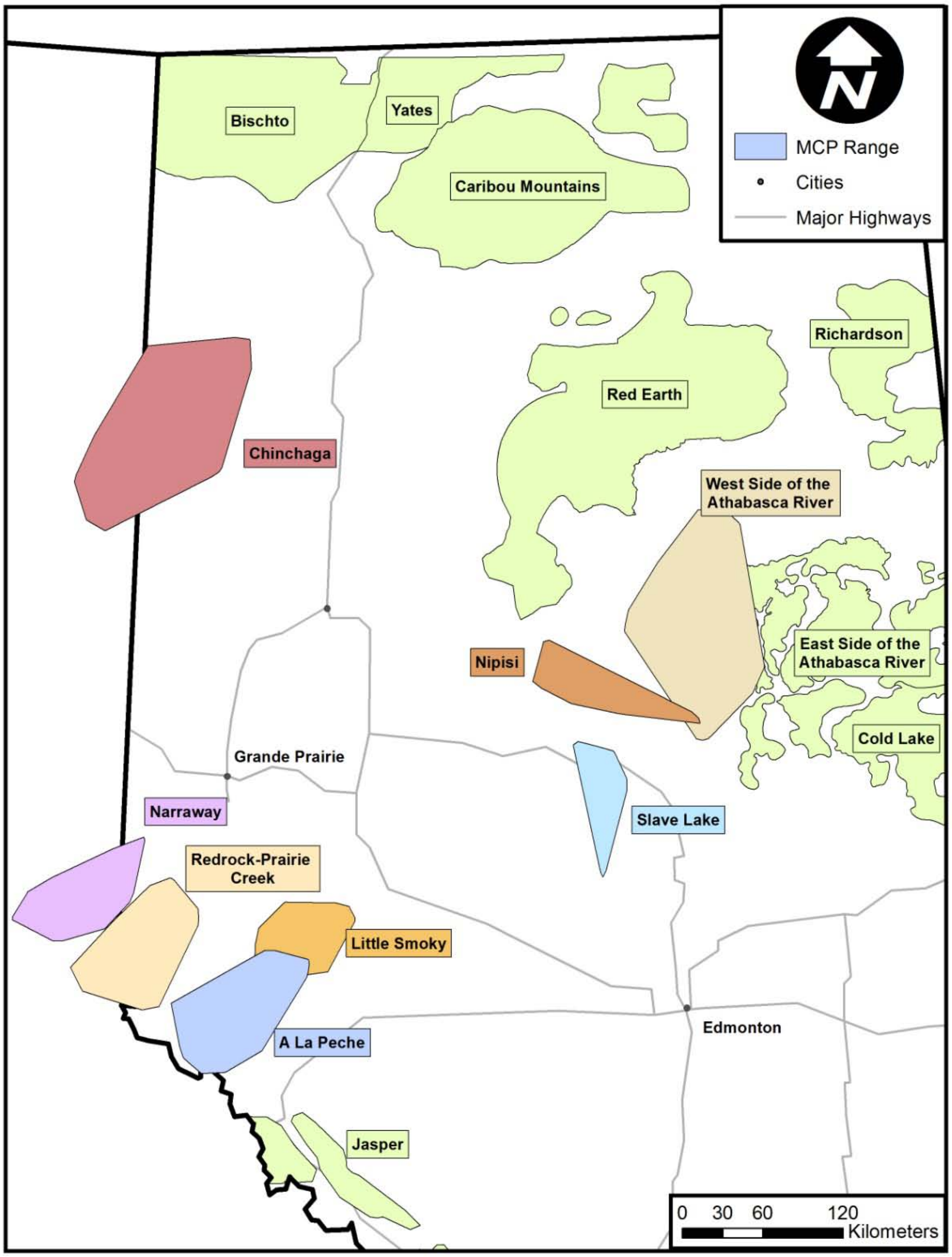


Figure 2.5: 95% Kernel Density Estimators (KDE) population ranges for both the boreal and mountain DUs. KDE areas represent the baseline condition for each population including all available animals and all available locations. Populations coloured in green were not included in the analysis.

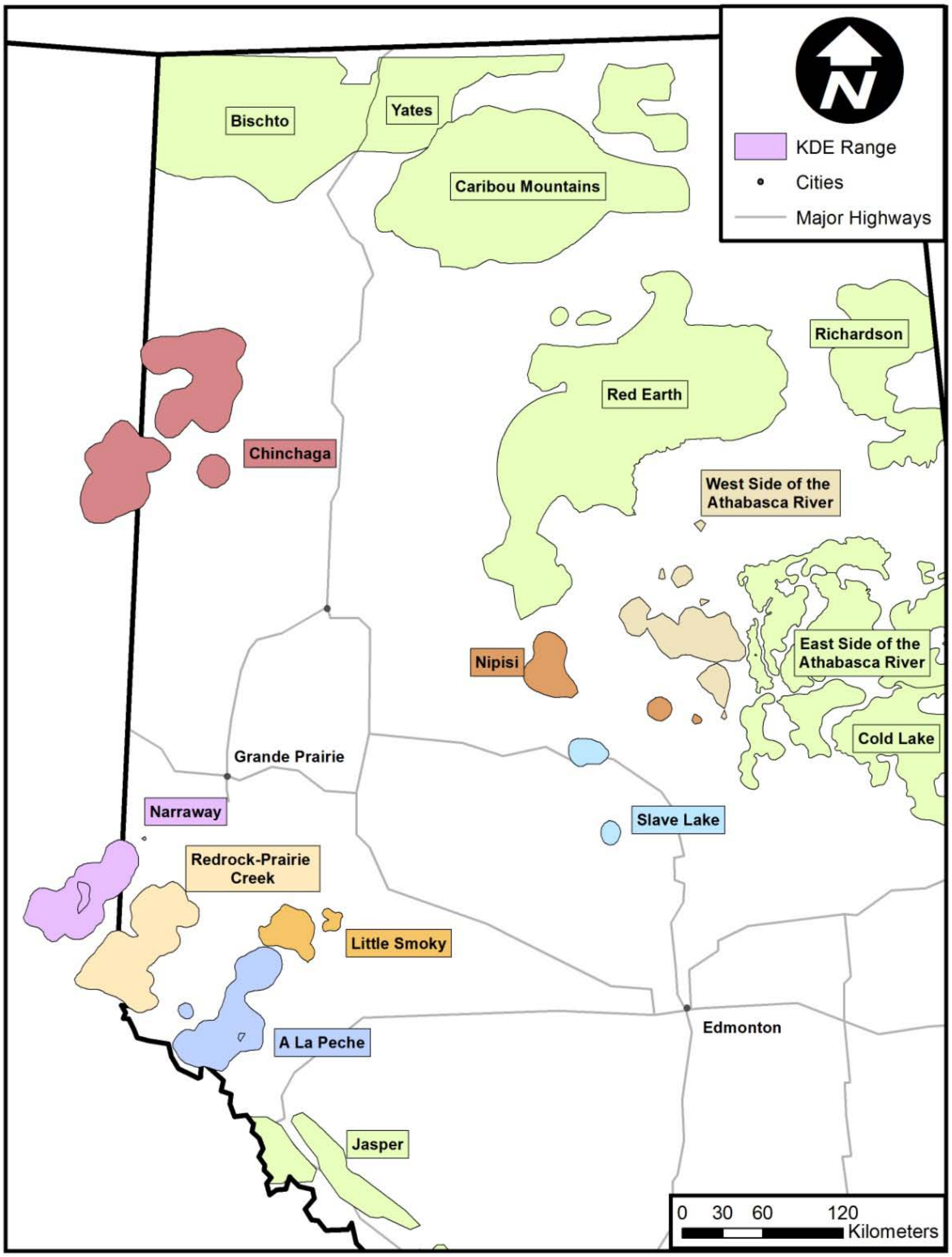
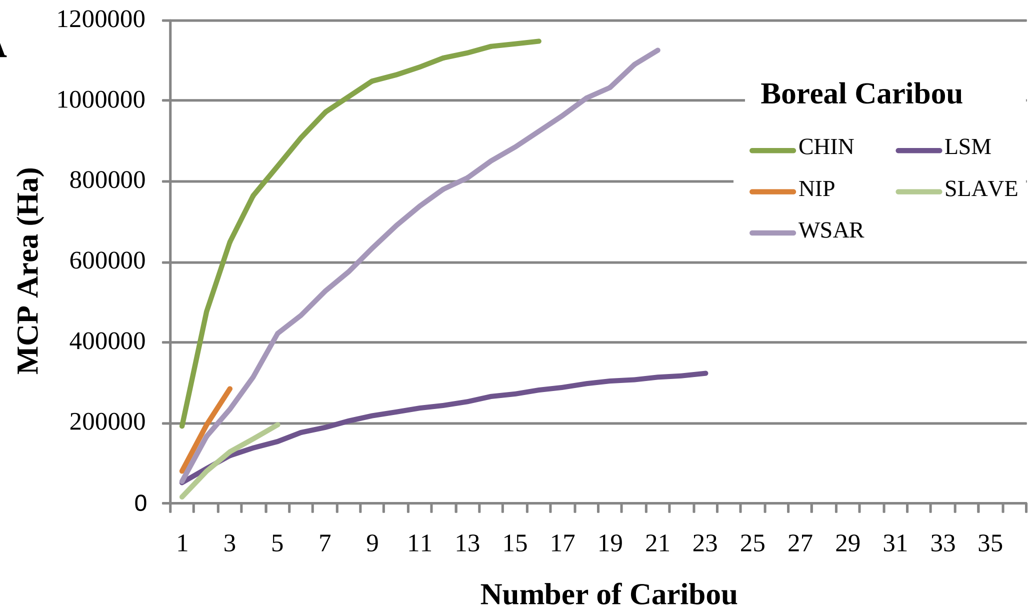


Figure 2.6: Population range results: Mean MCP area per number of caribou. Population range size represents the 1 location per day sampling frequency. A. Boreal caribou; B. Mountain caribou.

A



B

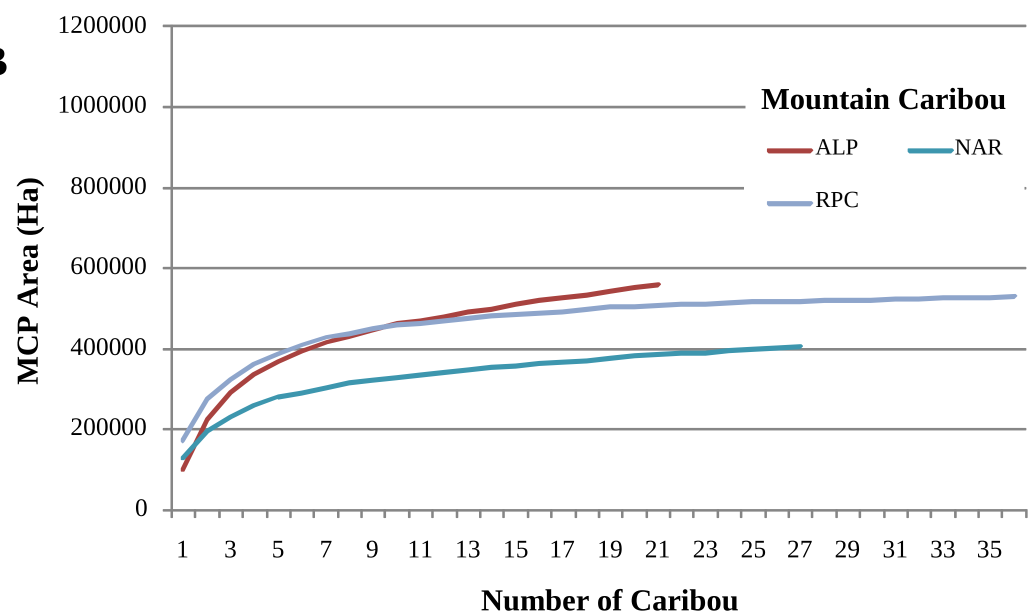
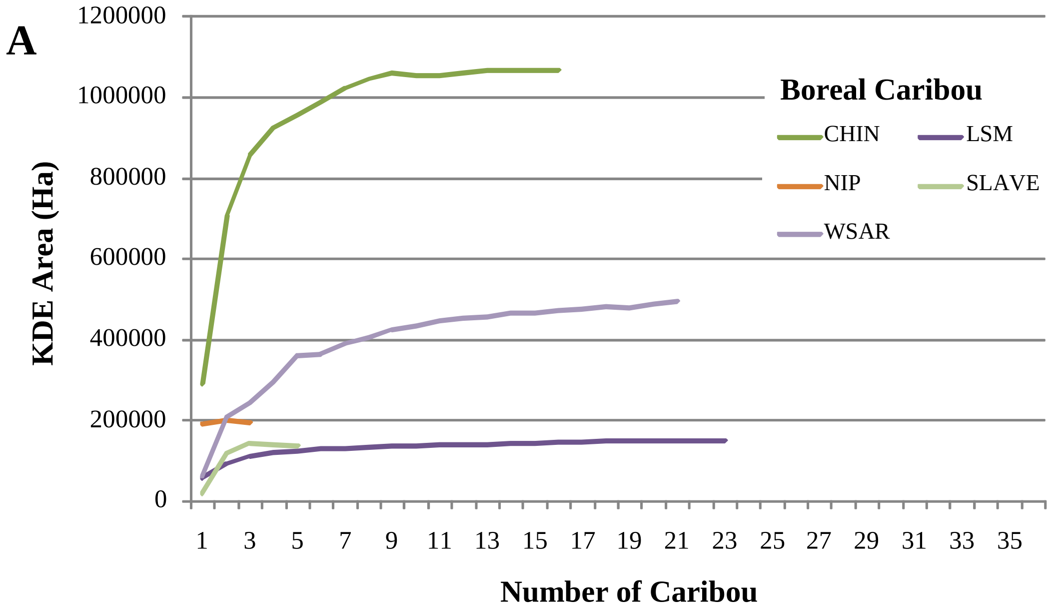


Figure 2.7: Population range results: Mean KDE area per number of caribou. Population range size represents the 1 location per day sampling frequency. A. Boreal caribou; B. Mountain caribou.



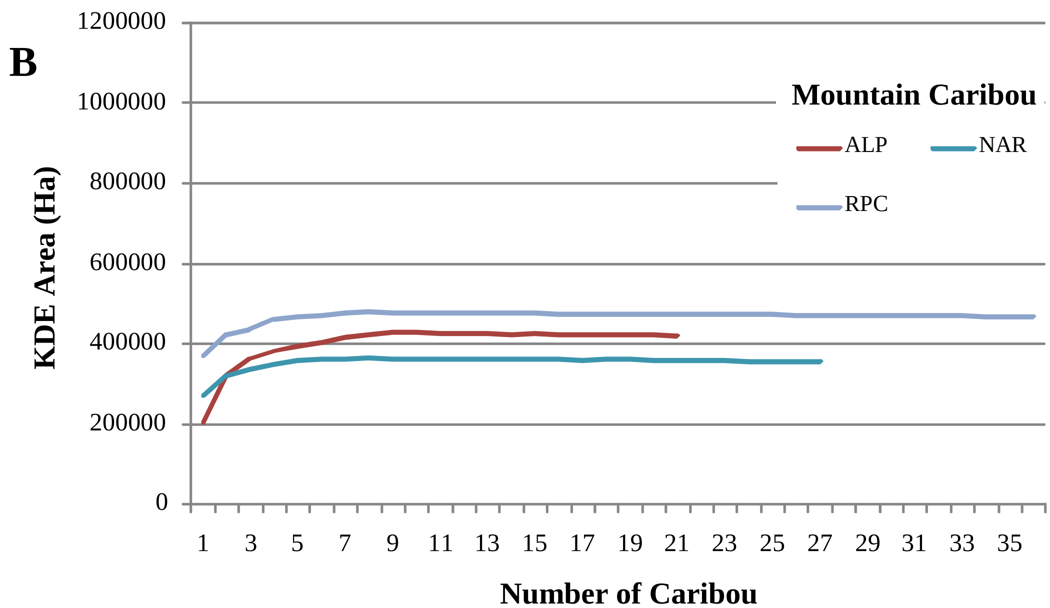


Figure 2.8: Population range results: Spatial representation of the Redrock Prairie Creek population range size for a selected number of caribou (1, 6, 11, 16, 21, 26, 31, and 36). Population range size represents the 1 location per day sampling frequency. A. 100% MCP Area; B. 95% KDE Area.

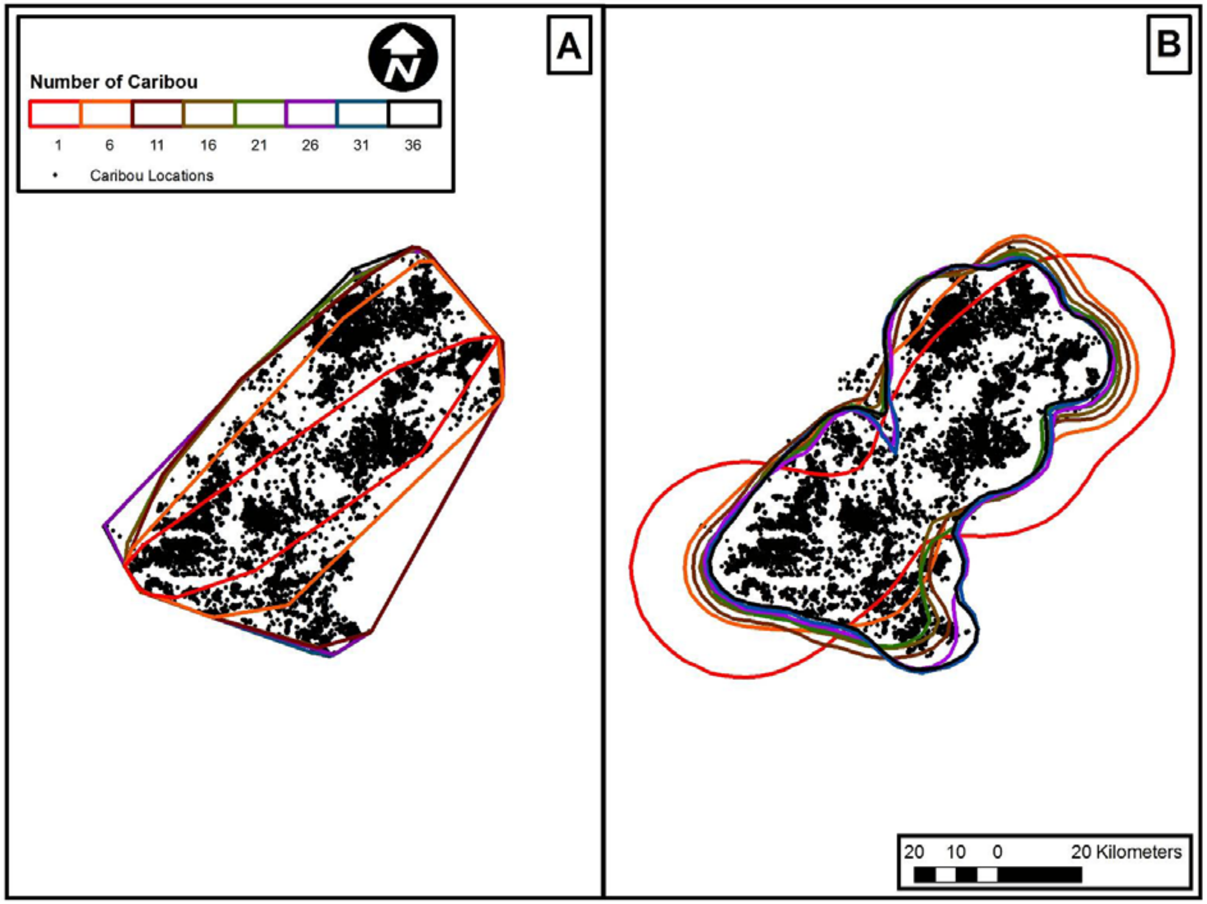


Figure 2.9: Population range results: Redrock Prairie Creek population range size per number of caribou for each of the six sampling frequencies (1 location per day, 4, 7, 14, 30, 60 days). A. Mean MCP Area; B. Mean KDE Area.

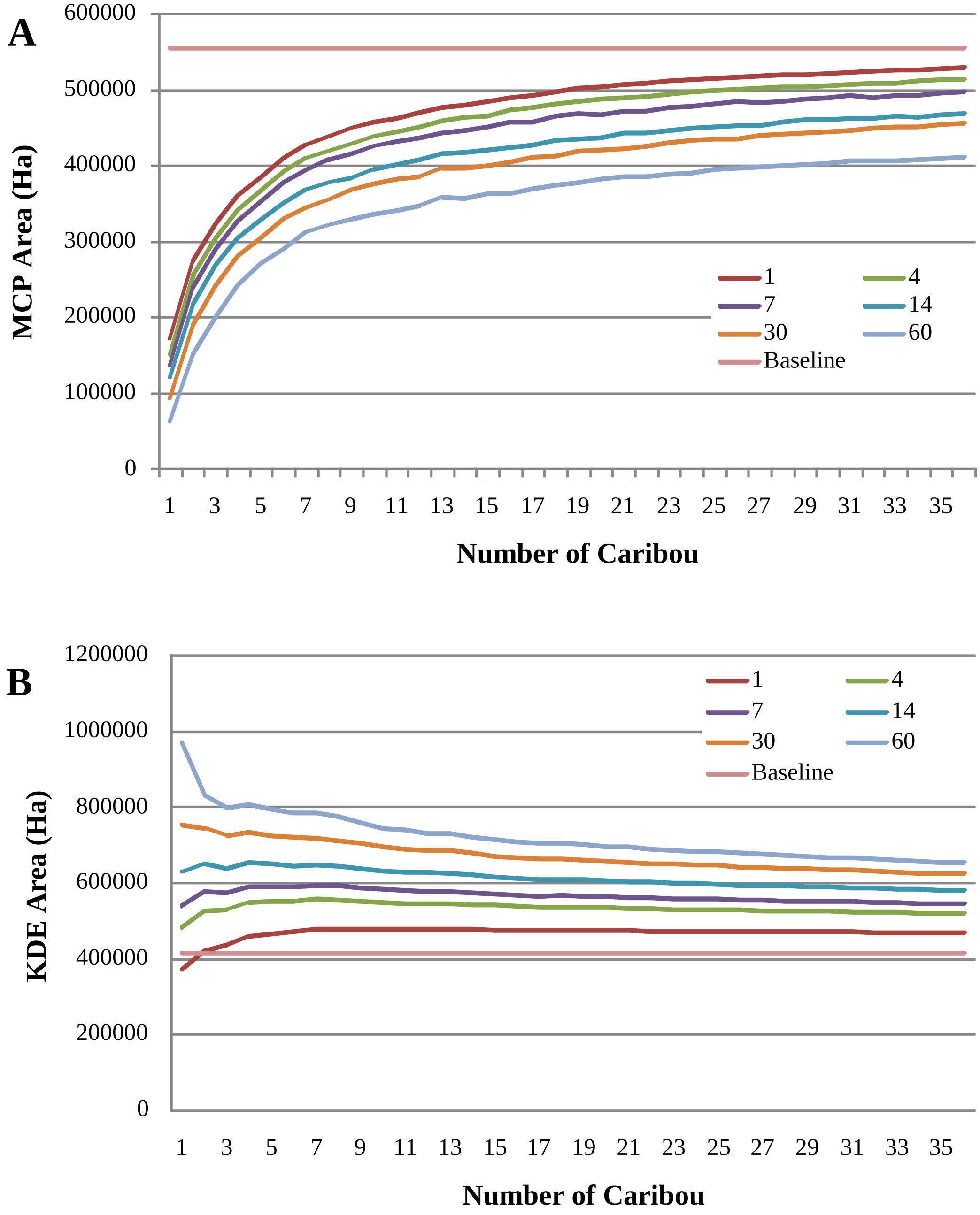


Figure 2.10: Population range results: Spatial representation of the Redrock Prairie Creek population range size for each of the six sampling frequencies (1 location per day, 4, 7, 14, 30, 60 days). A. 100% MCP Area; B. 95% KDE Area.

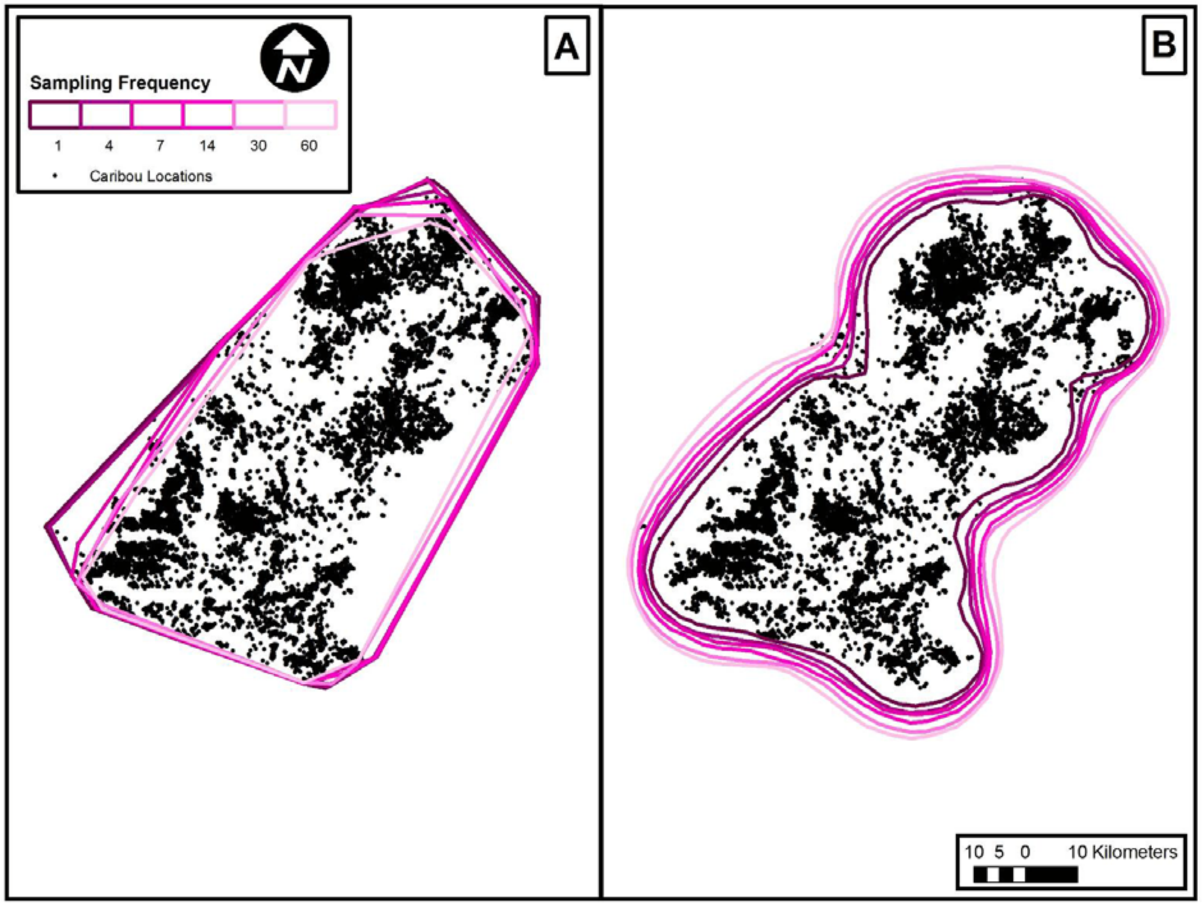


Figure 2.11: Population range results: Comparison of the baseline MCP and KDE population range areas with the jurisdictional area defined in the status of woodland caribou in Alberta update 2010 report (ASRD and ACA 2010). CHIN, NAR and RPC population range areas only represent the Alberta portion of the range.

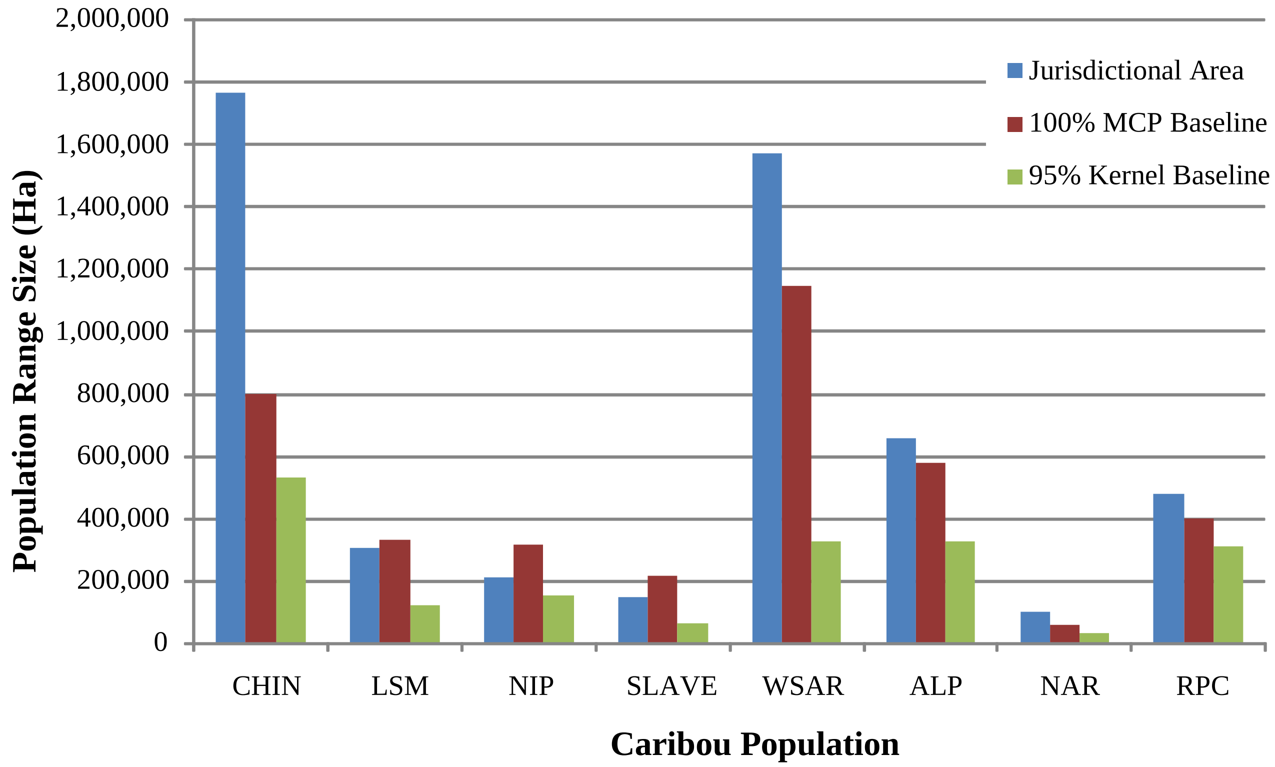


Figure 2.12: Population range results: Spatial representation of the Little Smoky population range size for a selected number of caribou (1, 3, 7, 11, 15, 19, 23). A. 100% MCP Area; B. 95% KDE Area.

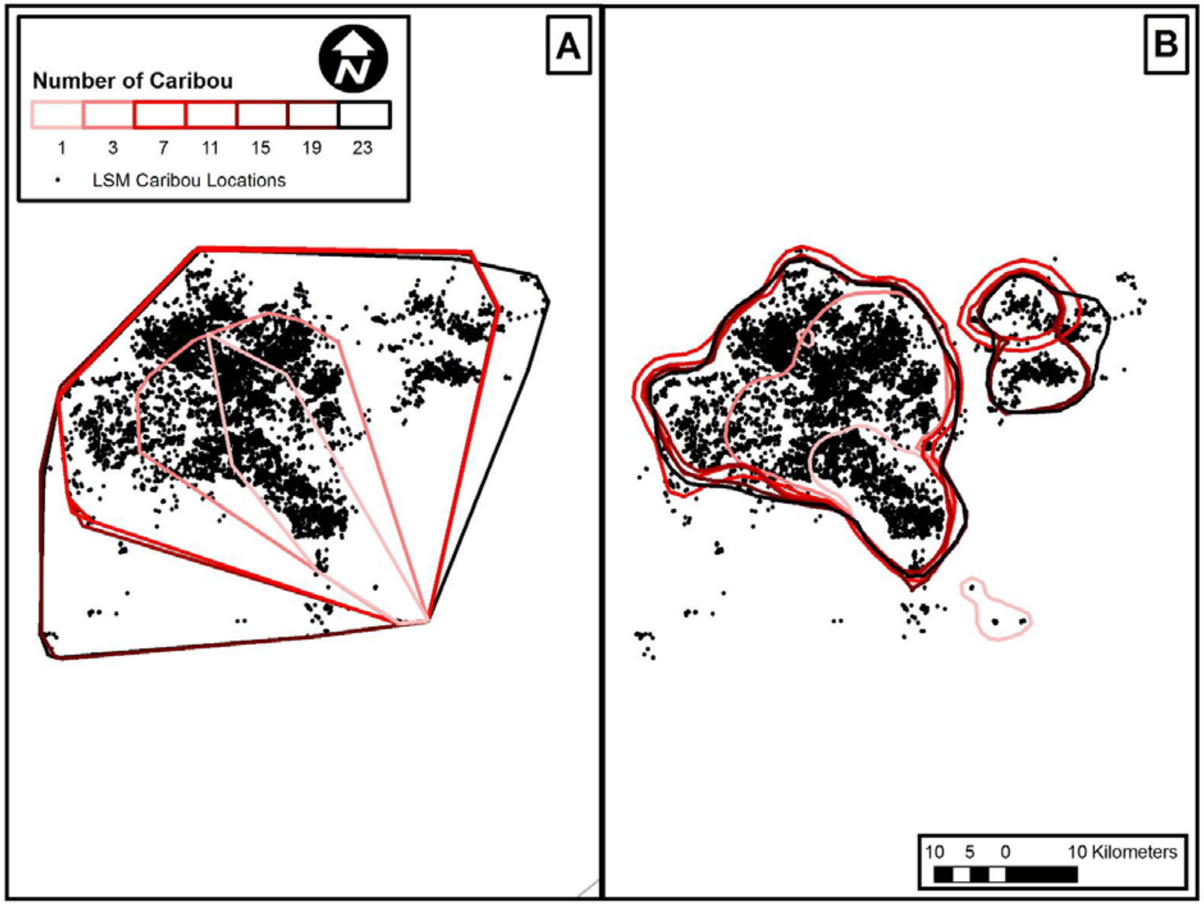


Figure 2.13: Population range results: Spatial representation of the Little Smoky population range size for each of the six sampling frequencies (1 location per day, 4, 7, 14, 30, 60 days). A. 100% MCP Area; B. 95% KDE Area.

7. Literature Cited

Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA). 2010. Status of the Woodland Caribou (*Rangifer tarandus caribou*) in Alberta: Update 2010., Alberta Sustainable Resource Development, Edmonton, AB.

Bethke, R., M. Taylor, S. Amstrup, and F. Messier. 1996. Population delineation of polar bears using satellite collar data. Ecological Applications 6:311-317.

Börger, L., N. Franconi, G. De Michele, A. Gantz, F. Meschi, A. Manica, S. Lovari, and T. Coulson. 2006. Effects of sampling regime on the mean and variance of home range size estimates. Journal of Animal Ecology 75:1393-1405.

Boulet, M., S. Couturier, S. D. Cote, R. D. Otto, and L. Bernatchez. 2007. Integrative use of spatial, genetic, and demographic analyses for investigating genetic connectivity between migratory, montane, and sedentary caribou herds. Molecular Ecology 16:4223-4240.

Brown, W. K. and D. P. Hobson. 1998. Caribou in west-central Alberta: information review and synthesis. Prepared for The Research Subcommittee of the West-central Alberta Caribou Standing Committee. Terrestrial & Aquatic Environmental Managers Ltd., Calgary.

Burt, W. H. 1943. Territorially and home range concepts as applied to Mammals. J. Mammal. Baltimoro 24:pp. 346-352.

Calenge, C. 2006. The package "adehabitat" for the R software: A tool for the analysis of space and habitat use by animals. Ecological Modelling 197:516-519.

Cooke, S. J. 2008. Biotelemetry and biologging in endangered species research and animal conservation: relevance to regional, national, and IUCN Red List threat assessments. Endangered Species Research 4:165-185.

COSEWIC. 2011. Designatable units for caribou (*Rangifer tarandus*) in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa. 88 pp.

D'Eon, R. G. and D. Delparte. 2005. Effects of radio-collar position and orientation on GPS radio-collar performance, and the implications of PDOP in data screening. Journal of Applied Ecology 42:383-388.

Dyer, S. J., J. P. O'Neill, S. M. Wasel, and S. Boutin. 2001. Avoidance of industrial development by woodland caribou. Journal of Wildlife Management 65:531-542.

Dzus, E. 2001. Status of the woodland caribou (*Rangifer tarandus caribou*) in Alberta. . Wildlife Status Report. Alberta Environment, Fisheries and Management Division, and Alberta Conservation Association.

Edmonds, E. J. 1988. Population status, distribution and movements of woodland caribou in west-central Alberta. Canadian Journal of Zoology-Revue Canadienne De Zoologie 66:817-826.

Edmonds, E. J. and M. Bloomfield. 1984. A study of woodland caribou (*Rangifer tarandus caribou*) in west-central Alberta, 1979-1983., Alberta Energy and Natural Resources Fish and Wildlife Division.

Environment Canada. 2008. Scientific Review for the Identification of Critical Habitat for Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population, in Canada., Environment Canada., Ottawa.

Environment Canada. 2011. Scientific Assessment to Inform the Identification of Critical Habitat for Woodland Caribou (*Rangifer tarandus caribou*), Boreal population, in Canada: 2011 Update. Ottawa, Ontario, Canada. 102 pp. plus appendices.

Fieberg, J. 2007. Kernel density estimators of home range: Smoothing and the autocorrelation red herring. Ecology 88:1059-1066.

Gautestad, A. O. and I. Mysterud. 1995. The home range ghost. Oikos 74:195-204.

Girard, I., C. Dussault, J. P. Ouellet, R. Courtois, and A. Caron. 2006. Balancing number of locations with number of individuals in telemetry studies. Journal of Wildlife Management 70:1249-1256.

Girard, I., J. P. Ouellet, R. Courtois, C. Dussault, and L. Breton. 2002. Effects of sampling effort based on GPS telemetry on home-range size estimations. Journal of Wildlife Management 66:1290-1300.

Hansteen, T. L., H. P. Andreassen, and R. A. Ims. 1997. Effects of spatiotemporal scale on autocorrelation and home range estimators. Journal of Wildlife Management 61:280-290.

Harris, S., W. J. Cresswell, P. G. Forde, W. J. Trewhella, T. Woollard, and S. Wray. 1990. Home-range analysis using radio-tracking data - A review of problems and techniques particularly as applied to the study of mammals. Mammal Review 20:97-123.

Hemson, G., P. Johnson, A. South, R. Kenward, R. Ripley, and D. Macdonald. 2005. Are kernels the mustard? Data from global positioning system (GPS) collars suggests problems for kernel home-range analyses with least-squares cross-validation. Journal of Animal Ecology 74:455-463.

Horne, J. S., E. O. Garton, and J. L. Rachlow. 2008. A synoptic model of animal space use: Simultaneous estimation of home range, habitat selection, and inter/intra-specific relationships. Ecological Modelling 214:338-348.

Johnson, C. J., D. R. Seip, and M. S. Boyce. 2004. A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. Journal of Applied Ecology 41:238-251.

Johnson, D. H. 1980. The Comparison of Usage and Availability Measurements for Evaluating Resource Preference. Ecology 61:65-71.

Joly, K. 2005. The effects of sampling regime on the analysis of movements of over-wintering female caribou in east-central Alaska. Rangifer 25:67-74.

Kenward, R. E. 2001. A Manual for Wildlife Radiotracking. Academic Press, London.

Kernohan, B. J., R. A. Gitzen, and J. J. Millspaugh. 2001. Analysis of animal space use and movements. Pages 126-166 *in* J. J. M. a. J. M. Marzluff, editor. Radiotracking and animal populations. Academic Press, San Diego, California, USA.

Kie, J. G., J. Matthiopoulos, J. Fieberg, R. A. Powell, F. Cagnacci, M. S. Mitchell, J. M. Gaillard, and P. R. Moorcroft. 2010. The home-range concept: are traditional estimators still relevant with modern telemetry technology? Philosophical Transactions of the Royal Society B-Biological Sciences 365:2221-2231.

Klaver, R. W., J. A. Jenks, C. S. Deperno, and S. L. Griffin. 2008. Associating seasonal range characteristics with survival of female white-tailed deer. Journal of Wildlife Management 72:343-353.

Laver, P. N. and M. J. Kelly. 2008. A critical review of home range studies. Journal of Wildlife Management 72:290-298.

Leban, F. A., M. J. Wisdom, E. O. Garton, B. K. Johnson, and J. G. Kie. 2001. Effect of sample size on the performance of resource selection analyses.

Pages 291-307 *in* J. J. Millspaugh and J. M. Marzluff, editors. Radio tracking and animal populations. Academic Press, New York, New York, USA.

Mohr, C. O. 1947. Table of Equivalent Populations of North American Small Mammals. American Midland Naturalist 37:223-249.

Natural Regions Committee. 2006. Natural Regions and Subregions of Alberta. Compiled by D.J. Downing and W.W. Pettapiece. Pub. No. T/852, Government of Alberta.

Ontario Woodland Caribou Recovery Team. 2008. Woodland Caribou (Rangifer tarandus caribou) (Forest-dwelling, Boreal Population) in Ontario. Ontario Ministry of Natural Resources, Peterborough, Ontario.

Otis, D. L. and G. C. White. 1999. Autocorrelation of location estimates and the analysis of radiotracking data. Journal of Wildlife Management 63:1039-1044.

Powell, R. A. 2000. Animal home ranges and territories and home range estimators. Pages 65-110 *in* L. Boitani and T. Fuller, editors. Research techniques in animal ecology: controversies and consequences. Columbia University Press, New York, New York, USA.

Racey, G. D. and A. A. Arsenault. 2007. In search of a critical habitat concept for woodland caribou, boreal population. Rangifer Special Issue 17:29-37.

Saher, D. J. 2005. Woodland caribou habitat selection during winter and along migratory routes in west-central Alberta. M.Sc. Thesis. University of Alberta, Edmonton, AB.

SARA. 2002. The Species At Risk Act (SARA), a guide. http://www.speciesatrisk.gc.ca.

Seaman, D. E., J. J. Millspaugh, B. J. Kernohan, G. C. Brundige, K. J. Raedeke, and R. A. Gitzen. 1999. Effects of sample size on kernel home range estimates. Journal of Wildlife Management 63:739-747.

Seaman, D. E. and R. A. Powell. 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. Ecology 77:2075-2085.

Swihart, R. K. and N. A. Slade. 1985. Testing for Independence of Observations in Animal Movements. Ecology 66:1176-1184.

Thomas, D. C. and D. R. Gray. 2002. Update COSEWIC status report on the woodland caribou Rangifer tarandus caribou in Canada, in COSEWIC assessment and update status report on the woodland caribou Rangifer tarandus caribou in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa.

White, G. C. and R. A. Garrott. 1990. Analysis of wildlife radio-tracking data. Academic Press, San Diego.

Worton, B. J. 1987. A Review of Models of Home Range for Animal Movement. Ecological Modelling 38:277-298.

Chapter 3: Woodland Caribou Winter Resource Selection In A Changing Landscape

1. Introduction

Understanding the impacts of landscape change on wildlife populations remains a challenge for their conservation and management (Houle et al. 2010). Both natural and anthropogenic disturbances alter landscape composition and configuration. However, given accelerating rate of development in many regions, recent efforts have focussed quantifying the effects of anthropogenic disturbance on wildlife species (Sawyer et al. 2009, Tracz et al. 2010, Hebblewhite 2011, Webb et al. 2011). Wildlife response to landscape change varies depending on the type of disturbance and the species. Ideally pre- and post-disturbance experiments are carried out to examine species-specific impacts, however it is logistically difficult and expensive to carry out such research, especially for wide ranging wildlife species (McLoughlin et al. 2011). An alternative approach involves examining patterns of resource selection by species across a gradient of landscape conditions (Harju et al. 2011, McLoughlin et al. 2011, Polfus et al. 2011, Moreau et al. 2012).

Resource selection occurs when resources are used disproportionately to availability, and as such is a fundamental process that structures animal distribution (Johnson 1980). Numerous approaches are available for mapping and predicting animal resource selection. Previous research has examined and summarised different resource selection methods (White and Garrott 1990, Alldredge and Ratti 1992, Erickson et al. 2001), categorized experimental designs (Thomas and Taylor 2006) and outlined common assumptions (Alldredge et al. 1998). More recently, Manly et al. (2002) introduced the concept of a resource selection function (RSF), where resource selection is modelled by a function of characteristics measured on resource units.

Resource selection functions are attractive because they can provide quantitative, spatially-explicit, predictive models for animal occurrence (Manly et al. 2002). With advancements in GPS telemetry technology and GIS (geographic information systems), RSFs are commonly used in wildlife biology as a framework for examining resource selection (Boyce et al. 2003, Johnson et al. 2004b, Gustine et al. 2006, Ciarniello et al. 2007, Mosnier et al. 2008). In addition, RSFs have been identified as an important tool for conservation and management planning because they are able to quantify important habitat resources for species of concern (Johnson et al. 2004b, Seip et al. 2007, Gustine and Parker 2008, Houle et al. 2010).

Despite these strengths, RSFs are considered static models and provide only a snapshot of resource selection and potential population distribution (Carroll et al. 2003). This is problematic, since a general objective of species conservation, and the specific goal of some species' recovery plans (Racey and Arsenault 2007), is to achieve the long-term persistence of self-sustaining wild populations, often in association with dynamic landscape conditions (i.e. given changing resources).

To date, most research has focused on how availability is defined for developing RSFs (McClean et al. 1998, Johnson and Gillingham 2008). However, a few studies have examined how resource selection changes as resource availability changes (Arthur et al. 1996, Alldredge and Griswold 2006, Moreau et al. 2012). Arthur et al. (1996) recognized that when habitat conditions change rapidly, defining availability on a shorter temporal scale (i.e. seasonally) would better represent the conditions present for resource selection. However, not only is it important to know how resource selection changes when temporal and spatial extent is varied, but also when natural and anthropogenic disturbances occur.

Johnson et al. (2004) acknowledged that RSFs should be considered in a temporal context, and suggested that changes in habitat availability might change the strength of selection for particular resources. For example, Johnson et al. (2004) found no effect of stand age on selection by caribou, but suggested this may have resulted from a prevalence of older forest in the current landscape, and postulated that selection might emerge if the amount of old forest in an area was reduced through natural or human disturbance. Examining whether resource selection changes with disturbance in dynamic landscapes (i.e. those where changes are occurring rapidly) requires temporal matching of animal location data and disturbance. Failure to accurately register the time of disturbance could result in misleading models of response. For example, a naïve approximation of landscape condition over multiple years could fail to detect avoidance response if disturbances were not present for much of the time over which animal use was measured.

Resource selection is a hierarchical process, with wildlife responding to different habitat components across a range of spatial scales (Johnson 1980, Hall et al. 1997). For example, forage requirements are more likely to influence resource selection at finer (within home range) scales, whereas predation and other population processes that operate at larger scales are more likely to have an effect on resource selection at coarser (landscape) scales (Rettie and Messier 2000; Boyce 2006). Drivers of landscape change may thus affect selection at different spatial scales, and therefore consideration of multiple scales is required to characterize scale-dependent habitat relationships (DeCesare et al. 2012).

Understanding how populations are likely to respond to landscape dynamics involving both natural and anthropogenic processes is critical to making informed conservation and management decisions. Such knowledge is urgently needed in Alberta, Canada, where anthropogenic disturbance driven by resource exploration and development has rapidly increased in recent decades (Schindler and Lee 2010). Effects on woodland caribou (*Rangifer tarandus caribou*), listed as a threatened species under both the Alberta Wildlife Act and the federal Species At Risk Act (SARA), are of particular concern. In Alberta, caribou have been extirpated from approximately 60% of their historical extent of occurrence (Festa-Bianchet et al. 2011). Numerous studies have shown that caribou are sensitive to various anthropogenic disturbances (e.g., Johnson et al. 2004b, Dunford et al. 2006, Vors et al. 2007), with reviews suggesting that habitat alterations by anthropogenic activities are the ultimate cause for woodland caribou decline (Vistnes and Nellemann 2008, Vors and Boyce 2009, Festa-Bianchet et al. 2011).

The main anthropogenic disturbances on caribou population ranges in Alberta are forest harvesting and oil and gas exploration and development. Habitat alteration associated with industrial development may have negative implications for caribou due to increased disturbance, human caused mortality, changes in other prey species, and predation (James and Stuart-Smith 2000). Predation by wolves (*Canis lupus*) is regarded as the proximate cause of caribou decline across Canada (Bergerud and Elliot 1986, Thomas and Gray 2002). Caribou are thought to avoid predators through spatial separation; that is, by selecting habitats where primary prey and their predators are less likely to occur (Bergerud and Page 1987, Seip 1992; James et al. 2004, Neufeld 2006). Disturbance caused by forest harvesting negatively affects caribou and can result in avoidance up to 12 km (Smith et al. 2000, James et al. 2004, Schaefer and Mahoney 2007, Hins et al. 2009, Houle et al. 2010). Similarly, caribou may avoid habitat alterations such as wellsites, roads, pipelines and seismic lines, caused by oil and gas sector activities, (Dyer et al. 2001, Saher 2005, Neufeld 2006). If habitat alteration becomes extensive, caribou may respond by shifting their range to adjacent areas with suitable habitat, if they exist (Vistnes and Nellemann 2008).

Vors et al. (2007) reported a permanent abandonment of harvested areas after 20 years by caribou in the southern portion of their range in boreal forest landscapes of Ontario. In west central Alberta, a caribou population appears to have abandoned their traditional winter range in response to forest harvesting (Smith 2004). However, another Alberta study found no evidence that caribou had altered either their annual or monthly ranges in response to oil and gas sector activities, at least in the short term (Tracz et al. 2010). Increased understanding of apparent variability in responses may be gained by examining patterns across a hierarchy of scales. Hereafter, I refer to second-order (landscape) and third-order (within home range) scales as coarse and fine scale, respectively, following Johnson (1980).

In this chapter, I evaluate how woodland caribou respond to changing resource availability by examining population level resource selection over a 13-year time-frame, in a landscape subject to forestry and energy sector activities, and wildfire. My specific objectives include: 1) quantifying coarse and fine-scale caribou resource selection on the winter range of the Redrock Prairie Creek population from 1998 to 2011; 2) investigating the response of caribou to landscape change due to anthropogenic and natural disturbance by examining resource selection on an annual basis; and 3) examining whether caribou shift their range use in response to disturbance, or resume use of previously disturbed areas over time (i.e. whether habitat recovery occurs). I predicted that accounting for changes in resource availability by matching annual disturbance layers with caribou location (use) data would result in a clearer depiction of resource selection and thus stronger RSF models. I also explore a method to account for variability over time in resource selection studies and provide recommendations for future development of RSFs for conservation and management strategies.

2. Study Area

The Redrock Prairie Creek (RPC) caribou winter range in west central Alberta and east central BC (54^{o}N, 119^{o}W) encompasses approximately 5200 km^2, extending north of the Kakwa River in the foothills to the Jackpine river to the south in Alberta, and including areas of the Rocky Mountains in British Columbia (Figure 3.1). RPC caribou occupy areas included in the Lower and Upper Foothills, Subalpine, and Alpine Natural Subregions of Alberta (Natural Regions Committee 2006). The Foothills subregion consists of upland areas characterized primarily by lodgepole pine (*Pinus contorta*) or lodgepole pine/white spruce (*Picea glauca*) forests with small patches of trembling aspen (*Populus tremuloides*) and lowland areas with poor drainage characterized by black spruce (*P. mariana*) and larch (*Larix laricina*). At higher elevations, in the Subalpine subregion is dominated by Engelmann spruce (*P. engelmannii*) and subalpine fir (*Abies lasiocarpa*), while the Alpine subregion has few trees and is characterized by graminoids, sedges (*Carex spp.*) and wind-swept ridges.

In addition to woodland caribou, moose (*Alces alces*), and elk (*Cervus elaphus*) are present and the most abundant ungulates, with smaller numbers of mule (*Odocoileus hemionus*) and white-tailed (*O. virginianus*) deer found in the area. Large carnivores that inhabit the caribou range include coyote (*Canis latrans*), wolves (*C. lupus*), cougars (*Felis concolor*), grizzly (*Ursus arctos*) and black (*U. americanus*) bears.

Various land use activities occur within the RPC winter range. Forest harvesting began in 1976 in the Alberta foothills, and in 1987 in the east central BC portion of the range. Oil and gas exploration in the area dates back to the

1950’s and development significantly increased in the 2000’s. Coal mining has been a feature of this landscape since 1969, with some expansion in the 1990’s. In general, industrial activity is concentrated in the Alberta foothills portion of the study area.

3. Methods

3.1 Caribou Location Data

From 1998-2011, in collaboration with Weyerhaeuser Company Ltd. Grande Prairie, the University of Alberta and Alberta Sustainable Resource Development, adult female caribou were captured using helicopter net-gunning techniques and fitted with GPS telemetry collars (Lotek GPS1000, 2000, 2200, 3300, 4400 and Iridium models; Lotek Engineering, Newmarket, Ontario, Canada). Capture protocols were approved by University of Alberta Animal Care Committees (FAPWC Protocol No. 99-75D; ACUC Protocol No. 731910). Between 1998 and 2011, GPS collars were deployed on 81 individual caribou from the RPC caribou population, resulting in 259,546 data locations.

Since the GPS location data were collected over multiple years and for different research and government projects, the tracking schedule of the collars deployed varies from a location every 15 minutes to every 24 hours, with an average GPS tracking schedule of one location every 4 hours. GPS collars with differential data were corrected using N4win (Lotek Wireless Inc. 1999). All locations with a horizontal dilution of precision (HDOP) greater than 12, indicating erroneous location accuracy, were removed prior to analysis (D'Eon and Delparte 2005). Caribou locations were also spatially mapped in geographic information system (GIS; ArcGIS 10, Environmental Systems Research Institute (ESRI), Redlands, CA, USA) and visual checks were used to identify and remove any additional erroneous locations. Erroneous locations were classified as those clearly outside the study area or involving extremely large movements between fixes (e.g. a location 20 km away between fixes).

Locations were further filtered and subsampled to 4-hour location intervals, and animals with less than a month of data over the winter season were excluded from analyses. To maintain consistency with previous work conducted in the region (Smith et al. 2000, Saher 2005) the winter season is defined here as December 1 - April 30 of each year. The winter season is important for caribou as it tends to be when forage is most limiting (Wittmer et al. 2006), and for the RPC population, most of the anthropogenic disturbance is concentrated in the winter range (Smith et al. 2000), and this period is thus of greatest management concern. Early spring or late fall migrations, identified by examining individual movements, were also removed from analyses as the locations do not represent typical winter movement patterns. The final number of animals and locations included in analyses was 79 and 79,359, respectively (Table 3.1, Figure 3.2).

GPS technology provides accurate animal locations <31m 95% of the time under boreal forest canopies (Rempel et al. 1995, D'Eon et al. 2002); however, missed or failed location attempts can result in biases in habitat selection studies if they occur consistently in the same habitats (Frair et al. 2004, Hebblewhite et al. 2007, Frair et al. 2010). GPS fix rate for my study area averaged 86% across individuals. However, sample weighting to correct for potential habitat biases cannot be applied to individual observations (Frair et al. 2010), and was thus not appropriate to the analytical approach I employed (conditional fixed effects logistic regression).

3.2 Resource Selection Function Modelling

I developed RSFs for woodland caribou to examine habitat use in the Redrock Prairie Creek winter range at second-order (landscape) and third-order (within home range) scales (Johnson 1980). Following Garshelis (2000), I define habitat as the set of specific environmental conditions associated with animal use and define habitat use as expression of the extent to which areas with different environmental conditions are used. I developed RSFs based on a use-available design (Manly et al. 2002), where resource covariates at a particular GPS location for a caribou are compared to the same covariates at randomly selected locations within the area of interest. Assumptions for the Manly et al. (2002) use-available design include: 1) animals are sampled randomly and assumed to be representative of the population; 2) relocations are independent of one another; 3) selection of a resource is independent of selections by other animals; 4) availability is the same for all animals; 5) availability is known and constant over the study period; and 6) resources are classified correctly.

3.2.1 Second-order scale

I estimated RSFs at the second-order scale by sampling at a 1:1 ratio of used to random available locations within the total population range for all GPS collared caribou (Johnson et al. 2006). The population range was estimated using the minimum convex polygon method (100% MCP, Mohr 1947). To account for unbalanced sample sizes among caribou and spatial and temporal autocorrelation of data within individual caribou, I used generalized linear mixed models (GLMM) with a random intercept for each animal ($\beta_0 + \gamma_{0j}$; Hebblewhite and Merrill 2008). I fitted mixed-effect models for location (*i*) and individual caribou (*j*) with a random intercept in the form:

$$w^*(x)_{ij} = \exp(\beta_0 + \gamma_{0j} + \beta_1 x_{1ij} + ... + \beta_n x_{nij}) \qquad [1]$$

where w^*(x) is relative probability of use, $\beta_{1\ldots n}$ are the estimated coefficients for covariate x_n, and γ_{0j} is the random per-individual intercept (Gillies et al. 2006).

3.2.2 Third-order scale

At the third-order scale, I used matched-case control logistic regression (conditional logistic regression) to estimate the relative probability of caribou selection from one time step to the next. I sampled at a 1:1 ratio of used to available locations where random available locations are generated within a circle centered on the use location with a radius equal to the 95th percentile of the daily distance traveled by all GPS collared caribou (Arthur et al. 1996; Geospatial Modelling Environment v 0.5.5, www.spatialecology.com). To account for unequal sample sizes between animals, I used sample weighting to give equal weight to each animal using the inverse of the probability that an individual caribou was included in the sample (Alldredge et al. 1998, Polfus et al. 2011). Statistical analyses were carried out in STATA 12.0 (StataCorp 2011).

3.2.3 Habitat covariates

I included resource covariates in my analysis based on their relevance to caribou habitat use, and as indicated from previous studies in the region (Table

3.2: Oberg 2001, Szkorupa and Schmiegelow 2003, Saher 2005). All covariates were screened for collinearity using Pearson's |r| > 0.7 as the threshold for considering removal of correlated variables, which were not included in the same model (Saher 2005, DeCesare et al. 2012). To characterize the anthropogenic features (cutblocks, linear features (seismic lines, roads and pipelines), and wellsites), I determined the density of each feature using a circular moving window with a radius of 70 m (3rd order scale) and 5 km (2nd order scale), reflecting recent regional analyses that found that caribou response to anthropogenic features was strongest when measured at these radii (DeCesare et al. 2012). Total anthropogenic disturbance was calculated by the proportion of area within the 70 m and 5 km moving window, where linear features were 25 m and seismic lines were 10 m to approximate their physical footprint on the landscape.

Habitat covariates necessary for caribou coarse-scale modeling were derived from digital elevation models, digital forest inventory and other spatial data using ArcGIS 10 (ESRI, Redlands, CA, USA). Landcover data derived from Landsat imagery were acquired from the University of Montana, Foothills Research Institute (FRI) and the University of Calgary (see DeCesare et al. 2012). Disturbance data were acquired from Alberta Sustainable Resource Development (D. Stepnisky) and Weyerhaeuser Company Limited (W. Crosina). A pixel resolution of 30 m was used for environmental and forest inventory data, to account for potential location inaccuracies associated with GPS location data (D'Eon et al. 2002) and maintain consistency with previous development of RSFs in the region (Saher and Schmiegelow 2005). Density calculations were made using the Spatial Analyst extension in ArcGIS 10. To incorporate a climate covariate, I derived a percent snow cover layer from 8-day composites of maximum snow extent maps at a 500m resolution produced by MODIS satellites (MOD10A2; Hall et al. 2006, DeCesare et al. 2012). Spatial models of percent snow cover were derived from the number of days snow occupied a cell, divided by the number of days data were available for each winter season.

3.2.4 Development of temporally matched landscape layers

In order to examine caribou response to landscape change I developed annual disturbance layers. Temporal attributes for cutblock and wildfire disturbance are well documented, however information regarding oil and gas exploration and development is difficult to acquire. The Alberta Government and industry rely on the Digital Integrated Disposition (DIDs) mapping database maintained by AltaLIS (http://www.altalis.com) for current activities on public lands as they are approved. Unfortunately this database does not include a date of when the disturbance occurred on the ground. However, the wellsite GIS layer obtained from ASRD includes a Spud Date which represents the actual drilling date of each well. I compared the Spud Date with the DIDs approval date and found that 89% of the wellsites were drilled within a year of the approval date. Therefore I used the DIDs approval date to time stamp the road and pipeline layers.

Seismic exploration is not included in the DIDs mapping database, so an alternative approach was required. Previous research in the RPC area showed that in 1998, >80% of the seismic lines were >23 years old (Smith 2004). According to the ASRD database, the number of kilometers of seismic lines digitized before 1998 was 4,048 km with another 697 km of new seismic lines digitized after 1998, which confirms that the bulk of seismic exploration in the RPC winter range occurred before 1998. Due to the lack of disturbance date for seismic features, I created a base ASRD seismic layer for pre 2005 data and an updated layer for post 2005.

I compiled annual disturbance layers from 1998 to 2011 for cutblocks, wildfires, linear features (roads and pipelines), and wellsites. I supplemented these data with two seismic data layers, the base layer from 1998 to 2005 and the updated layer used for 2006 to 2010 (Figure 3.3). Total density represents cumulative anthropogenic disturbance, including cutblocks, roads, pipelines, seismic lines and wellsites. Disturbance density covariates were transformed (ln_tden = ln(tden + 0.01)) to normalize the data.

3.2.5 Model Selection

I used a multiple working hypotheses paradigm (Anderson et al. 2000) to develop a set of a priori candidate models. Candidate models were developed based on previous research and hypotheses about caribou resource selection. The candidate set included variable formulations of disturbance only, landscape only, and a combination of disturbance and landscape variables for a total of 8 and 10 candidates for second- and third-order models, respectively (Table 3.3). A candidate set of models was developed for each year using temporally matched data, whereas the global-static set included all years of caribou data and was naïve to landscape change (i.e., a single covariate layer corresponding to the last year of caribou data collection was used). When models included the landcover type variable, I set the most abundant landcover type (closed conifer forest) as the reference category. Model selection followed an information theoretic approach for small sample sizes, AIC_c (Akaike's Information Criteria; Burnham and Anderson 2002).

$$AIC_c = -2\log_e(L(\hat{\theta}) + 2K + \frac{2K(K+1)}{(n-K-1)} \qquad [2]$$

where $\log_e(L(\hat{\theta} \mid data))$ is the value of the maximized log likelihood over the unknown parameters (θ), given the data and the model, K is the number of estimable parameters in that approximating model (Burnham and Anderson 2002), and n is the number of caribou. Models were then ranked based on the difference in AIC values (Δ_i) to select the top model (Manly et al. 2002). Akaike weights (w_i) were used to assess the strength of evidence that a particular model was the best of those in the candidate set (Burnham and Anderson 2002).

3.2.6 Model Validation

Models were mapped in ArcGIS 10.0 at a 30 m resolution, by standardizing the RSF values between 0 (low) and 1 (high) with respect to the relative probability of caribou use (Manly et al. 2002, Hebblewhite and Merrill 2008).

$$RSF\ scaled\ = \frac{(RSF - minimum_{value})}{maximum_{value} - minimum_{value}} \qquad [3]$$

Model fit was evaluated using a *k*-folds cross-validation, which measures the predictive capacity of the RSF model (Boyce et al. 2002). After fitting and identifying the best model, I used a testing to training ratio of 1:5, to partition the data into five groups. The model was trained iteratively on four groups, retaining the fifth for testing. I validated each model by comparing the ranked bins of the predicted RSF values using a Spearman rank correlation statistic (r_s). The average Spearman rank correlation statistic is a measure of the within sample predictive ability of the model (Fielding & Bell 1997; Boyce et al. 2002).

3.2.7 Global-dynamic model

Since the global-static model is naive to landscape change, and I expected annual RSF models to vary, I developed a global-dynamic RSF model that accounts for a changing landscape. Similar to the annual models, use and available locations were matched with the annual spatial data and merged into a final global-dynamic database that included all animals and all years.

3.3 Assessment of resource selection change over time

I examined change in resource selection between the global models (static and dynamic) and the annual models in two ways. First, I examined how the model selection process varied annually by examining how the top model varied between annual model and global model selections. Second, I plotted the model coefficients and approximate 95% confidence intervals for each coefficient. I reported “significant” effects when the confidence interval bars in the annual models did not overlap with confidence interval bars generated using the global model (Johnson and Gillingham 2008). In addition, in cases where the coefficient also changed sign, I concluded that the resource selection in that year was extremely different.

3.4 Evaluation of caribou winter range use over time

Using 13 years of caribou location data, I created annual and global 95% fixed kernel density estimates (KDE: h_{ref} smoothing parameter) for the RPC winter range. To examine RPC population spatial use from 1998 to 2011, I created a polygon to represent the historical core area (1981-1998) following Edmonds and Bloomfield (1984; Figure A.4) and Brown and Hobson (1998; Figure A.5). I defined the area south of the core area as mountain winter range. I quantified the proportion of each annual and global winter range that fell within the core and mountain areas of the RPC winter range (ArcGIS10 ESRI, Redlands, CA, USA), and examined the proportion of predicted high use areas within the core range from the 2nd order annual prediction maps. I divided the relative probability of use into quartiles and assigned the top quartile (76-100%) as high use area.

4. Results

4.1 Disturbance data

Disturbance in the RPC winter range occurs in the form of forest harvesting (cutblocks, roads), oil and gas exploration and development (pipelines, roads, seismic, wellsites) and natural disturbance (wildfires). The cumulative area of anthropogenic disturbance increased from 25,357 ha in 1998 to 44,587 ha in 2010 (Figure 3.4). Wildfire data in the RPC winter range date back to 1938, and the area of wildfire disturbance varied from 11,535 ha in 1998 to 34,405 ha in 2010, with 2 large fires in 2006 accounting for 20,132 ha (59%) of the area burned in the winter range (Table 3.4). On an annual basis, the amount of disturbance within the occupied winter range in a given year ranged from 3,657 ha in 2003 to 12,306 ha in 2009, representing between 1.7% and 5.0% of the range area, respectively.

4.2 Second-order resource selection

At a coarse scale, wellsite density and linear density (r = 0.80) were correlated, and total density was highly correlated with cutblock density (r = 0.95; Table 3.5). The most complex model, Model 8 (M8), was identified as the best model in all years except 2000, 2002, and 2006, where M7 was the top model (Table 3.6; Table A.2). Similar to the annual models, M8 was also the top global-static and global-dynamic model. The annual winter models validated well, indicating a high predictive capacity within a year (average r_s of 0.81, SD = 0.075; Table 3.10). The global-dynamic model had a higher Spearman rank than the global-static model, suggesting it was a better predictive model across all years (r_s = 0.82 vs r_s = 0.67; Table 3.10).

At a landscape level, across the winter range, caribou exhibited annual differences in response to disturbance, as well as to landcover covariates (Table 3.8). The strength of selection in the annual models was generally greater compared to the global static model, but less so for the global dynamic model based on coefficient values. In most years, caribou were negatively associated with areas of high cutblock density (exception 1999 and 2010; Figure 3.5). Response to linear density was more variable; caribou avoided areas with higher linear density in 5 of 13 years, and selected them in 8 of 13 years (Figure 3.5).

When M7 was the best model (2000, 2002, 2006), caribou selected areas with higher total disturbance density (Table 3.8; Figure 3.5). The top global-static model (M8) was consistent in part with the annual models. Essentially, caribou avoided areas of high cutblock density and showed weak selection for areas of high linear feature density. The top global-dynamic model (M8) indicated that caribou avoid areas of high cutblock and linear density.

Response to landcover covariates varied among annual models; however, the overall trend suggests that caribou select alpine, open conifer and shrub areas, and avoid burns, cutblocks, deciduous, herbaceous, mixed forest, rock and ice, water, wetlands and areas with higher snow cover (Table 3.8). The global models generally showed similar responses to landcover covariates; however, the global-static model suggested selection of burnt areas. For all models (annual and global), the topographic variables of elevation, slope and topographic position index all had small coefficient values (near zero), generally indicating that caribou are using these features proportionately to what is available within the winter range (i.e. no apparent selection).

4.3 Third-order resource selection

At a fine scale, total disturbance density was correlated with cutblock density (r = 0.78; Table 3.5), so these variables were not included in the same model. Two additional candidate models were added to the candidate set evaluating fine scale model selection, as the variables describing seismic and wellsite densities were not significantly correlated to the other disturbance variables. I measured selection by sampling availability within a 6.3 km buffer around used locations, representing the 95th percentile of daily movement distance over the winter season.

For the majority of the annual resource selection models, M10 (which included seismic line and wellsite density as additional covariates) was selected as the best model, with the exception of 1998, 2001, 2004, 2005, where M8 was selected (Table 3.7; Appendix A.3). The annual winter conditional logistic regression models had relatively high predictive performance (average $r_s = 0.91$, SD = 0.087; Table 3.10). The global-dynamic model had a slightly higher Spearman rank than the global-static model ($r_s = 0.81$ vs $r_s = 0.77$; Table 3.10), but it was still a poorer predictor, on average, than the annual models.

At a local level, caribou showed significant avoidance of areas with high cutblock, linear, wellsite and seismic densities in most years, and also in the global-dynamic model (Table 3.9). However, the global-static model suggested no response to wellsites, and much weaker response to linear density (Table 3.9). Caribou occurrence was positively related to alpine, open conifer, and shrub cover, and negatively related to burn, cutblocks, deciduous, herbaceous, mixed forest, water, and wetland. The topographic covariates had small coefficient values in all models. Similar to the second-order models, the annual models generally exhibited stronger coefficient values compared to the global models, although somewhat less so for the dynamic one (Table 3.9, Figure 3.8). Response to landcover covariates showed high annual variability; in particular, response to cutblocks. Coefficient values ranged from highly negative (-14.34) to highly positive (12.26); however, the variable was not significant in either of the global models (Table 3.9, Figure 3.8).

4.4 Evaluation of caribou winter range use over time

The historical core area of the Redrock Prairie Creek winter range, based on caribou location data from 1980-1996, roughly followed the division between upper foothills and the subalpine subregions, with the delineated mountain area containing areas southwest of both Caw Ridge and Sulphur Mountain (Figure 3.9). Annual 95% KDE ranges between 1998 and 2004 were disproportionally (51% to 92% of their annual range) within the historic core area of the winter range; however, after 2004, RPC caribou had a higher proportion (51% to 80% of their annual range) within the mountain area of the winter range (Figure 3.10; Figure A.3). This temporal shift is masked when examining the global dataset (44% and 56% in core and mountain areas, respectively). A similar trend was observed when examining the proportion of predicted high use areas within the core area of the winter range (Figure 3.10). From 1998 to 2004, a high proportion of predicted high use areas occurred within the core area, whereas after 2004, only a very small proportion of the high use areas occurred in the historic core winter range.

5. Discussion

Understanding patterns of resource selection across a gradient of landscape conditions is necessary to inform conservation and management strategies for threatened species, particularly in rapidly changing landscapes (e.g. Beckmann et al. 2012). Forested landscapes in Alberta are experiencing rapid change due to widespread resource development. The RPC woodland caribou population in west-central Alberta has been monitored since the 1980s, and studied more intensively since 1998, providing an excellent opportunity to examine how caribou are responding to these changes on the landscape. I used caribou location data to develop annual and global resource selection models using temporally-matched landscape variables and compared these to global models that did not incorporate landscape change. I quantified caribou resource selection during winter at both coarse and fine spatial scales, and examined general patterns in range use over time.

Caribou resource selection in the RPC winter range was quite variable between years and across individuals, and pooling location data across multiple years. Applying a static landscape, as represented in the global-static model, produced generally weaker selection coefficients than the annual models and in some cases gave potentially misleading results. In contrast, accounting for landscape change provided a stronger indication of resource selection over time. For disturbance covariates at a coarse scale, caribou generally avoided areas with higher densities of cutblocks and wildfires. Response to linear feature density (roads and pipelines) was more variable, and overall avoidance was less pronounced in the global dynamic model. At a fine scale, caribou generally avoid all disturbances, including areas with higher densities of cutblocks, linear and seismic feature, and wellsites, as well as wildfires. A shift in RPC winter range use was observed from the historical core range in the foothills, which experienced relatively high rates of anthropogenic disturbance, to the less disturbed mountain portion of their winter range.

Identification of resource selection requires consideration of multiple scales to characterize scale-dependent habitat relationships (Johnson 1980, Rettie and Messier 2000). I found support for scale-dependent response to anthropogenic disturbance in the RPC caribou population, suggesting that caribou respond to features differently across spatial scales. At a coarse scale, annual models generally indicated that areas of high cutblock density were significantly avoided, whereas areas of high linear feature density showed more variable selection that was positive in some years. At a finer scale, annual models indicated that all areas of high disturbance densities (including the addition of seismic and wellsite) were avoided, with the exception of high wellsite density in 2000.

In a regional study of multiple caribou populations in west-central Alberta, DeCesare et al. (2012) found that, at a coarse scale, caribou significantly avoided high cutblock densities and had a weak and inconsistent selection for higher linear feature densities, whereas at a fine scale, a weak and inconsistent selection of high cutblock densities and consistent avoidance of high linear feature densities was observed. Results from my study are similar; however, I observed consistent avoidance of areas with higher densities of cutblocks at both scales, although response was weaker at the fine scale.

Because the hypothesized mechanisms underlying caribou response to different disturbances varies, it is not surprising to see variation in response across scales. At a landscape scale, disturbances that alter the amount and configuration of habitats preferred by caribou, and that create habitats more favourable to other prey species, and hence predators, are likely to influence coarse-scale patterns of use by caribou (Seip 1992). As a result, the avoidance by caribou of areas with higher cutblock densities at a landscape scale may be driven by both loss and alteration of habitat, and also by increases in predators (Wittmer et al. 2007, DeCesare et al. 2012). Compared to cutblocks, linear features represent a more dispersed landscape disturbance that may exacerbate predation due to increased encounter rates (James and Stuart-Smith 2000, Latham et al. 2011). Local management can also confound modelled response of caribou to disturbance across scales. In the RPC range forest harvesting was deferred in the central portion of the core winter range from 2004 forward, in order to protect important caribou habitat. However, energy development proceeded, resulting in a high density of associated disturbance (seismic lines, pipelines, roads and wellsites) in high quality caribou habitat.

At a fine scale, caribou generally avoided all areas of high disturbance. Other studies of caribou response to wellsites (e.g. Dyer 1999, Neufeld 2006) were inconclusive. However, these studies examined distance to individual wellsites instead of wellsite density, suggesting that the intensity of disturbance is an important factor influencing caribou response.

Similar to a previous study in the RPC winter range, I found that caribou significantly avoided roads and pipelines, but not seismic lines (Oberg 2002). In contrast, a study of boreal caribou in Alberta reported caribou avoided seismic lines by 250m (Dyer 2001). Smith (2004) suggested the main reason for this discrepancy is that the majority of the seismic lines in the RPC range are > 30 years old, and recent exploration has implemented low impact seismic methods which greatly reduce line width and associated disturbance.

An important characteristic to consider when interpreting the RSF model results is that annual anthropogenic disturbance layers were based on an approval date and did not represent the actual date of disturbance on the landscape (with the exception of cutblocks). As well, seismic features were not temporally referenced beyond pre and post 2005. Therefore possible mismatches between caribou locations and actual landscape covariates may still have occurred. However, this still provides a better approximation of dynamic landscape conditions than a snapshot.

I examined the implications of using a static landscape representation that did not model change, and illustrated that this approach can provide misleading results, especially in areas where new disturbances have occurred. At a coarse scale, despite annual models indicating avoidance of burnt areas, the global-static model reported selection for burnt areas. For example, a wildfire occurred in 2006 and burnt over 14,000 ha of the RPC winter range. Caribou location data indicate that use of this area occurred prior to the wildfire; principally, in 2000 and 2005, and animals did not return to the area until 2010, four years after the fire (Figure A.3). Similarly, at a fine scale, annual models demonstrated consistent avoidance of areas with higher wellsite densities (except in 2000), whereas the global-static model indicated no avoidance. A pulse of wellsite development occurred between 2001 and 2008 in the foothills portion of the RPC winter range. Use locations in this portion of the winter range are concentrated between 1998 to 2000, suggesting that the global-static model response is driven again by use locations in an area before the disturbance occurred. By incorporating the annual landscape changes, the global-dynamic model provided a more accurate representation of caribou resource selection than the global-static model, and better predictive ability, and is thus better suited for management applications.

A shift in range use was observed over the 13-year monitoring period. Historical studies indicate that traditional RPC winter range was concentrated in the foothills (SE) portion of the winter range; however, as of 2004, a higher proportion of the annual RPC winter range occurred within the mountain region (Figure 3.10; Figure A.3). In addition, a higher proportion of high use areas were predicted to occur in the mountain portion of the winter range after 2004. The mountain portion of the winter range has high overlap with areas typically classified as summer range for RPC animals. Recent data from the past winter (2011-12) confirm that 9 of the 11 GPS collared females did not make traditional movements to the historical core winter range and remained in the mountains year round (Weyerhaeuser Company Ltd., unpublished data).

Caribou are thought to show strong fidelity to their winter range (Schaefer et al. 2000), but caribou distribution has changed in another mountain caribou population in west-central (A la Peche), where the traditional migration to the forested winter range has been abandoned since 1996 (Smith 2004). In this population, cumulative road and cutblock densities within home ranges of caribou were negatively correlated with adult female survival (Smith 2004), with proposed indirect mechanisms including an increase in predator hunting efficiency (James and Stuart-Smith 2000), and an increase in alternate prey resulting in higher densities of predators (Seip 1992, Kinley and Apps 2001).

The changes I report in the RPC winter range are also reflected in population demography. A recent Status Report on Woodland Caribou in Alberta (ASRD & ACA 2010) lists the RPC population as declining, whereas the previous report listed them as stable (Dzus 2001). A conspicuous drop in the estimated female population size occurred from 2003-04 to 2004-05 (ASRD & ACA 2010; Figure A.6). Similar to other populations in Alberta, RPC caribou may be altering their range use to avoid higher predator densities in the industrial portion of the range (James et al. 2004, Neufeld 2006, Latham et al. 2011), but this could force them into lower quality habitats.

6. Conclusion

Woodland caribou are sensitive to landscape change and numerous studies suggest that populations across their range in Canada will continue to decline given the current rate of industrial activity and associated disturbance (Environment Canada 2008, Sorensen et al. 2008, Vistnes and Nellemann 2008, Vors and Boyce 2009, Festa-Bianchet et al. 2011). In recent years, RSF models have been developed to assess wildlife response to natural and anthropogenic disturbances (Polfus et al. 2011, Moreau et al. 2012, Beckmann et al. 2012).

My analyses illustrate the importance of incorporating landscape dynamics in the development of RSF models. Caribou were negatively associated with high densities of disturbance at multiple scales, and complementary analyses revealed a spatial shift in the core winter range over a 13-year sampling period. RSFs should be viewed as one tool to support comprehensive management and conservation decisions, and they should be re-evaluated as new knowledge becomes available.

Future work should consider further refinement of time-series data to improve the temporal match between animal location data and disturbance data, in order to reduce potential data contamination (e.g. Johnson et al. 2006), and to address potential time lags in caribou response (e.g. Vors et al. 2007). Addressing spatial data issues and identifying lagged responses could improve the utility of dynamic RSF models for management applications. Additional population information should be incorporated to link resource selection and population demography, in order to address habitat quality and direct management actions (Van Horne 1983, DeCesare 2012).

Table 3.1: Redrock Prairie Creek (RPC) GPS location data for caribou available during the 1998-2011 winter seasons (December 1 to April 30 of each year). Individual caribou may appear in multiple years.

Winter	Number of Animals	Number of Locations
1998	5	3274
1999	9	4233
2000	16	11528
2001	10	6242
2002	10	6065
2003	6	3517
2004	9	5902
2005	8	6633
2006	10	8329
2007	8	6805
2008	8	6888
2009	8	4141
2010	8	5802
Global	115	79359

Table 3.2: GIS covariates used to model resource selection of woodland caribou in the RPC caribou winter range in west-central Alberta, Canada (1998-2011).

Description	Variable Code	Resolution (m)	2nd Order RSF	3rd Order RSF
Elevation (m)	elev	30	X	X
Slope (degrees)	slope	30	X	X
Percent Snow Cover	psnow	500	X	
Topographic Position Index	tpi	30	X	X
Proportion of Cutblocks within 70m and 5km radius	cden	30	5 km	70 m
Density of Linear within 70m and 5km radius	lden	30	5 km	70 m
Density of Seismic within 70m and 5km radius	wden	30	5 km	70 m
Density of Wellsites within 70m and 5km radius	wden	30	5 km	70 m
Density of Disturbance (cumulative)	tden	30	5 km	70 m
Closed Conifer (reference category)	con_c	30	X	X
Open Conifer	con_o	30	X	X
Mixed Forest	mix	30	X	X
Deciduous Forest	dec	30	X	X
Herbaceous	herb	30	X	X
Shrub	shrub	30	X	X
Alpine	alpine	30	X	X
Rock/Ice	rock/ice	30	X	X
Muskeg/Wetland	wetland	30	X	X
Water	water	30	X	X
Cutblock	cutblk	30	X	X
Burn	burn	30	X	X
No Data	no_data	30	X	X

Table 3.3: Candidate models used to model second-(A) and third-order (B) resource selection of woodland caribou in the RPC caribou winter range in west-central Alberta, Canada (1998-2011).

A. Second-order

Model Number	Variables
M1	total density
M2	cutblock + linear density
M3	elevation + slope + tpi
M4	landcover + snow + elevation + slope + tpi
M5	total + elevation + slope + tpi
M6	cutblock + linear + elevation + slope + tpi
M7	total + landcover + snow + elevation + slope + tpi
M8	cutblock + linear + landcover + snow + elevation + slope + tpi

B. Third-order

Model Number	Variables
M1	total density
M2	cutblock + linear density
M3	elevation + slope + tpi
M4	landcover + snow + elevation + slope + tpi
M5	total + elevation + slope + tpi
M6	cutblock + linear + elevation + slope + tpi
M7	total + landcover + snow + elevation + slope + tpi
M8	cutblock + linear + landcover + elevation + slope + tpi
M9	cutblock + linear + wellsite + seismic
M10	cutblock + linear + wellsite + seismic + landcover + elevation + slope + tpi

Table 3.4: Wildfire disturbance area (Ha) within the RPC caribou winter range in west-central Alberta, Canada from 1938 to 2010.

Year	Area (Ha)
1938	1541
1941	950
1956	173
1960	3538
1961	1266
1971	95
1974	332
1981	61
1982	1251
1987	1738
1988	43
1992	547
2002	1556
2005	2
2006	20132
2009	1166
2010	13
Total	34405

Table 3.5: Pearson's correlations r between modeling covariates for the RPC population in west-central Alberta. Second-order covariates shown shaded in the bottom left and third-order covariates shown in top right. Correlated covariates are shown in bold.

Covariate	Wellsite	Total	Seismic	Linear	Cutblk	Landcvr	Snow	Elev	Slope	TPI
Wellsite	1	0.136	0.033	0.066	0.022	0.024		-0.014	-0.020	0.006
Total	0.609	1	0.454	0.481	**0.775**	0.175		-0.166	-0.137	-0.002
Seismic	0.325	0.408	1	0.141	0.078	-0.043		-0.147	-0.128	-0.029
Linear	**0.795**	0.676	0.441	1	0.054	0.036		-0.098	-0.091	0.019
Cutblk	0.430	**0.953**	0.266	0.455	1	0.230		-0.092	-0.063	-0.002
Landcover	-0.198	-0.094	-0.104	-0.228	-0.006	1		0.399	0.131	0.245
Snow	-0.325	-0.218	-0.184	-0.395	-0.091	0.278	1			
Elevation	-0.533	-0.424	-0.282	-0.604	-0.276	0.415	0.601	1	0.434	0.486
Slope	-0.311	-0.258	-0.219	-0.309	-0.176	0.168	0.100	0.412	1	0.164
TPI	-0.088	-0.076	-0.052	-0.097	-0.059	0.247	0.157	0.494	0.1725	1

Table 3.6: Summary of top annual and global models for second-order RPC caribou resource selection in winter (December 1 to April 30). Akaike weights (w_i) indicate the likelihood of the model being the best of those tested. Full second-order model selection results are shown in Appendix A Table A.2.

Year	M#	Annual Top Model	w_i
1998	M8	ln_cut + ln_lin + lc + elev + slope + tpi	1.00
1999	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2000	M7	ln_tden + lc + snow + elev + slope + tpi	1.00
2001	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2002	M7	ln_tden + lc + snow + elev + slope + tpi	1.00
2003	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2004	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2005	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2006	M7	ln_tden + lc + snow + elev + slope + tpi	1.00
2007	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2008	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2009	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
2010	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
Global-Static	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00
Global-Dynamic	M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	1.00

Table 3.7: Summary of top annual and global models for third-order RPC caribou resource selection in winter (December 1 to April 30). Akaike weights (w_i) indicate the likelihood of the model being the best of those tested. Full third-order model selection results are shown in Appendix A Table A.3.

Year	M#	Annual Top Model	w_i
1998	M8	cut + lin + lc + elev + slope + tpi	0.81
1999	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.99
2000	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.95
2001	M8	cut + lin + lc + elev + slope + tpi	0.87
2002	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.51
2003	M10	cut + lin + well + seismic + lc + elev + slope + tpi	1.00
2004	M8	cut + lin + lc + elev + slope + tpi	0.83
2005	M8	cut + lin + lc + elev + slope + tpi	0.75
2006	M10	cut + lin + well + seismic + lc + elev + slope + tpi	1.00
2007	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.98
2008	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.94
2009	M10	cut + lin + well + seismic + lc + elev + slope + tpi	1.00
2010	M10	cut + lin + well + seismic + lc + elev + slope + tpi	0.58
Global-Static	M10	cut + lin + well + seismic + lc + elev + slope + tpi	1.00
Global-Dynamic	M10	cut + lin + well + seismic + lc + elev + slope + tpi	1.00

Table 3.8: Second-order selection coefficients (β) and standard errors (SE) for the RPC population in west-central Alberta. Selection was measured in winter (December 1 - April 30). Positive selection coefficients indicate selection for that covariate and negative selection coefficients indicate avoidance.

	ln_total	ln_cut	ln_lin	alpine	burn	cutblk	dec	herb	mix	o_con	rock	shrub	water	wetland	snow	elev	slope	tpi
1998		**-0.264**	**-0.223**	**0.957**	0.123	**-1.778**	**-0.624**	**-1.422**	0.250	**-0.501**	**-1.619**	**0.360**	-18.099	**-1.589**		**-0.001**	**-0.061**	**0.010**
		(0.054)	(0.029)	(0.095)	(0.231)	(0.742)	(0.300)	(0.563)	(0.190)	(0.137)	(0.179)	(0.106)	(4291.6)	(0.382)		(1.84E-04)	(0.004)	(0.001)
1999		**0.322**	**-0.362**	**0.723**	**-0.808**	**-1.303**	0.005	**-1.642**	-0.334	-0.218	**-1.018**	**0.288**	-18.402	**-2.799**	**-3.608**	**0.003**	**-0.166**	**0.006**
		(0.055)	(0.032)	(0.100)	(0.296)	(0.426)	(0.283)	(0.750)	(0.200)	(0.123)	(0.137)	(0.097)	(2671.7)	(0.742)	(0.184)	(2.16E-04)	(0.005)	(0.001)
2000	**0.462**			**-0.420**	**-0.754**	**-3.508**	**-1.004**	**-1.093**	**-0.470**	**0.177**	**-1.536**	**-0.217**	**-2.197**	**-0.451**	**3.521**	**0.002**	**-0.154**	**0.006**
	(0.025)			(0.061)	(0.144)	(0.283)	(0.215)	(0.319)	(0.137)	(0.061)	(0.087)	(0.060)	(0.770)	(0.193)	(0.240)	(1.22E-04)	(0.003)	(3.79E-04)
2001		**-1.286**	**0.889**	-0.122	-0.345	**-0.985**	**-1.973**	**-3.203**	**-0.936**	**0.252**	**-1.100**	**0.576**	-18.886	**-1.427**	**4.652**	**0.001**	**-0.085**	**0.008**
		(0.033)	(0.018)	(0.072)	(0.200)	(0.269)	(0.357)	(0.724)	(0.167)	(0.085)	(0.097)	(0.062)	(5989.0)	(0.429)	(0.310)	(1.40E-04)	(0.003)	(4.72E-04)
2002	**0.830**			**1.260**	-0.125	**-2.183**	**0.850**	**-1.915**	**0.329**	**1.295**	**-1.151**	**0.907**	-18.347	**-0.774**	0.032	**0.003**	**-0.135**	**0.005**
	(0.035)			(0.083)	(0.197)	(0.217)	(0.227)	(0.739)	(0.162)	(0.091)	(0.123)	(0.080)	(4682.7)	(0.388)	(0.242)	(1.68E-04)	(0.004)	(0.001)
2003		**-1.460**	**0.317**	**1.687**	-0.367	-19.313	-0.264	**-1.745**	-0.075	-0.247	**1.850**	**1.066**	-22.011	**-2.176**	**3.232**	**-0.002**	**-0.137**	**0.010**
		(0.070)	(0.027)	(0.130)	(0.316)	(6118.7)	(0.348)	(0.759)	(0.244)	(0.158)	(0.152)	(0.104)	(22596.1)	(0.648)	(0.416)	(2.51E-04)	(0.005)	(0.001)
2004		**-0.589**	**0.272**	**3.085**	**0.753**	-0.381	0.198	**0.646**	**0.848**	**0.630**	**0.863**	**0.764**	**-2.532**	0.321	**1.277**	**-0.001**	**-0.100**	**0.006**
		(0.037)	(0.020)	(0.097)	(0.153)	(0.230)	(0.223)	(0.248)	(0.144)	(0.102)	(0.121)	(0.075)	(1.049)	(0.235)	(0.290)	(1.72E-04)	(0.004)	(4.63E-04)
2005		**-0.792**	**0.191**	0.068	**-2.393**	**-2.687**	-0.254	**-1.335**	-0.061	**0.581**	**-0.333**	**0.414**	**-1.996**	-0.204	**-0.509**	**0.004**	**-0.098**	**0.002**
		(0.042)	(0.019)	(0.074)	(0.399)	(0.723)	(0.226)	(0.416)	(0.161)	(0.078)	(0.094)	(0.082)	(0.862)	(0.210)	(0.054)	(1.67E-04)	(0.003)	(4.14E-04)
2006	**0.408**			**0.705**	**-1.227**	**-4.410**	-0.139	**-2.208**	-0.315	**0.614**	**-0.679**	**-0.279**	-1.798	**0.952**	**-0.747**	**0.007**	**-0.082**	**0.002**
	(0.032)			(0.081)	(0.201)	(1.007)	(0.271)	(1.015)	(0.206)	(0.077)	(0.098)	(0.116)	(1.119)	(0.217)	(0.051)	(1.83E-04)	(0.003)	(3.78E-04)
2007		**-0.460**	**-0.288**	**-1.123**	**-2.947**	**-1.780**	-0.393	**-2.578**	0.010	-0.037	**-1.259**	-0.141	**-2.029**	**-3.872**	**-1.864**	**0.004**	**-0.089**	**0.006**
		(0.053)	(0.022)	(0.077)	(0.222)	(0.598)	(0.246)	(0.745)	(0.197)	(0.072)	(0.100)	(0.102)	(0.701)	(0.730)	(0.269)	(1.78E-04)	(0.003)	(4.49E-04)
2008		-0.062	**0.198**	**0.335**	**-3.785**	**-4.530**	0.042	**-0.950**	**-0.466**	**-0.260**	**-0.516**	**-0.283**	-18.908	**-2.643**	**-3.256**	**0.004**	**-0.108**	**0.005**
		(0.034)	(0.019)	(0.074)	(0.456)	(0.718)	(0.186)	(0.415)	(0.170)	(0.091)	(0.099)	(0.086)	(3285.7)	(0.533)	(0.345)	(1.60E-04)	(0.003)	(4.34E-04)
2009		**-0.139**	**0.157**	**0.712**	**-2.395**	-22.436	**-1.260**	0.405	**-1.087**	**-0.312**	-0.016	**0.326**	-21.114	0.372	**-0.468**	0.000	**-0.090**	**0.014**
		(0.038)	(0.021)	(0.104)	(0.298)	(6214.8)	(0.384)	(0.243)	(0.229)	(0.130)	(0.127)	(0.096)	(19207.2)	(0.242)	(0.062)	(2.02E-04)	(0.004)	(6.41E-04)
2010		**0.776**	**-0.179**	0.097	-0.234	**-3.619**	**-0.805**	-0.502	**-0.753**	0.131	**-1.346**	**0.881**	-1.252	-0.053	**-1.205**	**0.005**	**-0.062**	**-0.001**
		(0.045)	(0.024)	(0.076)	(0.122)	(0.433)	(0.284)	(0.372)	(0.222)	(0.084)	(0.099)	(0.086)	(0.672)	(0.230)	(0.323)	(1.78E-04)	(0.003)	(4.05E-04)
Global-Static		**-0.308**	**0.344**	**0.405**	**0.305**	**-1.439**	**-0.463**	**-0.664**	**-0.164**	**0.380**	**-0.617**	**0.313**	**-2.193**	**-0.163**	-0.167	**0.003**	**-0.100**	**0.004**
		(0.009)	(0.005)	(0.021)	(0.030)	(0.062)	(0.065)	(0.099)	(0.046)	(0.024)	(0.027)	(0.022)	(0.300)	(0.074)	(0.103)	(4.83E-05)	(8.94E-04)	(1.23E-04)
Global-Dynamic		**-1.364**	**-0.193**	**0.569**	**-1.081**	**-1.957**	**-0.790**	**-1.331**	**-0.513**	**0.318**	**-0.810**	**0.529**	**-2.274**	**-0.667**	**0.006**	**0.000**	**0.001**	**-0.002**
		(0.172)	(0.038)	(0.020)	(0.046)	(0.098)	(0.063)	(0.095)	(0.045)	(0.022)	(0.026)	(0.021)	(0.293)	(0.069)	(1.24E-04)	(6.28E-05)	(3.93E-05)	(2.45E-04)

Table 3.9: Third-order selection coefficients (β) and standard errors (SE) for the RPC population in west-central Alberta. Selection was measured in winter (December 1 - April 30). Positive selection coefficients indicate selection for that covariate and negative selection coefficients indicate avoidance.

	ln_cut	ln_lin	ln_well	ln_seis	alpine	burn	cutblk	dec	herb	mix	o_con	rock	shrub	water	wetland	elev	slope	tpi
1998	0.021	**-0.251**			**1.125**	0.025	-1.814	**-0.694**	-1.247	0.236	**-0.769**	-0.168	**0.517**		**1.855**	**0.003**	0.004	**3.45E-38**
	(0.236)	(0.086)			(0.121)	(0.203)	(1.319)	(0.285)	(0.643)	(0.187)	(0.143)	(0.209)	(0.101)		(0.700)	(2.41E-04)	(0.005)	(2.95E-39)
1999	**-1.060**	-0.118		**-0.040**	**0.599**	**-1.140**	2.838	-0.032	**-1.313**	-0.042	-0.149	0.023	**0.548**	**-10.967**	-0.134	**0.006**	**-0.125**	0.001
	(0.296)	(0.067)		(0.020)	(0.209)	(0.278)	(1.448)	(0.290)	(0.619)	(0.215)	(0.160)	(0.194)	(0.131)	(1.013)	(0.825)	(3.95E-04)	(0.009)	(0.001)
2000	**-0.560**	-0.012	**1.352**	**-0.023**	**0.248**	0.129	0.079	**-0.697**	**-0.799**	-0.269	**0.280**	**-1.160**	**0.152**	-1.677	-0.283	**0.004**	**-0.139**	**0.003**
	(0.080)	(0.030)	(0.067)	(0.009)	(0.087)	(0.184)	(0.461)	(0.216)	(0.357)	(0.139)	(0.085)	(0.110)	(0.064)	(1.130)	(0.255)	(1.65E-04)	(0.004)	(5.75E-04)
2001	-0.008	**-0.284**			**0.878**	**-0.692**	**-1.659**	**-1.076**	-1.335	-0.134	**0.425**	**0.469**	**0.760**	**-12.256**	1.113	**0.006**	**-0.024**	-0.001
	(0.092)	(0.039)			(0.115)	(0.274)	(0.508)	(0.470)	(0.826)	(0.187)	(0.136)	(0.125)	(0.070)	(1.016)	(0.688)	(2.44E-04)	(0.004)	(7.87E-04)
2002	**-0.245**	**-0.104**	**-1.540**	-0.013	**1.322**	0.047	-0.107	**0.772**	**-1.466**	**0.406**	**0.798**	-0.153	**0.752**		0.877	**0.004**	**-0.100**	0.001
	(0.077)	(0.031)	(0.077)	(0.012)	(0.102)	(0.160)	(0.382)	(0.235)	(0.737)	(0.184)	(0.100)	(0.152)	(0.078)		(0.608)	(2.51E-04)	(0.005)	(0.001)
2003	**-6.431**	**-0.338**	**-1.946**	**-0.083**	**1.060**	**-1.658**	**12.263**	-0.396	-1.203	-0.398	0.093	**1.204**	**1.173**	**-15.257**	-1.421	**0.001**	**-0.122**	**0.006**
	(0.272)	(0.059)	(0.099)	(0.020)	(0.165)	(0.382)	(1.386)	(0.337)	(0.673)	(0.219)	(0.236)	(0.186)	(0.119)	(0.750)	(0.767)	(3.40E-04)	(0.007)	(0.001)
2004	**-0.430**	**-0.135**			**2.543**	**0.913**	0.905	-0.260	**0.790**	**0.364**	**0.967**	**0.583**	**0.935**	**-14.987**	0.064	**0.001**	**-0.101**	**0.005**
	(0.129)	(0.030)			(0.139)	(0.200)	(0.628)	(0.217)	(0.254)	(0.136)	(0.119)	(0.152)	(0.081)	(0.467)	(0.268)	(2.14E-04)	(0.005)	(5.74E-04)
2005	**-0.545**	**-0.232**			**0.462**	**-1.666**	-0.086	-0.244	**-1.215**	-0.199	**0.338**	0.083	**0.610**	**-2.207**	-0.255	**0.003**	**-0.078**	**0.003**
	(0.184)	(0.042)			(0.077)	(0.393)	(1.126)	(0.200)	(0.430)	(0.159)	(0.076)	(0.110)	(0.083)	(0.671)	(0.256)	(1.83E-04)	(0.003)	(4.70E-04)
2006	**-0.751**	**-0.340**	**-1.538**	**-0.107**	**0.638**	**-1.338**	-0.706	-0.270	**-15.604**	**-0.717**	**0.300**	**-0.490**	0.139	**-2.779**	**1.050**	**0.006**	**-0.081**	**0.003**
	(0.365)	(0.061)	(0.117)	(0.018)	(0.099)	(0.272)	(2.012)	(0.329)	(0.717)	(0.237)	(0.086)	(0.131)	(0.155)	(1.091)	(0.317)	(2.51E-04)	(0.003)	(4.30E-04)
2007	**-0.456**	**-0.297**	**-1.318**	**-0.052**	**-0.567**	**-1.739**	0.542	-0.383	**-2.024**	**-0.648**	**-0.441**	**0.221**	2.84E-04	1.138	**-1.787**	**0.001**	**-0.050**	**0.008**
	(0.215)	(0.072)	(0.078)	(0.017)	(0.070)	(0.236)	(1.310)	(0.203)	(0.771)	(0.182)	(0.057)	(0.100)	(0.101)	(0.660)	(0.700)	(2.03E-04)	(0.003)	(4.80E-04)
2008	**-0.650**	**-0.251**	**-1.779**	0.001	**0.590**	**-3.345**	-1.138	0.169	-0.247	**-0.490**	**-0.703**	0.056	**0.254**	**-13.950**	-0.278	**0.004**	**-0.073**	**0.003**
	(0.150)	(0.034)	(0.072)	(0.014)	(0.084)	(0.476)	(0.976)	(0.221)	(0.488)	(0.204)	(0.095)	(0.115)	(0.110)	(0.513)	(0.756)	(2.07E-04)	(0.004)	(5.06E-04)
2009	**-1.405**	**-0.309**	**-1.791**	**-0.073**	**1.946**	**-3.186**	**-14.339**	-1.000	0.381	-0.236	0.024	**1.406**	**1.526**	**-16.028**	-0.354	**-0.002**	**-0.072**	**0.011**
	(0.349)	(0.050)	(0.079)	(0.021)	(0.165)	(0.369)	(1.421)	(0.561)	(0.304)	(0.327)	(0.169)	(0.198)	(0.137)	(0.512)	(0.269)	(2.79E-04)	(0.006)	(8.42E-04)
2010	-0.094	**-0.167**	**-1.654**	-0.023	**0.494**	**-0.564**	**-2.411**	-0.376	-0.030	-0.191	**-0.202**	**-0.901**	**1.077**	-1.467	**-0.604**	**0.002**	**-0.042**	**0.002**
	(0.101)	(0.051)	(0.115)	(0.021)	(0.076)	(0.147)	(0.663)	(0.278)	(0.388)	(0.246)	(0.079)	(0.110)	(0.092)	(0.772)	(0.199)	(2.10E-04)	(0.003)	(4.35E-04)
Global-	**-0.300**	**-0.029**	0.001	**-0.034**	**0.692**	**-0.123**	-0.090	**-0.268**	0.070	**-0.099**	**0.125**	**-0.087**	**0.610**	**-1.695**	-0.096	**0.003**	**-0.078**	**0.004**
Static	(0.026)	(0.006)	(0.028)	(0.004)	(0.027)	(0.046)	(0.133)	(0.070)	(0.115)	(0.050)	(0.026)	(0.036)	(0.027)	(0.317)	(0.098)	(6.25E-05)	(0.001)	(1.64E-04)
Global-	**-0.366**	**-0.172**	**-0.188**	**-0.026**	**0.718**	**-0.845**	-0.337	**-0.290**	0.070	**-0.142**	**0.097**	0.014	**0.666**	**-2.262**	-0.088	**0.003**	**-0.080**	**0.000**
Dynamic	(0.037)	(0.012)	(0.056)	(0.004)	(0.026)	(0.061)	(0.196)	(0.069)	(0.115)	(0.051)	(0.026)	(0.035)	(0.027)	(0.365)	(0.103)	(5.64E-05)	(0.001)	(2.94E-39)

Table 3.10: Summary of mean cross validated Spearman-rank correlation (mean r_s) and standard deviation (SD) for the second- and third-order scale caribou relative probability bins, derived from the AIC_c-selected best annual and global models from the RPC winter range in west-central Alberta. (Note: ^{1} represents second-order M7 and ^{2} represents third-order M8)

Second-Order Cross Validation			Third-Order Cross Validation		
Year	Mean r_s	(SD)	Year	Mean r_s	(SD)
1998	0.83	(0.01)	1998 ^{2}	0.74	(0.06)
1999	0.82	(0.01)	1999	0.90	(0.02)
2000 ^{1}	0.85	(0.01)	2000	0.99	(0.02)
2001	0.73	(0.03)	2001 ^{2}	0.88	(0.03)
2002 ^{1}	0.87	(0.02)	2002	1.00	(0.01)
2003	0.87	(0.02)	2003	0.99	(0.01)
2004	0.78	(0.03)	2004 ^{2}	0.95	(0.03)
2005	0.76	(0.03)	2005 ^{2}	0.79	(0.03)
2006 ^{1}	0.77	(0.02)	2006	1.00	(0.01)
2007	0.81	(0.03)	2007	0.81	(0.03)
2008	0.89	(0.02)	2008	0.99	(0.01)
2009	0.66	(0.03)	2009	0.92	(0.03)
2010	0.94	(0.01)	2010	0.84	(0.04)
Global-Static	0.67	(0.04)	Global-Static	0.77	(0.03)
Global-Dynamic	0.82	(0.03)	Global-Dynamic	0.81	(0.04)

Figure 3.1: Location of Redrock Prairie Creek (RPC) caribou range in west-central Alberta. Polygons represent the 100% MCP population ranges for the entire RPC range, and the winter range, based on caribou location data collected from 1998-2011.

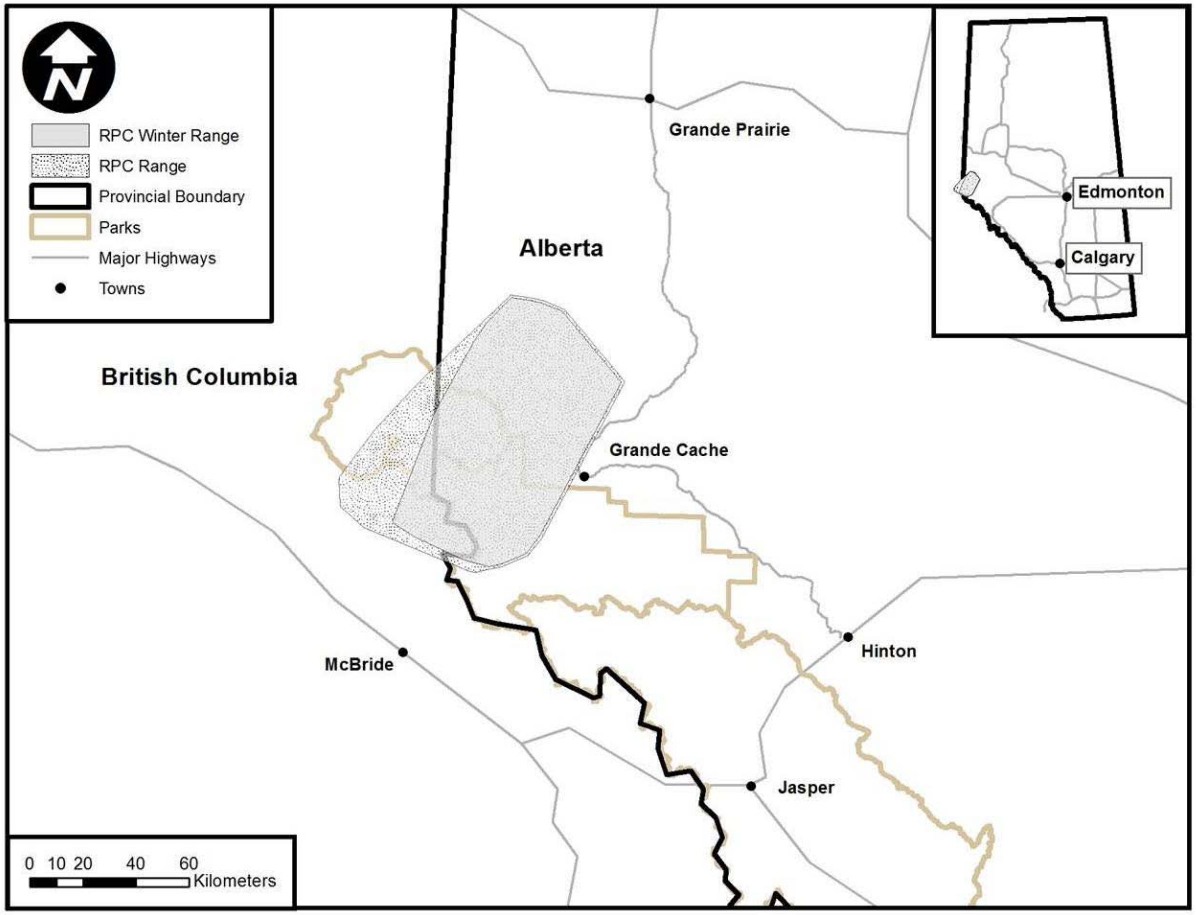


Figure 3.2: RPC winter range GPS data locations for the winter season (December 1 – April 30) from 1998-2011.

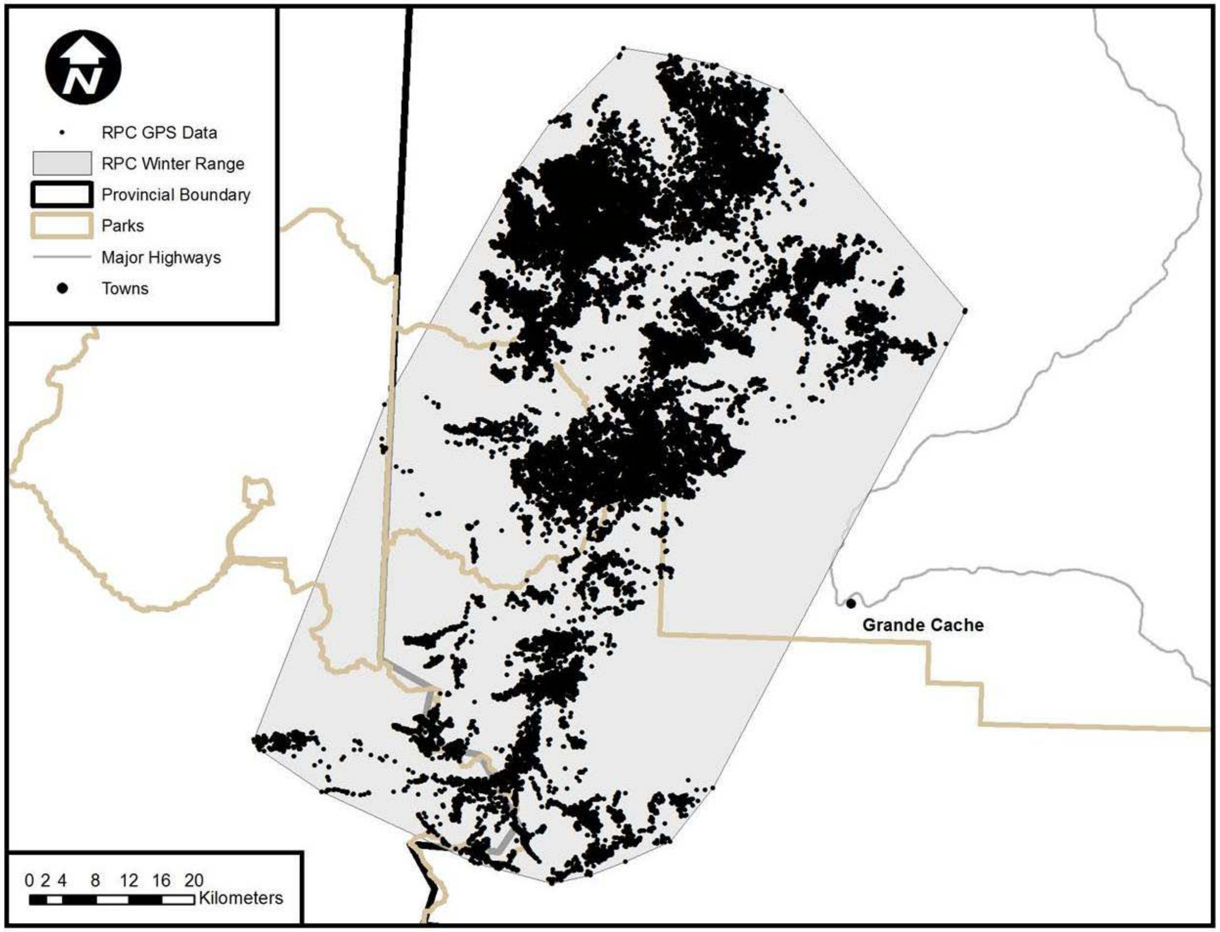


Figure 3.3: Cumulative natural and anthropogenic disturbance in RPC caribou range in west-central Alberta as of April 30, 2011.

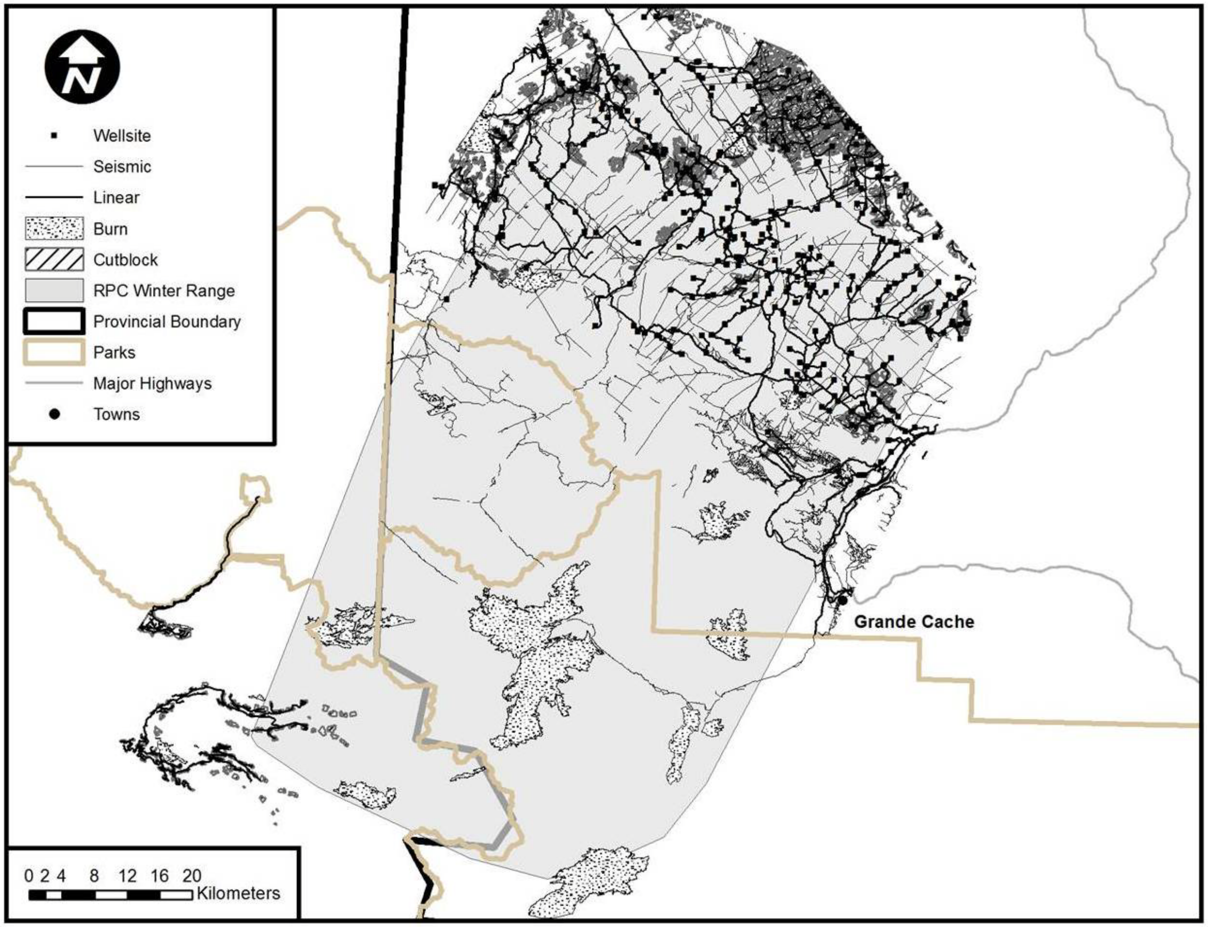


Figure 3.4: Total amount of anthropogenic disturbance within the entire RPC winter range and amount of anthropogenic disturbance within the RPC annual population range (100% MCP). Seismic and other linear features (pipelines and roads) were buffered by 10 m and 25 m respectively.

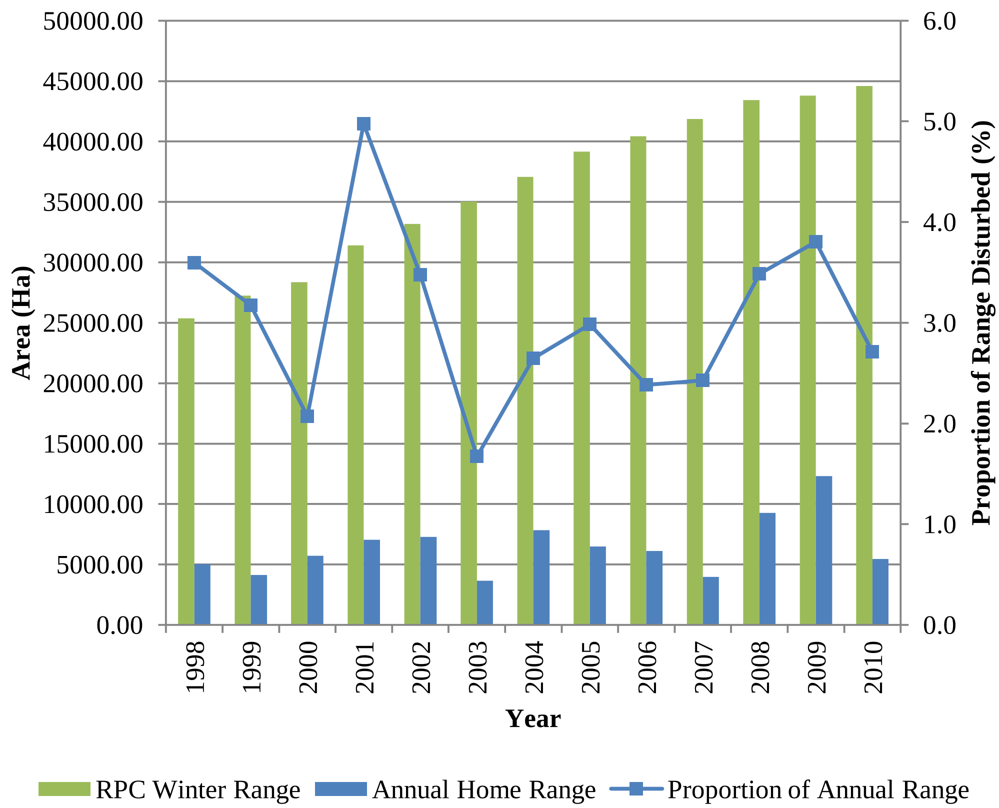


Figure 3.5: Second-order selection coefficients and confidence intervals (CI) for RPC winter resource selection by caribou, for annual models from 1998 to 2010, and the global models (G-Static and G-Dynamic) which included all years. CI values not overlapping zero are significant. A. Total density. B. Cutblock density; B. Linear density; C. Burn. (Note: ** represents M7 coefficient value).

A: Total Density

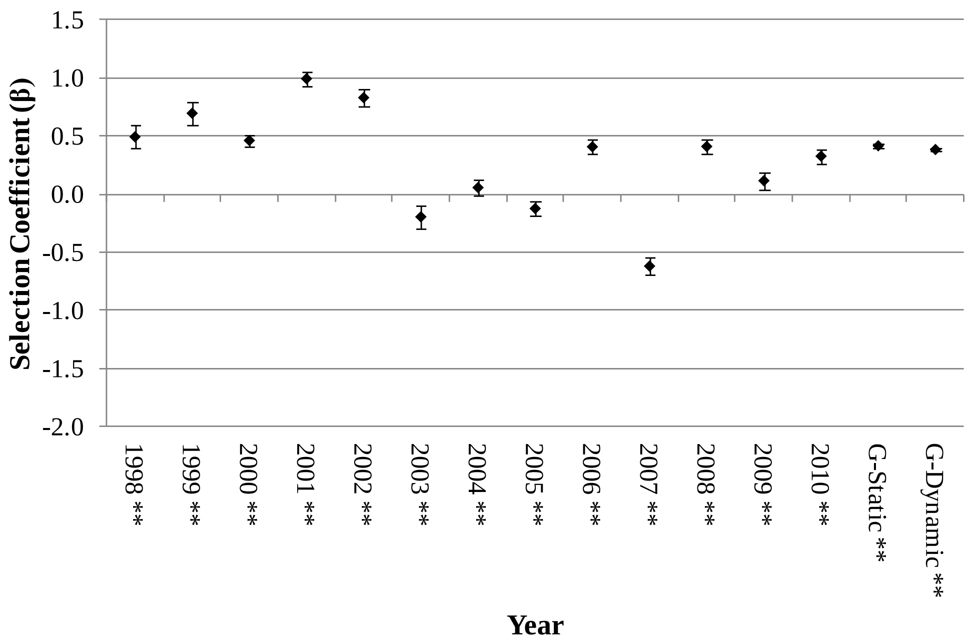
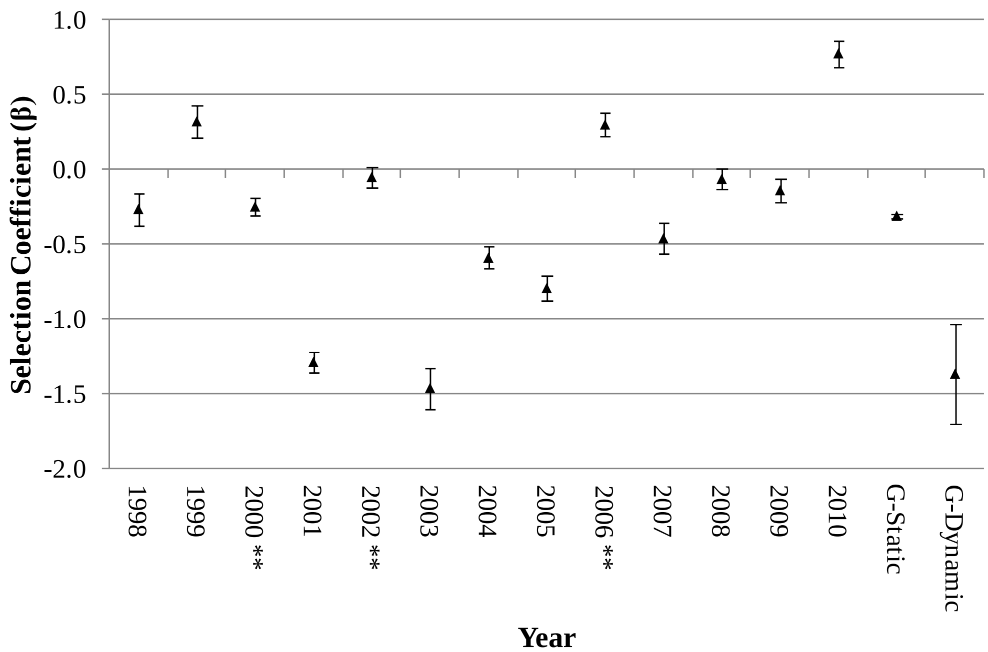


Figure 3.5: Continued. Second-order selection coefficients.

B: Cutblock Density



C: Linear Density

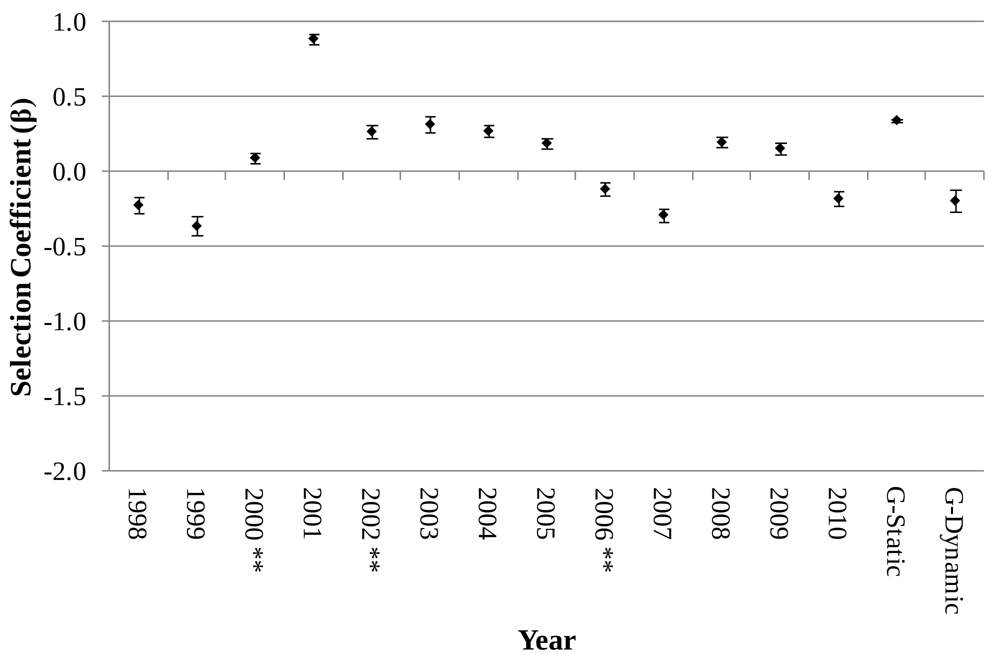


Figure 3.5: Continued. Second-order selection coefficients.

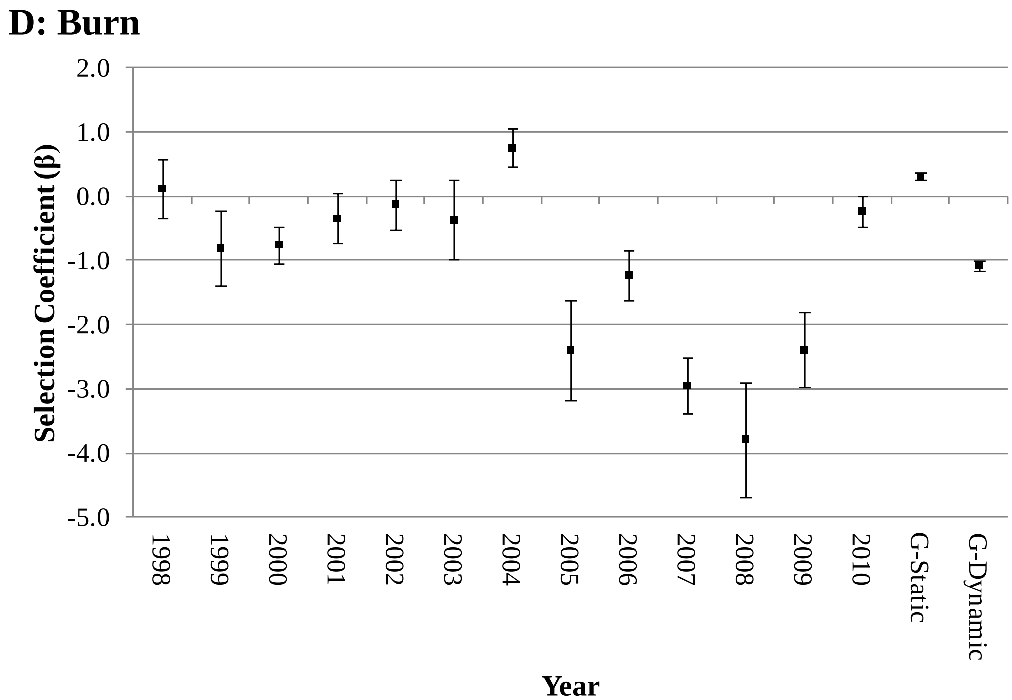


Figure 3.6: Second-order winter resource selection function maps for annual models 2000 (left), 2006 (middle) and the Global-Dynamic model (right) for the RPC winter range in west-central Alberta (December 1 - April 30). The relative probability of selection is scaled between low (0, blue) and high (1, red). See Appendix A Figure A.1 for additional second-order annual prediction maps.

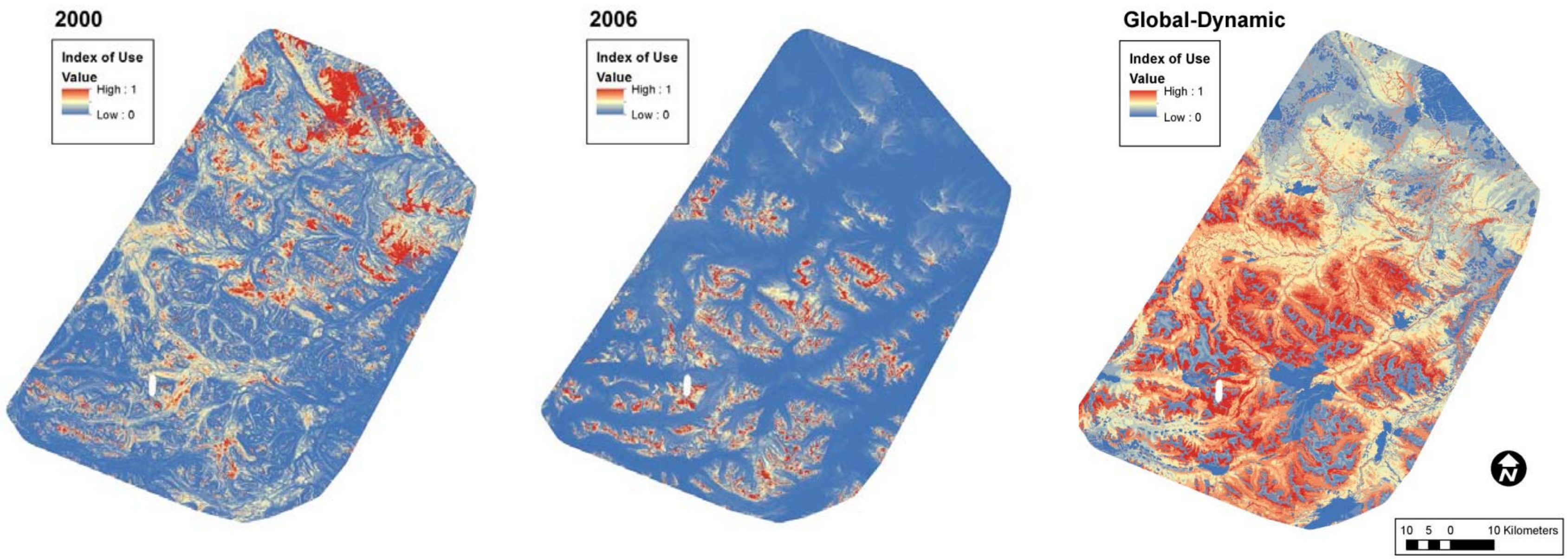


Figure 3.7: Third-order selection coefficients and confidence intervals (CI) for RPC winter resource selection by caribou, for annual models from 1998 to 2010, and the global models (G-Static and G-Dynamic) which included all years. CI values not overlapping zero are significant. A. Cutblock density; B. Linear density; C. Wellsite density; D. Seismic density; E. Burn. (Note: ** represents M8 coefficient value).

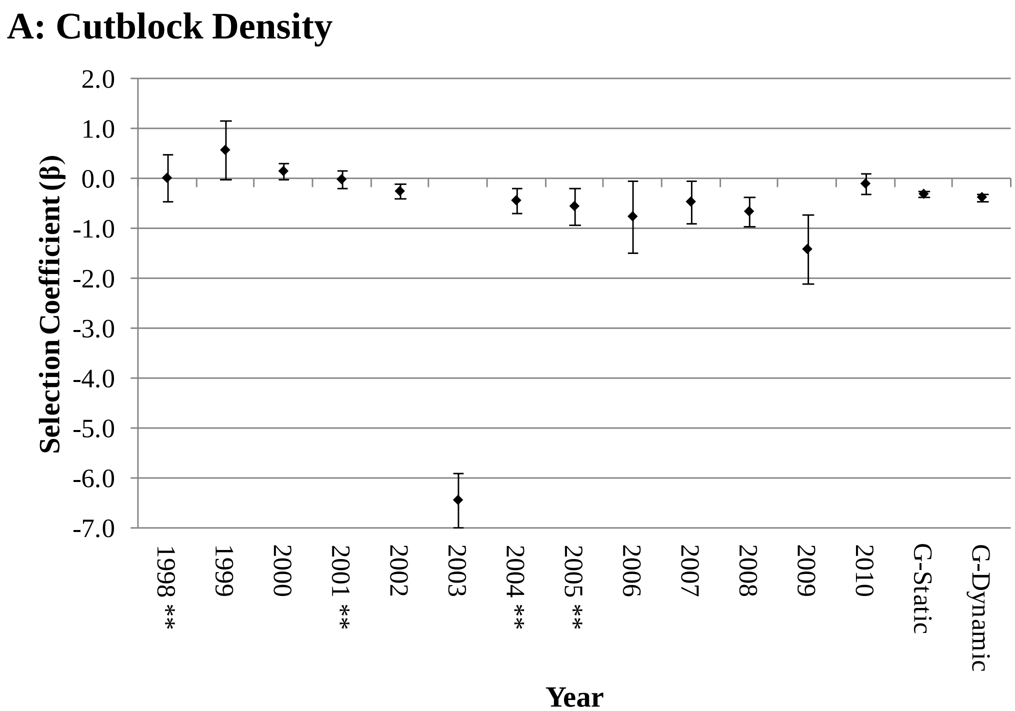
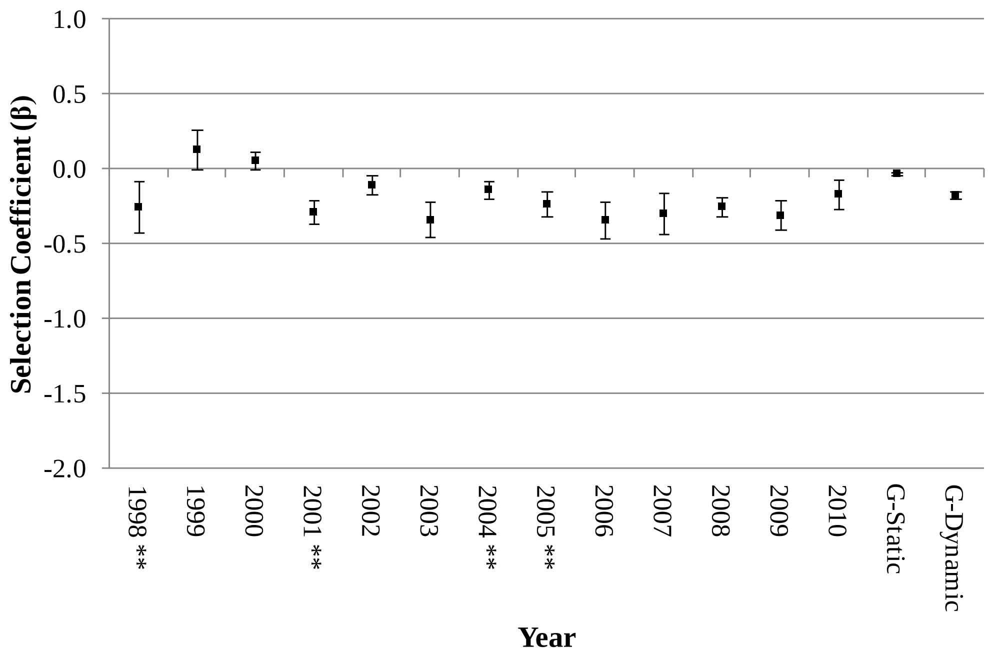


Figure 3.7: Continued. Third-order selection coefficients.

B: Linear Density



C: Wellsite Density

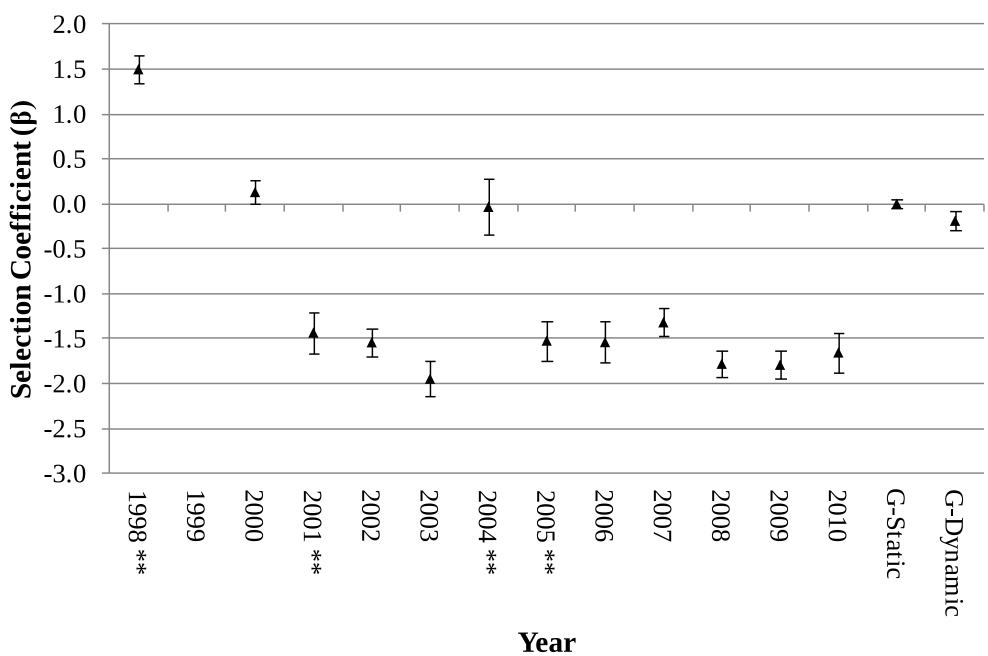
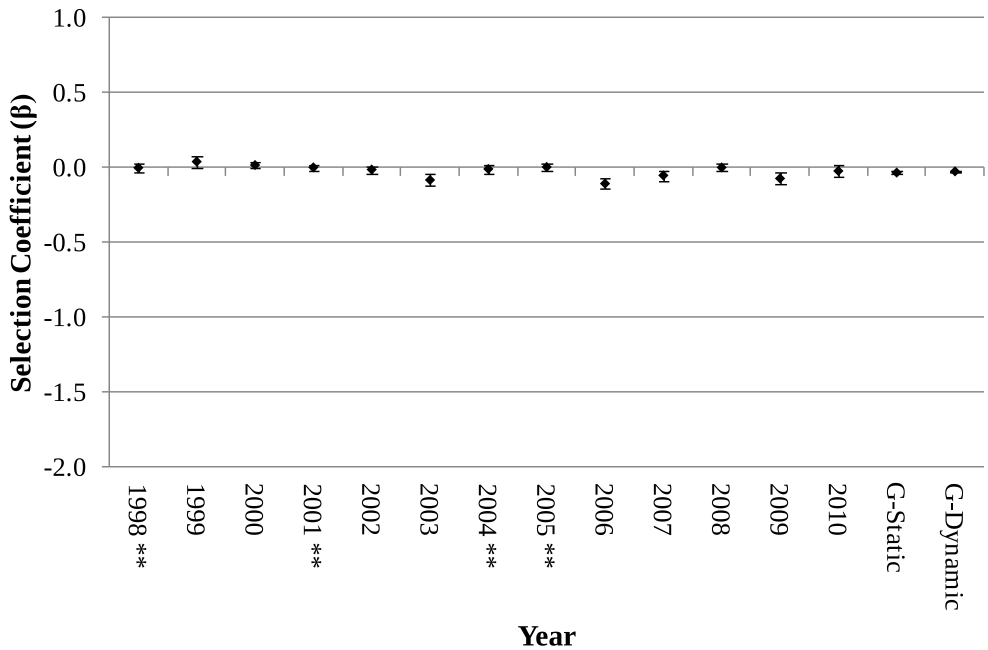


Figure 3.7: Continued. Third-order selection coefficients.

D: Seismic Density



E: Burn

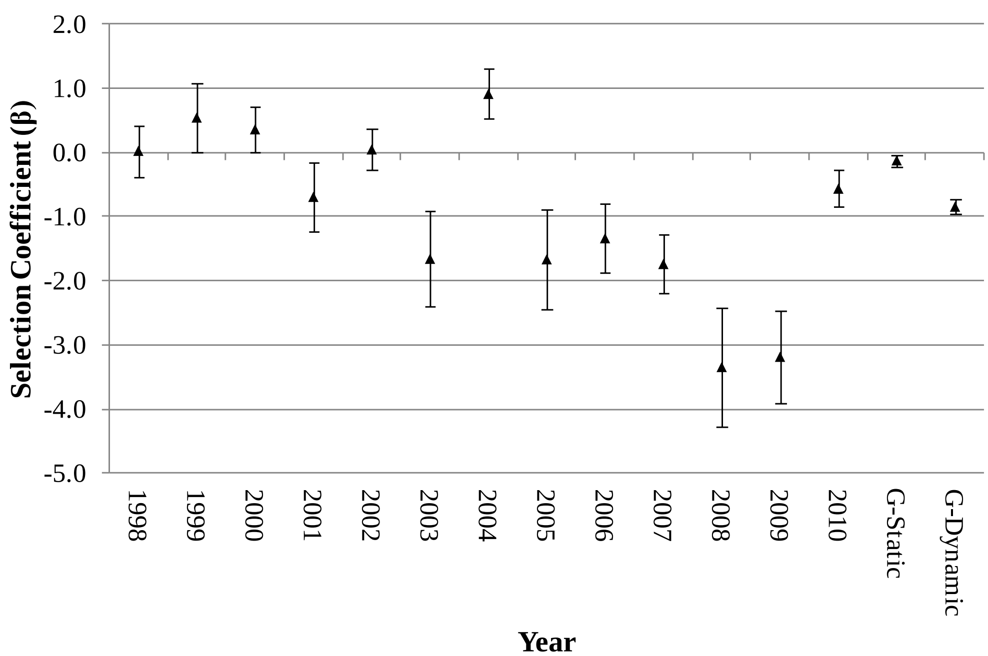


Figure 3.8: Third-order winter resource selection function maps for annual models 2000 (left), 2006 (middle) and the Global-Dynamic model (right) for the RPC winter range in west-central Alberta (December 1 - April 30). The relative probability of selection is scaled between low (0, blue) and high (1, red). See Appendix A Figure A.2 for additional third-order annual prediction maps.

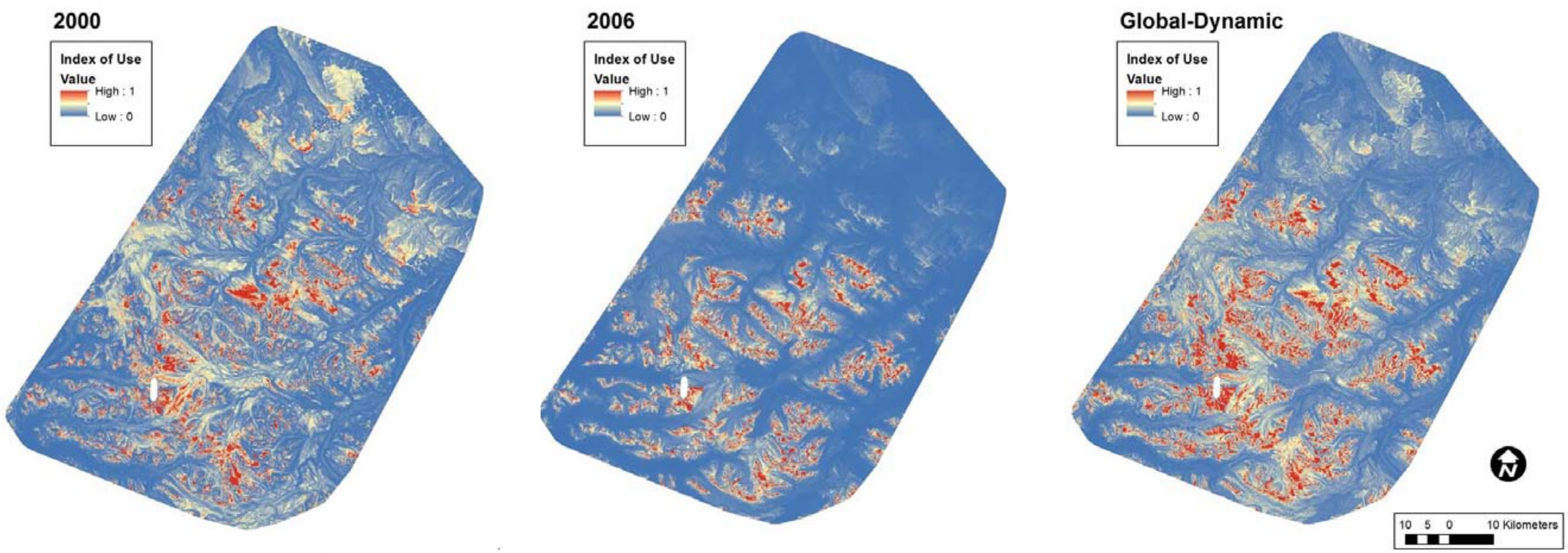


Figure 3.9: Delineation of core and mountain areas of the RPC winter range in west-central Alberta. The division between core and mountain areas is based on historical RPC data (source: Edmonds and Bloomfield 1984, Brown and Hobson 1998).

RPC Winter Range: Core and Mountain Areas

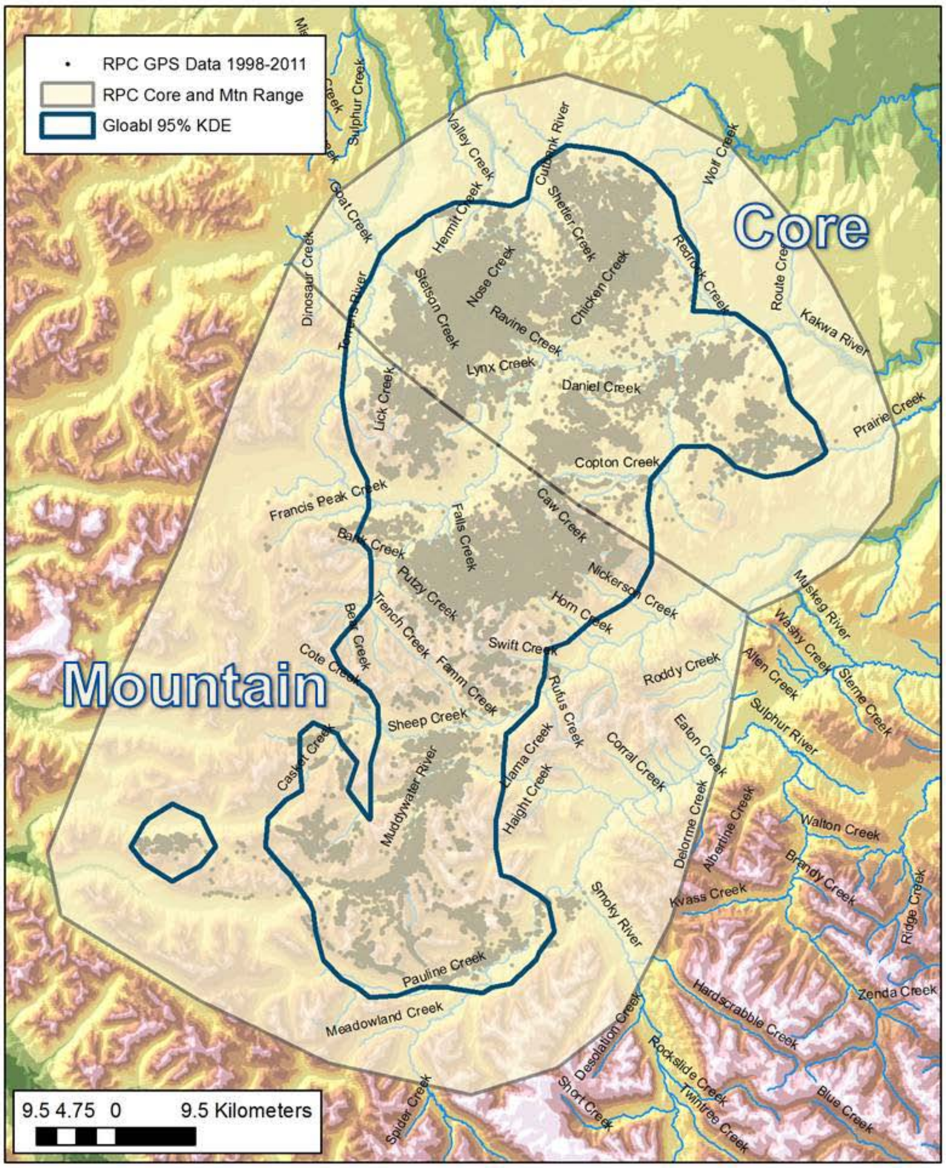
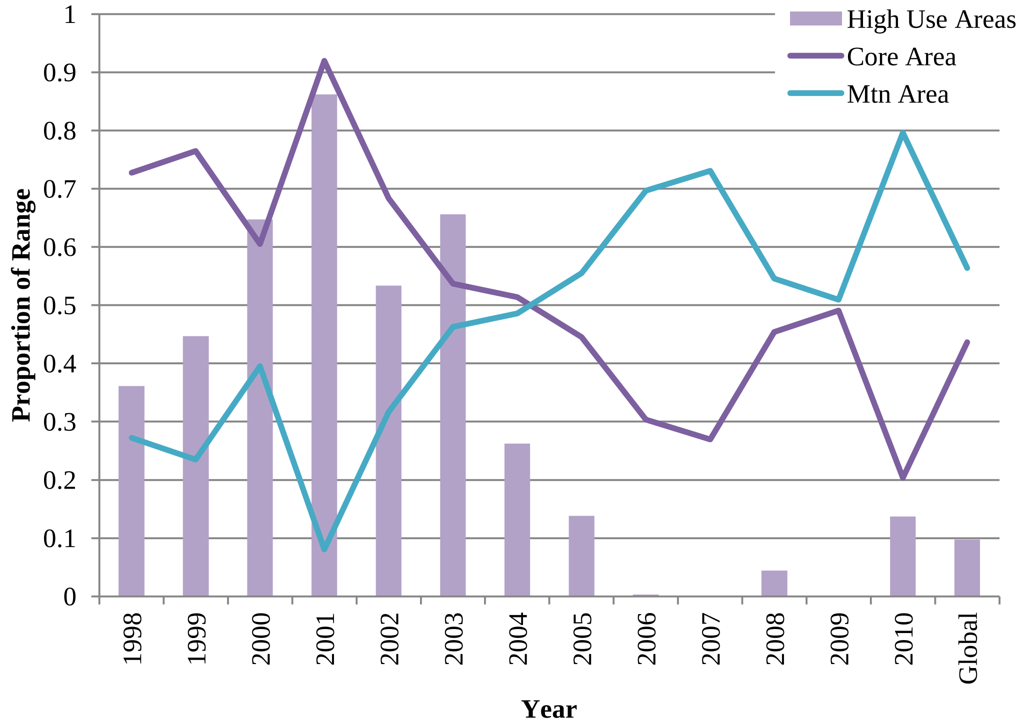


Figure 3.10: Proportion of annual and global 95% KDE range within the core and mountain areas of the RPC winter range in west-central Alberta from 1998-2010 (lines). Proportion of predicted high use areas occurring with the core area of the RPC winter range (bars).



7. Literature Cited

Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA). 2010. Status of the Woodland Caribou (*Rangifer tarandus caribou*) in Alberta: Update 2010., Alberta Sustainable Resource Development, Edmonton, AB.

Alldredge, J. R. and J. Griswold. 2006. Design and analysis of resource selection studies for categorical resource variables. Journal of Wildlife Management 70:337-346.

Alldredge, J. R. and J. T. Ratti. 1992. Further comparison of some statistical techniques for analysis of resource selection. Journal of Wildlife Management 56:1-9.

Alldredge, J. R., D. L. Thomas, and L. L. McDonald. 1998. Survey and comparison of methods for study of resource selection. Journal of Agricultural Biological and Environmental Statistics 3:237-253.

Anderson, D. R., K. P. Burnham, and W. L. Thompson. 2000. Null hypothesis testing: problems, prevalence, and an alternative. Journal of Wildlife Management 64:912-923.

Arthur, S. M., B. F. J. Manly, L. L. McDonald, and G. W. Garner. 1996. Assessing habitat selection when availability changes. Ecology 77:215-227.

Beckmann, J. P., K. Murray, R. G. Seidler, and J. Berger. 2012. Human-mediated shifts in animal habitat use: Sequential changes in pronghorn use of a natural gas field in Greater Yellowstone. Biological Conservation 147:222-233.

Bergerud, A. T. and J. P. Elliot. 1986. Dynamics of Caribou and Wolves in Northern British-Columbia. Canadian Journal of Zoology-Revue Canadienne De Zoologie 64:1515-1529.

Bergerud, A. T. and R. E. Page. 1987. Displacement and Dispersion of Parturient Caribou at Calving as Antipredator Tactics. Canadian Journal of Zoology-Revue Canadienne De Zoologie 65:1597-1606.

Boyce, M. S. 2006. Scale for resource selection functions. Diversity and Distributions 12:269-276.

Boyce, M. S., J. S. Mao, E. H. Merrill, D. Fortin, M. G. Turner, J. Fryxell, and P. Turchin. 2003. Scale and heterogeneity in habitat selection by elk in Yellowstone National Park. Ecoscience 10:421-431.

Boyce, M. S., P. R. Vernier, S. E. Nielsen, and F. K. A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modelling 157:281-300.

Brown, W. K. and D. P. Hobson. 1998. Caribou in west-central Alberta: information review and synthesis. Prepared for The Research Subcommittee of the West-central Alberta Caribou Standing Committee. Terrestrial & Aquatic Environmental Managers Ltd., Calgary.

Burnham, K. P. and D. R. Anderson. 2002. Model selection and multimodal inference: a practical information theoretic approach. 2nd edition. Springer-Verlag, New York, NY.

Carroll, C., R. E. Noss, P. C. Paquet, and N. H. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications 13:1773-1789.

Ciarniello, L. M., M. S. Boyce, D. R. Seip, and D. C. Heard. 2007. Grizzly bear habitat selection is scale dependent. Ecological Applications 17:1424-1440.

D'Eon, R. G. and D. Delparte. 2005. Effects of radio-collar position and orientation on GPS radio-collar performance, and the implications of PDOP in data screening. Journal of Applied Ecology 42:383-388.

D'Eon, R. G., R. Serrouya, G. Smith, and C. O. Kochanny. 2002. GPS radiotelemetry error and bias in mountainous terrain. Wildlife Society Bulletin 30:430-439.

DeCesare, N. J. 2012. Resource selection, predation risk, and population dynamics of woodland caribou. PhD Thesis. University of Montana, Missoula, MT.

DeCesare, N. J., M. Hebblewhite, F. Schmiegelow, D. Hervieux, G. J. McDermid, L. Neufeld, M. Bradley, J. Whittington, K. G. Smith, L. E. Morgantini, M. Wheatley, and M. Musiani. 2012. Transcending scale dependence in identifying habitat with resource selection functions. Ecological Applications 22:1068-1083.

Dunford, J. S., P. D. McLoughlin, F. Dalerum, and S. Boutin. 2006. Lichen abundance in the peatlands of northern Alberta: Implications for boreal caribou. Ecoscience 13:469-474.

Dyer, S. J. 1999. Movement and distribution of woodland caribou (*Rangifer tarandus caribou*) in response to industrial development in northeastern Alberta. M.Sc. Thesis. University of Alberta, Edmonton, AB.

Dyer, S. J., J. P. O'Neill, S. M. Wasel, and S. Boutin. 2001. Avoidance of industrial development by woodland caribou. Journal of Wildlife Management 65:531-542.

Dzus, E. 2001. Status of the woodland caribou (*Rangifer tarandus caribou*) in Alberta. . Wildlife Status Report. Alberta Environment, Fisheries and Management Division, and Alberta Conservation Association.

Edmonds, E. J. and M. Bloomfield. 1984. A study of woodland caribou (*Rangifer tarandus caribou*) in west-central Alberta, 1979-1983., Alberta Energy and Natural Resources Fish and Wildlife Division.

Environment Canada. 2008. Scientific Review for the Identification of Critical Habitat for Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population, in Canada. Environment Canada, Ottawa.

Erickson, W. P., T. L. McDonald, K. G. Gerow, S. Howlin, and J. W. Kern. 2001. Statistical issues in resource selection studies with radio-marked animals. Pages 211-245 *in* J. J. Millspaugh and J. J. Marzluff, editors. Radio tracking and animal populations. Academic, San Diego, California, USA.

Festa-Bianchet, M., J. C. Ray, S. Boutin, S. Cote, and A. Gunn. 2011. Conservation of caribou (Rangifer tarandus) in Canada: an uncertain future. Canadian Journal of Zoology-Revue Canadienne De Zoologie 89:419-434.

Fielding, A. H. and J. F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 23:38-49.

Frair, J. L., J. Fieberg, M. Hebblewhite, F. Cagnacci, N. J. DeCesare, and L. Pedrotti. 2010. Resolving issues of imprecise and habitat-biased locations in ecological analyses using GPS telemetry data. Philosophical Transactions of the Royal Society B-Biological Sciences 365:2187-2200.

Frair, J. L., S. E. Nielsen, E. H. Merrill, S. R. Lele, M. S. Boyce, R. H. M. Munro, G. B. Stenhouse, and H. L. Beyer. 2004. Removing GPS collar bias in habitat selection studies. Journal of Applied Ecology 41:201-212.

Garshelis, D. L. 2000. Delusions in habitat evaluation: measuring use, selection, and importance. Pages 111-164 *in* L. Biotani and T. K. Fuller, editors.

Research Techniques in Animal Ecology: Controversies and Consequences. Columbia University Press, New York.

Gillies, C. S., M. Hebblewhite, S. E. Nielsen, M. A. Krawchuk, C. L. Aldridge, J. L. Frair, D. J. Saher, C. E. Stevens, and C. L. Jerde. 2006. Application of random effects to the study of resource selection by animals. Journal of Animal Ecology 75:887-898.

Gustine, D. D. and K. L. Parker. 2008. Variation in the seasonal selection of resources by woodland caribou in northern British Columbia. Canadian Journal of Zoology-Revue Canadienne De Zoologie 86:812-825.

Gustine, D. D., K. L. Parker, R. J. Lay, M. P. Gillingham, and D. C. Heard. 2006. Interpreting resource selection at different scales for woodland caribou in winter. Journal of Wildlife Management 70:1601-1614.

Hall, D. K., G. A. Riggs, and V. V. Salomonson. 2006. MODIS/Terra Snow Cover 8-day L3 Global 500m Grid V005, February 2000 to April 2011. Digital media. National Snow and Ice Data Center, Boulder, CO, USA.

Hall, L. S., P. R. Krausman, and M. L. Morrison. 1997. The habitat concept and a plea for standard terminology. Wildlife Society Bulletin 25:173-182.

Harju, S. M., M. R. Dzialak, R. G. Osborn, L. D. Hayden-Wing, and J. B. Winstead. 2011. Conservation planning using resource selection models: altered selection in the presence of human activity changes spatial prediction of resource use. Animal Conservation 14:502-511.

Hebblewhite, M. 2011. Effects of Energy Development on Ungulates. Pages 71-94 *in* D. E. Naugle and M. S. Boyce, editors. Energy Development and Wildlife Conservation in Western North America. Island Press, Washington, DC, USA.

Hebblewhite, M. and E. Merrill. 2008. Modelling wildlife-human relationships for social species with mixed-effects resource selection models. Journal of Applied Ecology 45:834-844.

Hebblewhite, M., M. Percy, and E. H. Merrill. 2007. Are all global positioning system collars created equal? Correcting habitat-induced bias using three brands in the Central Canadian Rockies. Journal of Wildlife Management 71:2026-2033.

Hins, C., J. P. Ouellet, C. Dussault, and M. H. St-Laurent. 2009. Habitat selection by forest-dwelling caribou in managed boreal forest of eastern Canada: Evidence of a landscape configuration effect. Forest Ecology and Management 257:636-643.

Houle, M., D. Fortin, C. Dussault, R. Courtois, and J. P. Ouellet. 2010. Cumulative effects of forestry on habitat use by gray wolf (Canis lupus) in the boreal forest. Landscape Ecology 25:419-433.

James, A. R. C., S. Boutin, D. M. Hebert, and A. B. Rippin. 2004. Spatial separation of caribou from moose and its relation to predation by wolves. Journal of Wildlife Management 68:799-809.

James, A. R. C. and A. K. Stuart-Smith. 2000. Distribution of caribou and wolves in relation to linear corridors. Journal of Wildlife Management 64:154-159.

Johnson, C. J. and M. P. Gillingham. 2008. Sensitivity of species-distribution models to error, bias, and model design: An application to resource selection functions for woodland caribou. Ecological Modelling 213:143-155.

Johnson, C. J., S. E. Nielsen, E. H. Merrill, T. L. McDonald, and M. S. Boyce. 2006. Resource selection functions based on use-availability data: Theoretical motivation and evaluation methods. Journal of Wildlife Management 70:347-357.

Johnson, C. J., D. R. Seip, and M. S. Boyce. 2004. A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. Journal of Applied Ecology 41:238-251.

Johnson, D. H. 1980. The Comparison of Usage and Availability Measurements for Evaluating Resource Preference. Ecology 61:65-71.

Kinley, T. A. and C. D. Apps. 2001. Mortality patterns in a subpopulation of endangered mountain caribou. Wildlife Society Bulletin 29:158-164.

Latham, A. D. M., M. C. Latham, M. S. Boyce, and S. Boutin. 2011. Movement responses by wolves to industrial linear features and their effect on woodland caribou in northeastern Alberta. Ecological Applications 21:2854-2865.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource selection by animals : statistical design and analysis for field studies. 2nd edition. Kluwer Academic Publishers, Dordrecht ; Boston.

McClean, S. A., M. A. Rumble, R. M. King, and W. L. Baker. 1998. Evaluation of resource selection methods with different definitions of availability. Journal of Wildlife Management 62:793-801.

McLoughlin, P. D., E. Vander Wal, S. J. Lowe, B. R. Patterson, and D. L. Murray. 2011. Seasonal shifts in habitat selection of a large herbivore and the influence of human activity. Basic and Applied Ecology 12:654-663.

Mohr, C. O. 1947. Table of Equivalent Populations of North American Small Mammals. American Midland Naturalist 37:223-249.

Moreau, G., D. Fortin, S. Couturier, and T. Duchesne. 2012. Multi-level functional responses for wildlife conservation: the case of threatened caribou in managed boreal forests. Journal of Applied Ecology 49:611-620.

Mosnier, A., J. P. Ouellet, and R. Courtois. 2008. Black bear adaptation to low productivity in the boreal forest. Ecoscience 15:485-497.

Natural Regions Committee. 2006. Natural Regions and Subregions of Alberta. Compiled by D.J. Downing and W.W. Pettapiece. Pub. No. T/852, Government of Alberta.

Neufeld, L. 2006. Spatial dynamics of wolves and woodland caribou in an industrial forest landscape in west-central Alberta. M.Sc. Thesis. University of Alberta, Edmonton, AB.

Oberg, P. R. 2001. Responses of mountain caribou to linear features in a west-central Alberta landscape. M.Sc Thesis, University of Alberta, Edmonton, AB.

Polfus, J. L., M. Hebblewhite, and K. Heinemeyer. 2011. Identifying indirect habitat loss and avoidance of human infrastructure by northern mountain woodland caribou. Biological Conservation 144:2637-2646.

Racey, G. D. and A. A. Arsenault. 2007. In search of a critical habitat concept for woodland caribou, boreal population. Rangifer Special Issue 17:29-37.

Rempel, R. S., A. R. Rodgers, and K. F. Abraham. 1995. Performance of a GPS animal location system under boreal forest canopy. Journal of Wildlife Management 59:543-551.

Rettie, W. J. and F. Messier. 2000. Hierarchical habitat selection by woodland caribou: its relationship to limiting factors. Ecography 23:466-478.

Saher, D. J. 2005. Woodland caribou habitat selection during winter and along migratory routes in west-central Alberta. M.Sc. Thesis. University of Alberta, Edmonton, AB.

Saher, D. J. and F. K. A. Schmiegelow. 2005. Movement pathways and habitat selection by woodland caribou during spring migration. Rangifer Special Issue 16:143-153.

Sawyer, H., M. J. Kauffman, and R. M. Nielson. 2009. Influence of Well Pad Activity on Winter Habitat Selection Patterns of Mule Deer. Journal of Wildlife Management 73:1052-1061.

Schaefer, J. A., C. M. Bergman, and S. N. Luttich. 2000. Site fidelity of female caribou at multiple spatial scales. Landscape Ecology 15:731-739.

Schaefer, J. A. and S. P. Mahoney. 2007. Effects of progressive clearcut logging on Newfoundland Caribou. Journal of Wildlife Management 71:1753-1757.

Schindler, D. W. and P. G. Lee. 2010. Comprehensive conservation planning to protect biodiversity and ecosystem services in Canadian boreal regions under a warming climate and increasing exploitation. Biological Conservation 143:1571-1586.

Seip, D. R. 1992. Factors Limiting Woodland Caribou Populations and Their Interrelationships with Wolves and Moose in Southeastern British-Columbia. Canadian Journal of Zoology-Revue Canadienne De Zoologie 70:1494-1503.

Seip, D. R., C. J. Johnson, and G. S. Watts. 2007. Displacement of mountain caribou from winter habitat by snowmobiles. Journal of Wildlife Management 71:1539-1544.

Smith, K. G. 2004. Woodland caribou demography and presistence relative to landscape change in west-central Alberta. M.Sc. Thesis. University of Alberta, Edmonton, AB.

Smith, K. G., E. J. Ficht, D. Hobson, T. C. Sorensen, and D. Hervieux. 2000. Winter distribution of woodland caribou in relation to clear-cut logging in west-central Alberta. Canadian Journal of Zoology-Revue Canadienne De Zoologie 78:1433-1440.

Sorensen, T., P. D. McLoughlin, D. Hervieux, E. Dzus, J. Nolan, B. Wynes, and S. Boutin. 2008. Determining sustainable levels of cumulative effects for boreal caribou. Journal of Wildlife Management 72:900-905.

Szkorupa, T. and F. K. A. Schmiegelow. 2003. Multi-scale habitat selection by mountain caribou in West Central Alberta. Rangifer Special Issue 14:293-294.

Thomas, D. C. and D. R. Gray. 2002. Update COSEWIC status report on the woodland caribou Rangifer tarandus caribou in Canada, in COSEWIC assessment and update status report on the woodland caribou Rangifer tarandus caribou in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa.

Thomas, D. L. and E. J. Taylor. 2006. Study designs and tests for comparing resource use and availability II. Journal of Wildlife Management 70:324-336.

Tracz, B. V., J. M. LaMontagne, E. M. Bayne, and S. Boutin. 2010. Annual and monthly range fidelity of female boreal woodland caribou in response to petroleum development. Rangifer 30:31-44.

Van Horne, B. 1983. Density as a misleading indication of habitat quality. Journal of Wildlife Management 47:893-901.

Vistnes, I. and C. Nellemann. 2008. The matter of spatial and temporal scales: a review of reindeer and caribou response to human activity. Polar Biology 31:399-407.

Vors, L. S. and M. S. Boyce. 2009. Global declines of caribou and reindeer. Global Change Biology 15:2626-2633.

Vors, L. S., J. A. Schaefer, B. A. Pond, A. R. Rodgers, and B. R. Patterson. 2007. Woodland caribou extirpation and anthropogenic landscape disturbance in Ontario. Journal of Wildlife Management 71:1249-1256.

Webb, S. L., M. R. Dzialak, J. J. Wondzell, S. M. Harju, L. D. Hayden-Wing, and J. B. Winstead. 2011. Survival and cause-specific mortality of female Rocky Mountain elk exposed to human activity. Population Ecology 53:483-493.

White, G. C. and R. A. Garrott. 1990. Analysis of wildlife radio-tracking data. Academic Press, San Diego.

Wittmer, H. U., B. N. McLellan, and F. W. Hovey. 2006. Factors influencing variation in site fidelity of woodland caribou (Rangifer tarandus caribou) in southeastern British Columbia. Canadian Journal of Zoology-Revue Canadienne De Zoologie 84:537-545.

Wittmer, H. U., B. N. McLellan, R. Serrouya, and C. D. Apps. 2007. Changes in landscape composition influence the decline of a threatened woodland caribou population. Journal of Animal Ecology 76:568-579.

Chapter 4: General Discussion

1. Synthesis

In a time of increasing land use intensity on many landscapes, there is a need for effective management of resource development to maintain biodiversity, including species at risk. Effective management includes the application of reliable knowledge in an appropriate decision-support system. Woodland caribou are of significant conservation concern in both Alberta (Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA) 2010) and Canada (Environment Canada 2011), due to declining population sizes. Recovery strategies have been developed for jurisdictions where caribou are listed as threatened, with the overarching goal of recovering populations to self-sustaining levels. Definition of critical habitat is at the heart of these recovery plans, and remains a key component for addressing issues related to woodland caribou decline. In this thesis, I attempted to address research gaps regarding woodland caribou conservation in Alberta to support the recovery planning process. Specifically, I examined the application of animal location data for delineating population ranges and understanding how caribou respond to landscape change.

I investigated the effect of sampling regime on delineation of population ranges for woodland caribou in the boreal and mountain ecotypes in Alberta. Previous research has addressed sampling requirements to effectively address individual animal home ranges (Girard et al. 2002, Borger et al. 2006). However, recovery plans require delineation of local population ranges. I estimated population range size for eight caribou local populations in Alberta, compared two different home range estimation methods, and addressed the role of sampling regime when defining population ranges. Several range estimates failed to approach asymptotes, indicating sample size was inadequate for those ranges, and that resultant range delineation may not represent the population.

Depending on the population and estimation method, up to 26 caribou with annual data were required for stable population range estimates. I compared the minimum convex polygon (MCP) and fixed kernel density (KDE) methods, which required a minimum of 22 caribou with a 14 day sampling frequency, and 8 caribou with a 1 day sampling frequency, respectively. My results suggest that adequate sampling of the local population must address both temporal and spatial variability in range use to provide a representative population range estimate. MCP estimation of population ranges represents a precautionary approach to range delineation, and is appropriate for use in broad-scale conservation and management, whereas KDE of core areas within a population range can augment understanding of range use.

Understanding the response to natural and anthropogenic disturbance by woodland caribou is critical if populations are to persist on the landscape. Disturbance within caribou ranges is increasing as resource exploration and development expands into previously undisturbed habitats (Schindler and Lee 2010). Resource selection functions have been identified as a valuable tool for examining wildlife response to disturbance because they are able to quantify important habitat attributes (McLoughlin et al. 2011, Beckmann et al. 2012). Selection of habitats is measured relative to availability, however availability changes over space and time, therefore incorporation of landscape change is crucial to understanding and predicting woodland caribou resource selection over time. I examined the resource selection patterns of caribou in the Redrock Prairie Creek range of west-central Alberta, during the winter season, over a 13-year period (1998-2011).

At a coarse-scale within their winter range, caribou avoided areas of high cutblock density and burnt areas, but were associated in some years with areas of high linear feature density (roads and pipelines). At a fine-scale, caribou generally avoided areas with a higher density of anthropogenic disturbances, including cutblocks, roads and pipelines, seismic lines, and wellsites, as well as burnt areas.

I developed a dynamic model to account for landscape change over time while including caribou sampled from 1998-2011. This model demonstrated better predictive performance than a static model that was naive to landscape change. I also documented a shift in RPC winter range use. From 1998-2004, caribou had a higher proportion of their winter range in the historic foothills core area. After 2004, RPC caribou shifted their range use to include higher proportions of the less disturbed mountainous region of the winter range. Over the same time period, the proportion of predicted high use habitat shifted from the core region to the mountain region, reflecting increased disturbance in historically occupied areas. Finally, government surveys indicate that between 2003-04 and

2004-05, a dramatic decline in estimated female population size occurred in the RPC range, which may be associated with the observed change in range use.

Results from my work can support the recovery planning process for woodland caribou by addressing issues related to range delineation for local populations and incorporating landscape change when quantifying resource selection patterns.

2. Management Recommendations

Woodland caribou recovery is multi-faceted and represents an immense challenge for conservation in Alberta and across Canada. Results from this study can support sound decision-making for the conservation of woodland caribou in Alberta, with possible extensions to other jurisdictions in Canada.

Population range analyses showed that the number of animals and sampling frequency required for appropriate delineation of population range is variable. Sample size considerations should be addressed by examining when population range estimates asymptote (Environment Canada 2011), and the number of animals required to represent the population is likely to vary with population size. Alberta populations are considered data sufficient by Environment Canada (2011). However, my assessment of population range estimates indicate that boreal caribou populations require increased number of animals sampled to provide unbiased population ranges (i.e. the estimates did not asymptote, suggesting that range size is underestimated). In contrast, data from west-central Alberta suggest that sample sizes are adequate for delineating central mountain population ranges. Increased attention to less known populations in

Alberta, such as Caribou Mountains, Red Earth or areas of Wood Buffalo National Park is needed. In addition, augmenting sampling in populations such as Slave Lake or East Side of the Athabasca at the southern extent of boreal caribou occurrence in Alberta would provide a more robust foundation for delineating conservation units in areas of high disturbance and identify areas needed to maintain connectively between populations. Finally, concerted effort should be made to develop and maintain integrated databases for caribou ranges that cross jurisdictional boundaries. Coordinated management and conservation planning is necessary for trans-boundary populations, including the Chinchaga, Narraway and Redrock Prairie Creek populations in Alberta.

A product of RSF modelling is spatially explicit prediction maps that identify where caribou are likely to occur. Prediction maps based on dynamic models can be used to identify areas with high potential for caribou use over time, and conversely, areas where industrial development might be concentrated to minimize effects on caribou. I illustrated that the RPC population shifted their winter range over a 13 year sampling period in response to increasing disturbance. Management initiatives should include efforts to minimize further industrial disturbance within the RPC core winter range and application of restoration measures to recover areas where caribou were once present.

Population range use and resource selection patterns are dynamic in space and time, requiring a sustained monitoring effort. I have illustrated that long-term GPS datasets can provide valuable knowledge when studying a threatened species. However, radio collaring is invasive and stressful to animals, therefore clear objectives should be established to support additional collaring efforts. There are tradeoffs between broad-scale sampling for range delineation and more targeted efforts to address detailed use and movement patterns that inform appropriate sampling strategy.

Recovery plans and status reports in Alberta continue to suggest the ultimate cause of woodland caribou decline is loss of habitat due to industrial disturbance (Alberta Woodland Caribou Recovery Team 2005, Alberta Caribou Committee 2008, ASRD and ACA 2010). Actions, policies and strategies to address habitat loss and maintenance of intact habitat need to be implemented if populations are to recover. As suggested by the West-Central Landscape Planning Team, an adaptive management approach to caribou recovery and conservation will ensure strategies involving habitat intactness, predator and prey control, and management of industrial footprint are assessed for their effectiveness and adjusted where necessary to support caribou recovery (Alberta Caribou Committee 2008).

3. Future Research and Limitations

This research has highlighted several issues related to the use of location data in conservation planning and provided a foundation for future research. Further development of models to examine which component of the sampling regime has more influence on population range estimates should be undertaken, including determining appropriate sample sizes and number of years sampled to delineate valid population ranges. Future work should also consider how to incorporate seasonal variation when delineating local populations. Sample size and sampling frequency requirements for boreal populations should be re-examined when additional GPS data are available. Finally, in addition to defining populations based on location data and expert opinion, conservation units should also consider metapopulation genetic structure and identify landscape elements that maintain connectivity among caribou populations (Weckworth et al. 2012).

Future development of RSFs for conservation planning should incorporate landscape change to provide an appropriate depiction of resource selection over time (Beckmann et al. 2012). To understand impacts of rapid resource development, it is crucial that data about industrial disturbance become more readily available and reliable, in order to support a rigorous approach to predicting caribou response. In recognition of the limitation of resource selection modeling to provide a direct link to habitat quality, future work should also incorporate concurrent measurement of population parameters such as adult survival and calf recruitment (Van Horne 1983, DeCesare 2012). Predictions based on potential futures in consideration of land-use plans and climate change could also be included to provide an indication on how caribou may respond to future landscape change. Finally, understanding the impact of range shift or contraction on population demography must be enhanced.

4. Literature Cited

Alberta Caribou Committee. 2008. Recommendations for a West Central Alberta Caribou Landscape Plan. Prepared by the Alberta Caribou Committee Governance Board. Edmonton, AB.

Alberta Sustainable Resource Development and Alberta Conservation Association (ASRD and ACA). 2010. Status of the Woodland Caribou (*Rangifer* *tarandus caribou*) in Alberta: Update 2010., Alberta Sustainable Resource Development, Edmonton, AB.

Alberta Woodland Caribou Recovery Team. 2005. Alberta woodland caribou recovery plan, 2004/05-2013/14. Alberta Sustainable Resource Development, Fish and Wildlife Division, Alberta Species at Risk Recovery Plan No. 4, Edmonton, AB. 48 pp.

Beckmann, J. P., K. Murray, R. G. Seidler, and J. Berger. 2012. Human-mediated shifts in animal habitat use: Sequential changes in pronghorn use of a natural gas field in Greater Yellowstone. Biological Conservation 147:222-233.

Borger, L., N. Franconi, G. De Michele, A. Gantz, F. Meschi, A. Manica, S. Lovari, and T. Coulson. 2006. Effects of sampling regime on the mean and variance of home range size estimates. Journal of Animal Ecology 75:1393-1405.

DeCesare, N. J. 2012. Resource selection, predation risk, and population dynamics of woodland caribou. PhD Thesis. University of Montana, Missoula, MT.

Environment Canada. 2011. Scientific Assessment to Inform the Identification of Critical Habitat for Woodland Caribou (*Rangifer tarandus caribou*), Boreal population, in Canada: 2011 Update. Ottawa, Ontario, Canada. 102 pp. plus appendices.

Girard, I., J. P. Ouellet, R. Courtois, C. Dussault, and L. Breton. 2002. Effects of sampling effort based on GPS telemetry on home-range size estimations. Journal of Wildlife Management 66:1290-1300.

McLoughlin, P. D., E. Vander Wal, S. J. Lowe, B. R. Patterson, and D. L. Murray. 2011. Seasonal shifts in habitat selection of a large herbivore and the influence of human activity. Basic and Applied Ecology 12:654-663.

Schindler, D. W. and P. G. Lee. 2010. Comprehensive conservation planning to protect biodiversity and ecosystem services in Canadian boreal regions under a warming climate and increasing exploitation. Biological Conservation 143:1571-1586.

Van Horne, B. 1983. Density as a misleading indication of habitat quality. Journal of Wildlife Management 47:893-901.

Weckworth, B. V., M. Musiani, A. D. McDevitt, M. Hebblewhite, and S. Mariani. 2012. Reconstruction of caribou evolutionary history in Western North

America and its implications for conservation. Molecular Ecology 21:3610-3624.

Appendix A - Chapter 3 Additional Information

Table A.1: Redrock Prairie Creek GPS location data for caribou available during the 1998-2011 winter seasons (December 1 to April 30 of each year). Individual caribou may appear in multiple years.

Winter	ID	Start Date	End Date	Number of Locations
1998/99	F300	1-Dec-98	30-Apr-99	696
	F301	1-Dec-98	30-Apr-99	761
	F306	10-Dec-98	30-Apr-99	367
	F309	1-Dec-98	30-Apr-99	737
	F324	1-Dec-98	29-Apr-99	713
1999/00	F306	2-Dec-99	4-Jan-00	56
	F309	1-Dec-99	30-Apr-00	631
	F325	1-Dec-99	30-Apr-00	635
	F326	1-Dec-99	28-Jan-00	163
	F329	1-Dec-99	30-Apr-00	608
	F330	1-Dec-99	30-Apr-00	537
	F331	1-Dec-99	30-Apr-00	351
	F332	1-Dec-99	30-Apr-00	631
	F334	1-Dec-99	30-Apr-00	621
2000/01	F332	1-Dec-00	31-Jan-01	345
	F334	1-Dec-00	5-Apr-01	708
	F335	1-Dec-00	29-Mar-01	665
	F336	1-Dec-00	30-Apr-01	819
	F337	1-Dec-00	30-Apr-01	839
	F338	1-Dec-00	30-Apr-01	703
	F339	1-Dec-00	30-Apr-01	730
	F341	1-Dec-00	1-Jan-01	185
	F342	1-Dec-00	30-Apr-01	814
	F343	1-Dec-00	30-Apr-01	834
	F344	1-Dec-00	30-Apr-01	814
	F346	1-Dec-00	30-Apr-01	824
	F347	1-Dec-00	30-Apr-01	830
	F348	1-Dec-00	30-Apr-01	837
	F349	1-Dec-00	30-Apr-01	809
	F350	1-Dec-00	30-Apr-01	772

Winter	ID	Start Date	End Date	Number of Locations
2001/02	F346	1-Dec-01	30-Apr-02	250
	F352	1-Dec-01	30-Apr-02	640
	F353	1-Dec-01	30-Apr-02	694
	F354	1-Dec-01	30-Apr-02	723
	F355	1-Dec-01	30-Apr-02	711
	F356	1-Dec-01	29-Apr-02	707
	F357	1-Dec-01	30-Apr-02	629
	F358	1-Dec-01	29-Apr-02	704
	F359	3-Dec-01	26-Apr-02	672
	F360	1-Dec-01	30-Apr-02	512
2002/03	F352	1-Dec-02	30-Apr-03	653
	F353	1-Dec-02	30-Apr-03	652
	F354	1-Dec-02	30-Apr-03	628
	F355	1-Dec-02	30-Apr-03	738
	F361	1-Dec-02	30-Apr-03	526
	F362	1-Dec-02	30-Apr-03	523
	F364	1-Dec-02	30-Apr-03	547
	F365	1-Dec-02	30-Apr-03	796
	F366	1-Dec-02	2-Mar-03	489
	F367	1-Dec-02	30-Apr-03	513
2003/04	F355	1-Dec-03	14-Feb-04	126
	F365	1-Dec-03	26-Jan-04	302
	F369	1-Dec-03	30-Apr-04	796
	F370	1-Dec-03	30-Apr-04	884
	F371	1-Dec-03	30-Apr-04	647
	F372	1-Dec-03	30-Apr-04	762
2004/05	F369	1-Dec-04	30-Apr-05	752
	F371	1-Dec-04	12-Jan-05	216
	F373	1-Dec-04	30-Apr-05	711
	F374	1-Dec-04	30-Apr-05	852

Table A.1: Continued.

Winter	ID	Start Date	End Date	Number of Locations
2004/05	F378	1-Dec-04	12-Mar-05	497
	F379	1-Dec-04	19-Mar-05	636
	F380	1-Dec-04	16-Mar-05	505
	F381	1-Dec-04	30-Apr-05	899
	F382	1-Dec-04	30-Apr-05	834
2005/06	F383	1-Dec-05	30-Apr-06	897
	F384	1-Dec-05	30-Apr-06	886
	F385	1-Dec-05	30-Apr-06	869
	F386	1-Dec-05	30-Apr-06	886
	F387	1-Dec-05	30-Apr-06	854
	F389	1-Dec-05	28-Apr-06	682
	F391	1-Dec-05	30-Apr-06	799
	F392	12-Dec-05	30-Apr-06	760
2006/07	F383	1-Dec-06	17-Feb-07	471
	F384	2-Dec-06	30-Apr-07	881
	F385	2-Dec-06	30-Apr-07	890
	F387	1-Dec-06	30-Apr-07	900
	F391	1-Dec-06	30-Apr-07	842
	F392	1-Dec-06	30-Apr-07	835
	F393	1-Dec-06	30-Apr-07	878
	F396	1-Dec-06	30-Apr-07	865
	F397	1-Dec-06	30-Apr-07	884
	F398	1-Dec-06	30-Apr-07	883
2007/08	F332	1-Dec-07	30-Apr-08	858
	F393	1-Dec-07	30-Apr-08	877
	F397	1-Dec-07	30-Apr-08	881
	F398	1-Dec-07	27-Apr-08	790
	F404	9-Dec-07	30-Apr-08	788
	F408	1-Dec-07	30-Apr-08	878
	F409	1-Dec-07	30-Apr-08	865
	F410	6-Dec-07	30-Apr-08	868

Winter	ID	Start Date	End Date	Number of Locations
2008/09	F332	1-Dec-08	30-Apr-09	874
	F404	1-Dec-08	30-Apr-09	847
	F408	1-Dec-08	30-Apr-09	871
	F409	1-Dec-08	30-Apr-09	903
	F410	1-Dec-08	30-Apr-09	887
	F411	1-Dec-08	30-Apr-09	813
	F413	1-Dec-08	30-Apr-09	807
	F414	1-Dec-08	30-Apr-09	886
2009/10	F408	1-Dec-09	18-Feb-10	472
	F409	1-Dec-09	16-Jan-10	269
	F410	1-Dec-09	13-Jan-10	110
	F413	1-Dec-09	30-Apr-10	846
	F414	1-Dec-09	30-Apr-10	879
	F416	14-Jan-10	30-Apr-10	572
	F417	20-Feb-10	28-Apr-10	383
	F418	16-Jan-10	30-Apr-10	610
2010/11	F416	1-Dec-10	3-Mar-11	500
	F417	1-Dec-10	30-Apr-11	868
	F418	1-Dec-10	30-Apr-11	862
	F421	1-Dec-10	30-Apr-11	843
	F423	1-Dec-10	30-Apr-11	894
	F424	1-Dec-10	30-Apr-11	904
	F425	1-Dec-10	11-Mar-11	604
	F426	1-Dec-10	25-Jan-11	327

Table A.2: A comparison of candidate models for second-order RPC caribou resource selection in winter (December 1 to April 30). Models are ranked by ΔAIC_c values. Akaike weights (w_i) indicate the likelihood of the model being the best of those tested. K indicates the number of parameters, including the intercept in the model.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
1998 Annual Model							
M8	**ln_cut + ln_lin + lc + elev + slope + tpi**	**-3929.12**	**7**	**7834.91**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + elev + slope + tpi	-3967.76	6	7905.52	70.6	2	0.00
M4	lc + elev + slope + tpi	-4016.86	5	7983.72	148.8	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-4129.05	6	8228.10	393.2	4	0.00
M5	ln_tden + elev + slope + tpi	-4181.07	5	8312.14	477.2	5	0.00
M3	elev + slope + tpi	-4225.04	4	8458.09	623.2	6	0.00
M1	ln_tden	-4437.23	2	8884.45	1049.5	7	0.00
M2	ln_cut + ln_lin	-4483.84	3	8997.68	1162.8	8	0.00
1999 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-4395.88**	**8**	**8807.75**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-4361.34	7	8848.68	40.9	2	0.00
M4	lc + snow + elev + slope + tpi	-4466.06	6	8986.13	178.4	3	0.00
M5	ln_tden + elev + slope + tpi	-4637.88	5	9305.75	498.0	4	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-4736.63	6	9527.25	719.5	5	0.00
M3	elev + slope + tpi	-4780.35	4	9578.70	770.9	6	0.00
M1	ln_tden	-5632.36	2	11270.72	2463.0	7	0.00
M2	ln_cut + ln_lin	-5834.80	3	11680.40	2872.6	8	0.00
2000 Annual Model							
M7	**ln_tden + lc + snow + elev + slope + tpi**	**-12493.54**	**7**	**25015.08**	**0.0**	**1**	**1.00**
M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	-12640.44	8	25317.46	302.4	2	0.00
M4	lc + snow + elev + slope + tpi	-12672.42	6	25366.17	351.1	3	0.00
M5	ln_tden + elev + slope + tpi	-12996.84	5	26009.67	994.6	4	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-13076.64	6	26174.61	1159.5	5	0.00
M3	elev + slope + tpi	-13122.57	4	26256.78	1241.7	6	0.00
M1	ln_tden	-15680.31	2	31365.53	6350.4	7	0.00
M2	ln_cut + ln_lin	-15922.96	3	31853.92	6838.8	8	0.00
2001 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-8536.75**	**8**	**17233.49**	**0.0**	**1**	**1.00**
M6	ln_cut + ln_lin + elev + slope + tpi	-8913.59	6	17867.18	633.7	2	0.00
M7	ln_tden + lc + snow + elev + slope + tpi	-9653.17	7	19376.34	2142.9	3	0.00
M2	ln_cut + ln_lin	-9822.66	3	19655.31	2421.8	4	0.00
M5	ln_tden + elev + slope + tpi	-10179.93	5	20384.87	3151.4	5	0.00
M4	lc + snow + elev + slope + tpi	-10245.20	6	20530.40	3296.9	6	0.00
M3	elev + slope + tpi	-10736.44	4	21488.88	4255.4	7	0.00
M1	ln_tden	-10937.07	2	21879.86	4646.4	8	0.00

Table A.2: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
2002 Annual Model							
M7	**ln_tden + lc + snow + elev + slope + tpi**	**-6078.09**	**7**	**12196.18**	**5284.1**	**1**	**1.00**
M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	-6302.07	8	12674.14	5762.1	2	0.00
M5	ln_tden + elev + slope + tpi	-6540.64	5	13121.28	6209.2	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-6731.69	6	13488.37	6576.3	4	0.00
M3	elev + slope + tpi	-6804.04	4	13624.07	6712.0	5	0.00
M1	ln_tden	-8199.98	2	16405.68	9493.6	6	0.00
M2	ln_cut + ln_lin	-8369.28	3	16748.55	9836.5	7	0.00
M4	ln_tden + elev + slope + tpi		Did not converge				
2003 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-3472.04**	**8**	**6912.08**	**0.0**	**1**	**1.00**
M6	ln_cut + ln_lin + elev + slope + tpi	-3727.67	6	7383.34	471.3	2	0.00
M7	ln_tden + lc + snow + elev + slope + tpi	-3749.48	7	7456.96	544.9	3	0.00
M4	lc + snow + elev + slope + tpi	-3757.14	6	7526.29	614.2	4	0.00
M5	ln_tden + elev + slope + tpi	-4023.39	5	8056.78	1144.7	5	0.00
M3	elev + slope + tpi	-4035.05	4	8118.11	1206.0	6	0.00
M2	ln_cut + ln_lin	-4497.24	3	9012.48	2100.4	7	0.00
M1	ln_tden	-4870.05	2	9748.10	2836.0	8	0.00
2004 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-6453.63**	**8**	**12923.26**	**0.0**	**1**	**1.00**
M4	lc + snow + elev + slope + tpi	-6599.98	6	13211.96	288.7	2	0.00
M7	ln_tden + lc + snow + elev + slope + tpi	-6598.56	7	13323.11	399.9	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-7209.16	6	14472.32	1549.1	4	0.00
M5	ln_tden + elev + slope + tpi	-7432.11	5	14874.23	1951.0	5	0.00
M3	elev + slope + tpi	-7433.77	4	14885.53	1962.3	6	0.00
M2	ln_cut + ln_lin	-7985.69	3	15982.19	3058.9	7	0.00
M1	ln_tden	-8154.56	2	16315.12	3391.9	8	0.00
2005 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-7587.96**	**8**	**15047.92**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-7780.40	7	15574.80	526.9	2	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-7761.01	6	15618.02	570.1	3	0.00
M4	lc + snow + elev + slope + tpi	-7787.66	6	15671.31	623.4	4	0.00
M5	ln_tden + elev + slope + tpi	-8011.43	5	16062.87	1015.0	5	0.00
M3	elev + slope + tpi	-8024.24	4	16069.80	1021.9	6	0.00
M2	ln_cut + ln_lin	-8814.25	3	17640.51	2592.6	7	0.00
M1	ln_tden	-9043.91	2	18094.23	3046.3	8	0.00
2006 Annual Model							
M7	**ln_tden + lc + snow + elev + slope + tpi**	**-7641.42**	**7**	**15352.85**	**0.0**	**1**	**1.00**
M4	lc + snow + elev + slope + tpi	-7723.03	6	15486.06	133.2	2	0.00
M8	ln_cut + ln_lin + lc + snow + elev + slope + tpi	-7693.15	8	15546.31	193.5	3	0.00
M5	ln_tden + elev + slope + tpi	-8112.58	5	16250.16	897.3	4	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-8156.31	6	16352.62	999.8	5	0.00
M3	elev + slope + tpi	-8168.91	4	16353.83	1001.0	6	0.00
M2	ln_cut + ln_lin	-10503.84	3	21017.67	5664.8	7	0.00
M1	ln_tden	-11079.71	2	22165.13	6812.3	8	0.00

Table A.2: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
2007 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-7239.92**	**8**	**14351.85**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-7382.92	7	14779.85	428.0	2	0.00
M4	lc + snow + elev + slope + tpi	-7533.39	6	15162.78	810.9	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-7621.74	6	15339.48	987.6	4	0.00
M5	ln_tden + elev + slope + tpi	-7827.55	5	15695.09	1343.2	5	0.00
M3	elev + slope + tpi	-8024.31	4	16069.95	1718.1	6	0.00
M2	ln_cut + ln_lin	-8446.59	3	16905.18	2553.3	7	0.00
M1	ln_tden	-8731.53	2	17469.46	3117.6	8	0.00
2008 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-7595.94**	**8**	**15063.88**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-7572.77	7	15159.54	95.7	2	0.00
M4	lc + snow + elev + slope + tpi	-7663.27	6	15422.54	358.7	3	0.00
M5	ln_tden + elev + slope + tpi	-7974.34	5	15988.67	924.8	4	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-7957.63	6	16011.25	947.4	5	0.00
M3	elev + slope + tpi	-8051.46	4	16124.26	1060.4	6	0.00
M2	ln_cut + ln_lin	-9504.44	3	19020.88	3957.0	7	0.00
M1	ln_tden	-9530.28	2	19066.96	4003.1	8	0.00
2009 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-4675.67**	**8**	**9223.34**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-4698.08	7	9410.16	186.8	2	0.00
M4	lc + snow + elev + slope + tpi	-4703.24	6	9502.49	279.1	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-4922.50	6	9941.00	717.7	4	0.00
M3	elev + slope + tpi	-4971.88	4	9965.09	741.7	5	0.00
M5	ln_tden + elev + slope + tpi	-4968.82	5	9977.64	754.3	6	0.00
M2	ln_cut + ln_lin	-5685.75	3	11383.51	2160.2	7	0.00
M1	ln_tden	-5710.56	2	11427.52	2204.2	8	0.00
2010 Annual Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-6499.28**	**8**	**12870.57**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-6631.21	7	13276.42	405.9	2	0.00
M4	lc + snow + elev + slope + tpi	-6687.01	6	13470.02	599.5	3	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-6807.20	6	13710.40	839.8	4	0.00
M5	ln_tden + elev + slope + tpi	-6906.20	5	13852.40	981.8	5	0.00
M3	elev + slope + tpi	-6933.50	4	13888.34	1017.8	6	0.00
M2	ln_cut + ln_lin	-7679.44	3	15370.87	2500.3	7	0.00
M1	ln_tden	-7930.82	2	15868.03	2997.5	8	0.00

Table A.2: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
Global-Static Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-92336.85**	**8**	**184691.76**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-93661.63	7	187338.83	2647.1	2	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-93891.77	6	187796.70	3104.9	3	0.00
M4	lc + snow + elev + slope + tpi	-94947.72	6	189908.60	5216.8	4	0.00
M5	ln_tden + elev + slope + tpi	-95772.68	5	191556.18	6864.4	5	0.00
M3	elev + slope + tpi	-96650.27	4	193309.08	8617.3	6	0.00
M2	ln_cut + ln_lin	-108543.37	3	217093.06	32401.3	7	0.00
M1	ln_tden	-109746.04	2	219496.24	34804.5	8	0.00
Global-Dynamic Model							
M8	**ln_cut + ln_lin + lc + snow + elev + slope + tpi**	**-105944.02**	**8**	**211905.40**	**0.0**	**1**	**1.00**
M7	ln_tden + lc + snow + elev + slope + tpi	-106031.16	7	212077.37	172.0	2	0.00
M6	ln_cut + ln_lin + elev + slope + tpi	-110695.19	6	221403.16	9497.8	4	0.00
M4	lc + snow + elev + slope + tpi	-106047.70	6	212108.18	202.8	3	0.00
M5	ln_tden + elev + slope + tpi	-110915.40	5	221841.35	9936.0	6	0.00
M3	elev + slope + tpi	-110915.40	4	221839.16	9933.8	5	0.00
M2	ln_cut + ln_lin	-112341.43	3	224689.08	12783.7	7	0.00
M1	ln_tden	-113015.19	2	226034.49	14129.1	8	0.00

Table A.3: A comparison of candidate models for third-order RPC caribou resource selection in winter (December 1 to April 30). Models are ranked by ΔAIC_c values. Akaike weights (w_i) indicate the likelihood of the model being the best of those tested. K indicates the number of parameters, including the intercept in the model.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
1998 Annual Model							
M8	**cut + lin + lc + elev + slope + tpi**	**-2224.61**	**7**	**4463.23**	**0.0**	**1**	**0.81**
M10	cut + lin +well + seismic + lc + elev + slope + tpi	-2224.11	9	4466.23	3.0	2	0.18
M4	lc + elev + slope + tpi	-2231.31	5	4472.62	9.4	3	0.01
M7	tden + lc + elev + slope + tpi	-2230.96	6	4473.93	10.7	4	0.00
M6	cut + lin + elev + slope + tpi	-2367.18	6	4746.36	283.1	5	0.00
M5	tden + elev + slope + tpi	-2375.79	5	4761.58	298.4	6	0.00
M3	elev + slope + tpi	-2377.86	4	4763.72	300.5	7	0.00
M9	cut + lin + well + seismic	-2624.13	5	5258.25	795.0	8	0.00
M2	cut + lin	-2628.15	3	5262.30	799.1	9	0.00
M1	tden	-2631.45	2	5266.91	803.7	10	0.00
1999 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-2735.34**	**9**	**5488.67**	**0.0**	**1**	**0.99**
M8	cut + lin + lc + elev + slope + tpi	-2741.55	7	5497.11	8.4	2	0.01
M7	tden + lc + elev + slope + tpi	-2764.81	6	5541.62	53.0	3	0.00
M4	lc + elev + slope + tpi	-2781.03	5	5572.06	83.4	4	0.00
M6	cut + lin + elev + slope + tpi	-2818.86	6	5649.73	161.1	5	0.00
M5	tden + elev + slope + tpi	-2848.43	5	5706.87	218.2	6	0.00
M3	elev + slope + tpi	-2891.60	4	5791.19	302.5	7	0.00
M2	cut + lin	-3885.30	3	7776.59	2287.9	8	0.00
M9	cut + lin + well + seismic	-3884.87	5	7779.74	2291.1	9	0.00
M1	tden	-3937.22	2	7878.43	2389.8	10	0.00
2000 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-6910.48**	**9**	**13838.97**	**0.0**	**1**	**0.95**
M8	cut + lin + lc + elev + slope + tpi	-6915.37	7	13844.75	5.8	2	0.05
M7	tden + lc + elev + slope + tpi	-6943.89	6	13899.78	60.8	3	0.00
M4	lc + elev + slope + tpi	-6951.60	5	13913.20	74.2	4	0.00
M6	cut + lin + elev + slope + tpi	-7065.93	6	14143.85	304.9	5	0.00
M5	tden + elev + slope + tpi	-7125.73	5	14261.46	422.5	6	0.00
M3	elev + slope + tpi	-7164.33	4	14336.65	497.7	7	0.00
M9	cut + lin + well + seismic	-9196.81	5	18403.62	4564.6	8	0.00
M2	cut + lin	-9198.94	3	18403.88	4564.9	9	0.00
M1	tden	-9291.58	2	18587.17	4748.2	10	0.00

Table A.3: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
2001 Annual Model							
M8	**cut + lin + lc + elev + slope + tpi**	**-3917.84**	**7**	**7849.69**	**0.0**	**1**	**0.87**
M10	cut + lin +well + seismic + lc + elev + slope + tpi	-3917.72	9	7853.44	3.8	2	0.13
M7	tden + lc + elev + slope + tpi	-3954.80	6	7921.61	71.9	3	0.00
M4	lc + elev + slope + tpi	-3966.00	5	7941.99	92.3	4	0.00
M6	cut + lin + elev + slope + tpi	-4076.55	6	8165.10	315.4	5	0.00
M5	tden + elev + slope + tpi	-4115.25	5	8240.50	390.8	6	0.00
M3	elev + slope + tpi	-4137.62	4	8283.24	433.6	7	0.00
M2	cut + lin	-4955.79	3	9917.57	2067.9	8	0.00
M9	cut + lin + well + seismic	-4955.16	5	9920.32	2070.6	9	0.00
M1	tden	-4994.47	2	9992.94	2143.3	10	0.00
2002 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-3829.63**	**9**	**7677.25**	**0.0**	**1**	**0.51**
M8	cut + lin + lc + elev + slope + tpi	-3832.65	7	7677.31	0.1	2	0.49
M7	tden + lc + elev + slope + tpi	-3841.23	6	7694.46	17.2	3	0.00
M4	lc + elev + slope + tpi	-3855.05	5	7720.11	42.9	4	0.00
M6	cut + lin + elev + slope + tpi	-4049.57	6	8109.14	431.9	5	0.00
M5	tden + elev + slope + tpi	-4071.06	5	8152.13	474.9	6	0.00
M3	elev + slope + tpi	-4119.77	4	8247.53	570.3	7	0.00
M2	cut + lin	-5420.71	3	10847.43	3170.2	8	0.00
M9	cut + lin + well + seismic	-5418.77	5	10847.54	3170.3	9	0.00
M1	tden	-5465.43	2	10934.85	3257.6	10	0.00
2003 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-2681.73**	**9**	**5381.46**	**0.0**	**1**	**1.00**
M8	cut + lin + lc + elev + slope + tpi	-2705.67	7	5425.34	43.9	2	0.00
M7	tden + lc + elev + slope + tpi	-2712.07	6	5436.15	54.7	3	0.00
M4	lc + elev + slope + tpi	-2760.34	5	5530.68	149.2	4	0.00
M5	tden + elev + slope + tpi	-2962.72	5	5935.44	554.0	5	0.00
M6	cut + lin + elev + slope + tpi	-2962.78	6	5937.56	556.1	6	0.00
M3	elev + slope + tpi	-3018.71	4	6045.42	664.0	7	0.00
M9	cut + lin + well + seismic	-3617.99	5	7245.97	1864.5	8	0.00
M2	cut + lin	-3640.67	3	7287.34	1905.9	9	0.00
M1	tden	-3643.18	2	7290.36	1908.9	10	0.00
2004 Annual Model							
M8	**cut + lin + lc + elev + slope + tpi**	**-4044.68**	**7**	**8103.36**	**0.0**	**1**	**0.83**
M10	cut + lin +well + seismic + lc + elev + slope + tpi	-4044.23	9	8106.46	3.1	2	0.17
M7	tden + lc + elev + slope + tpi	-4071.14	6	8154.28	50.9	3	0.00
M4	lc + elev + slope + tpi	-4082.70	5	8175.40	72.0	4	0.00
M6	cut + lin + elev + slope + tpi	-4632.58	6	9277.15	1173.8	5	0.00
M5	tden + elev + slope + tpi	-4651.63	5	9313.26	1209.9	6	0.00
M3	elev + slope + tpi	-4676.11	4	9360.22	1256.9	7	0.00
M2	cut + lin	-5582.99	3	11171.98	3068.6	8	0.00
M9	cut + lin + well + seismic	-5582.95	5	11175.89	3072.5	9	0.00
M1	tden	-5592.80	2	11189.60	3086.2	10	0.00

Table A.3: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
2005 Annual Model							
M8	**cut + lin + lc + elev + slope + tpi**	**-3683.80**	**7**	**7381.60**	**0.0**	**1**	**0.75**
M10	cut + lin +well + seismic + lc + elev + slope + tpi	-3682.89	9	7383.77	2.2	2	0.25
M7	tden + lc + elev + slope + tpi	-3703.07	6	7418.14	36.5	3	0.00
M4	lc + elev + slope + tpi	-3707.48	5	7424.96	43.4	4	0.00
M6	cut + lin + elev + slope + tpi	-3757.14	6	7526.28	144.7	5	0.00
M5	tden + elev + slope + tpi	-3781.39	5	7572.78	191.2	6	0.00
M3	elev + slope + tpi	-3790.06	4	7588.11	206.5	7	0.00
M2	cut + lin	-4561.37	3	9128.73	1747.1	8	0.00
M9	cut + lin + well + seismic	-4560.66	5	9131.33	1749.7	9	0.00
M1	tden	-4588.90	2	9181.80	1800.2	10	0.00
2006 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-3573.29**	**9**	**7164.58**	**0.0**	**1**	**1.00**
M7	tden + lc + elev + slope + tpi	-3586.38	6	7184.76	20.2	2	0.00
M8	cut + lin + lc + elev + slope + tpi	-3591.82	7	7197.65	33.1	3	0.00
M4	lc + elev + slope + tpi	-3629.06	5	7268.12	103.5	4	0.00
M5	tden + elev + slope + tpi	-3778.78	5	7567.55	403.0	5	0.00
M6	cut + lin + elev + slope + tpi	-3782.41	6	7576.81	412.2	6	0.00
M3	elev + slope + tpi	-3866.67	4	7741.35	576.8	7	0.00
M9	cut + lin + well + seismic	-6142.80	5	12295.60	5131.0	8	0.00
M2	cut + lin	-6165.96	3	12337.93	5173.3	9	0.00
M1	tden	-6175.03	2	12354.07	5189.5	10	0.00
2007 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-4265.90**	**9**	**8549.81**	**0.0**	**1**	**0.98**
M8	cut + lin + lc + elev + slope + tpi	-4271.76	7	8557.53	7.7	2	0.02
M7	tden + lc + elev + slope + tpi	-4281.66	6	8575.33	25.5	3	0.00
M4	lc + elev + slope + tpi	-4292.19	5	8594.39	44.6	4	0.00
M6	cut + lin + elev + slope + tpi	-4413.17	6	8838.33	288.5	5	0.00
M5	tden + elev + slope + tpi	-4419.81	5	8849.63	299.8	6	0.00
M3	elev + slope + tpi	-4431.48	4	8870.95	321.1	7	0.00
M9	cut + lin + well + seismic	-4861.75	5	9733.50	1183.7	8	0.00
M2	cut + lin	-4869.56	3	9745.11	1195.3	9	0.00
M1	tden	-4874.31	2	9752.61	1202.8	10	0.00
2008 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-3494.22**	**9**	**7006.44**	**0.0**	**1**	**0.94**
M8	cut + lin + lc + elev + slope + tpi	-3499.04	7	7012.08	5.6	2	0.06
M7	tden + lc + elev + slope + tpi	-3542.44	6	7096.88	90.4	3	0.00
M4	lc + elev + slope + tpi	-3553.82	5	7117.64	111.2	4	0.00
M6	cut + lin + elev + slope + tpi	-3659.64	6	7331.28	324.8	5	0.00
M5	tden + elev + slope + tpi	-3737.22	5	7484.43	478.0	6	0.00
M3	elev + slope + tpi	-3772.84	4	7553.68	547.2	7	0.00
M9	cut + lin + well + seismic	-4796.02	5	9602.04	2595.6	8	0.00
M2	cut + lin	-4798.68	3	9603.35	2596.9	9	0.00
M1	tden	-4868.42	2	9740.84	2734.4	10	0.00

Table A.3: Continued.

Model	Candidate Model Parameters	LL	K	AIC_c	ΔAIC_c	Rank	w_i
2009 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-3290.19**	**9**	**6598.38**	**0.0**	**1**	**1.00**
M8	cut + lin + lc + elev + slope + tpi	-3311.11	7	6636.23	37.8	2	0.00
M7	tden + lc + elev + slope + tpi	-3338.22	6	6688.44	90.1	3	0.00
M4	lc + elev + slope + tpi	-3435.99	5	6881.99	283.6	4	0.00
M6	cut + lin + elev + slope + tpi	-4071.77	6	8155.54	1557.2	5	0.00
M5	tden + elev + slope + tpi	-4114.30	5	8238.60	1640.2	6	0.00
M3	elev + slope + tpi	-4290.62	4	8589.25	1990.9	7	0.00
M9	cut + lin + well + seismic	-4670.12	5	9350.24	2751.9	8	0.00
M2	cut + lin	-4688.43	3	9382.86	2784.5	9	0.00
M1	tden	-4739.70	2	9483.40	2885.0	10	0.00
2010 Annual Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-4238.44**	**9**	**8494.88**	**0.0**	**1**	**0.58**
M8	cut + lin + lc + elev + slope + tpi	-4240.94	7	8495.87	1.0	2	0.35
M7	tden + lc + elev + slope + tpi	-4243.69	6	8499.38	4.5	3	0.06
M4	lc + elev + slope + tpi	-4252.52	5	8515.03	20.2	4	0.00
M6	cut + lin + elev + slope + tpi	-4553.50	6	9118.99	624.1	5	0.00
M5	tden + elev + slope + tpi	-4562.41	5	9134.82	639.9	6	0.00
M3	elev + slope + tpi	-4592.88	4	9193.76	698.9	7	0.00
M9	cut + lin + well + seismic	-4941.20	5	9892.41	1397.5	8	0.00
M2	cut + lin	-4943.87	3	9893.73	1398.9	9	0.00
M1	tden	-4957.29	2	9918.58	1423.7	10	0.00
Global-Static Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-58596.19**	**9**	**117210.37**	**0.0**	**1**	**1.00**
M8	cut + lin + lc + elev + slope + tpi	-58663.38	7	117340.76	130.4	2	0.00
M7	tden + lc + elev + slope + tpi	-58718.39	6	117448.77	238.4	3	0.00
M4	lc + elev + slope + tpi	-58850.69	5	117711.38	501.0	4	0.00
M6	cut + lin + elev + slope + tpi	-60048.16	6	120108.32	2897.9	5	0.00
M5	tden + elev + slope + tpi	-60301.97	5	120613.95	3403.6	6	0.00
M3	elev + slope + tpi	-60694.29	4	121396.57	4186.2	7	0.00
M9	cut + lin + well + seismic	-71921.11	5	143852.22	26641.9	8	0.00
M2	cut + lin	-71972.81	3	143951.61	26741.2	9	0.00
M1	tden	-72463.94	2	144931.88	27721.5	10	0.00
Global-Dynamic Model							
M10	**cut + lin +well + seismic + lc + elev + slope + tpi**	**-53019.69**	**9**	**106057.37**	**0.0**	**1**	**1.00**
M8	cut + lin + lc + elev + slope + tpi	-53063.07	7	106140.14	82.8	2	0.00
M7	tden + lc + elev + slope + tpi	-53217.22	6	106446.44	389.1	3	0.00
M4	lc + elev + slope + tpi	-53418.29	5	106846.59	789.2	4	0.00
M6	cut + lin + elev + slope + tpi	-54676.00	6	109364.01	3306.6	5	0.00
M5	tden + elev + slope + tpi	-54969.25	5	109948.50	3891.1	6	0.00
M3	elev + slope + tpi	-55429.64	4	110867.29	4809.9	7	0.00
M9	cut + lin + well + seismic	-65453.74	5	130917.47	24860.1	8	0.00
M2	cut + lin	-65475.11	3	130956.21	24898.8	9	0.00
M1	tden	-65848.40	2	131700.80	25643.4	10	0.00

Figure A.1: Second-order relative probability of occurrence maps for the annual models for the RPC winter range in west-central Alberta (December 1 - April 30). The relative probability of selection is scaled between low (0, blue) and high (1, red).

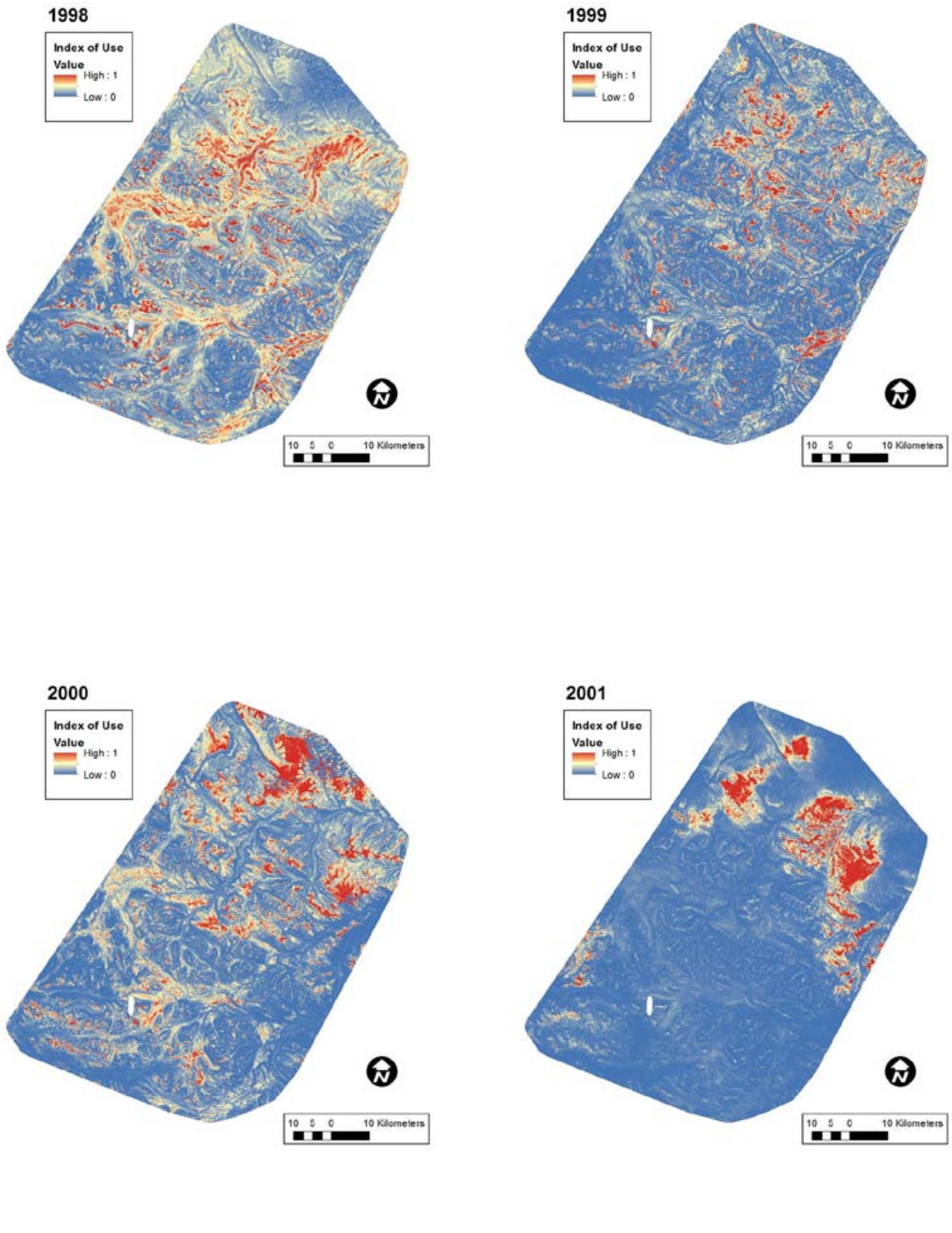


Figure A.1: Continued.

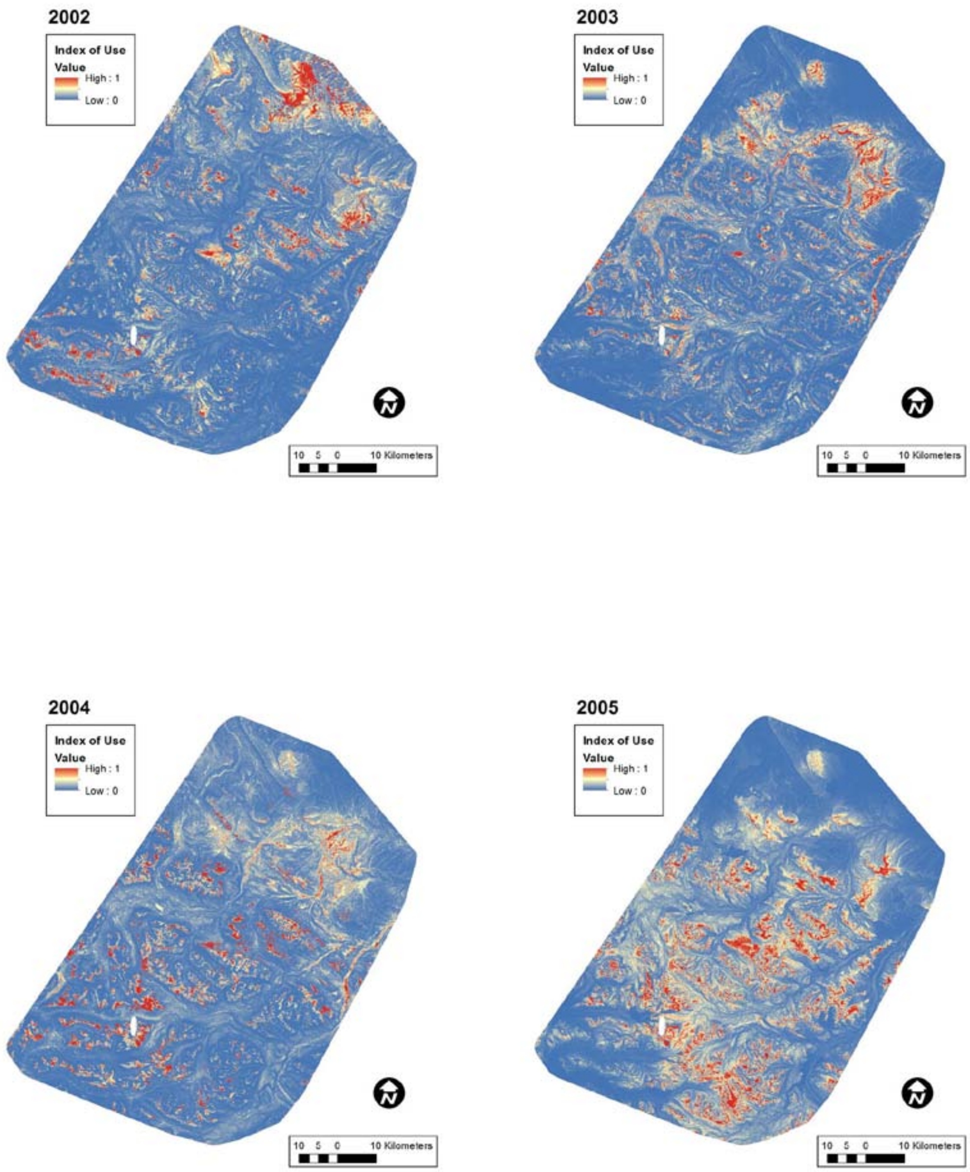


Figure A.1: Continued.

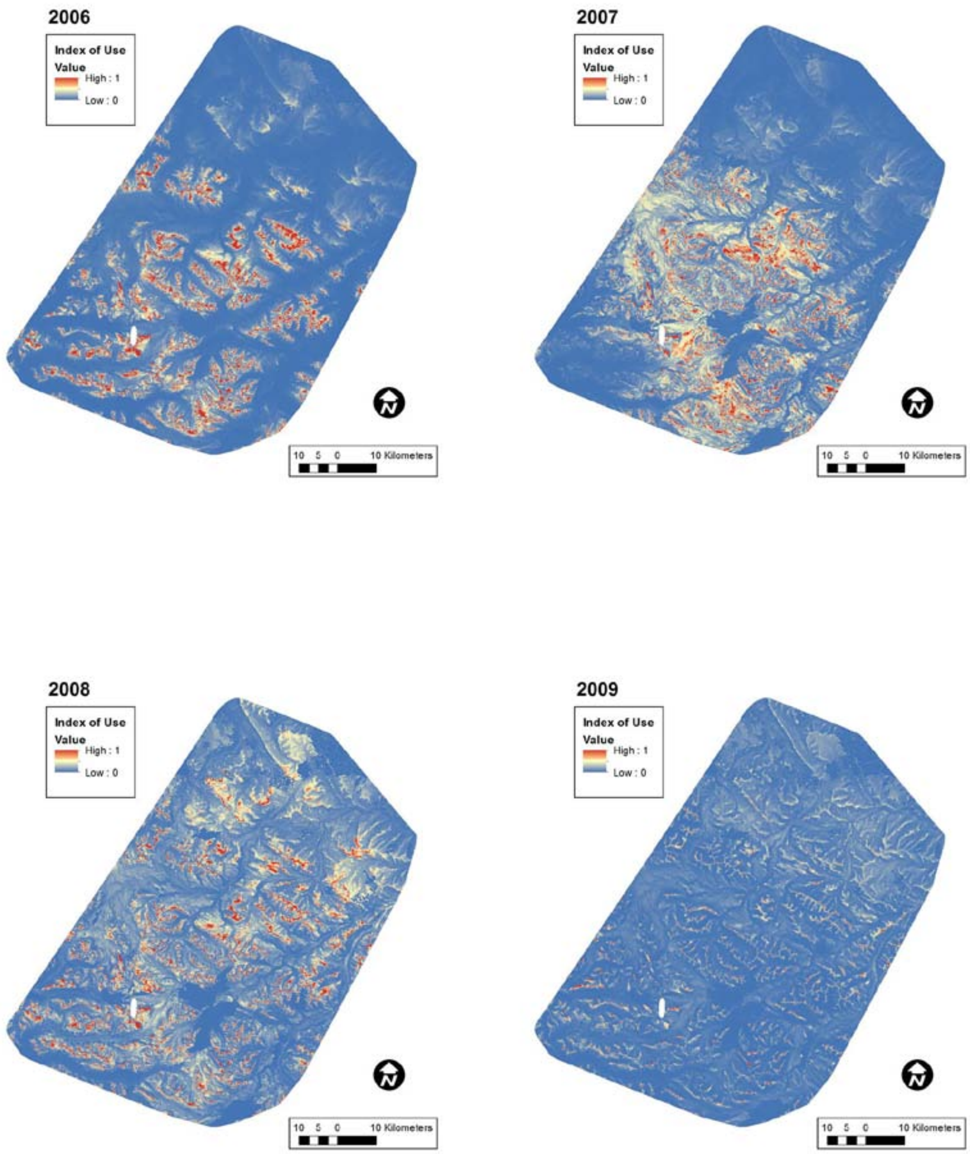
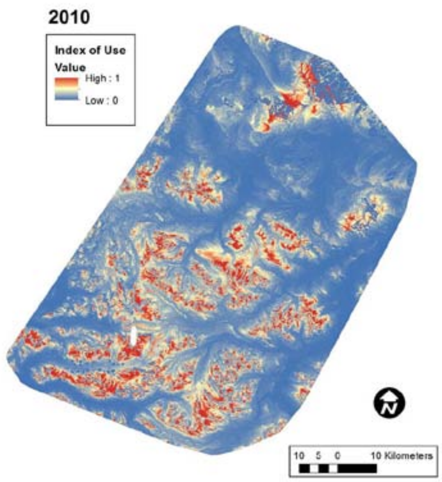
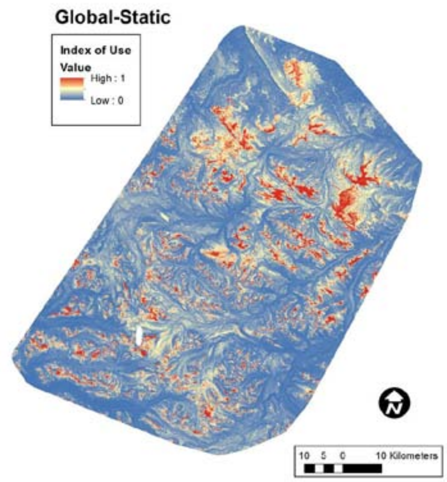


Figure A.1: Continued.





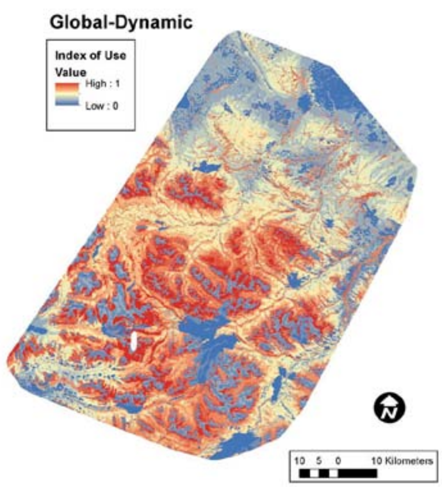


Figure A.2: Third-order relative probability of occurrence maps for the annual models for the RPC winter range in west-central Alberta (December 1 - April 30). The relative probability of selection is scaled between low (0, blue) and high (1, red).

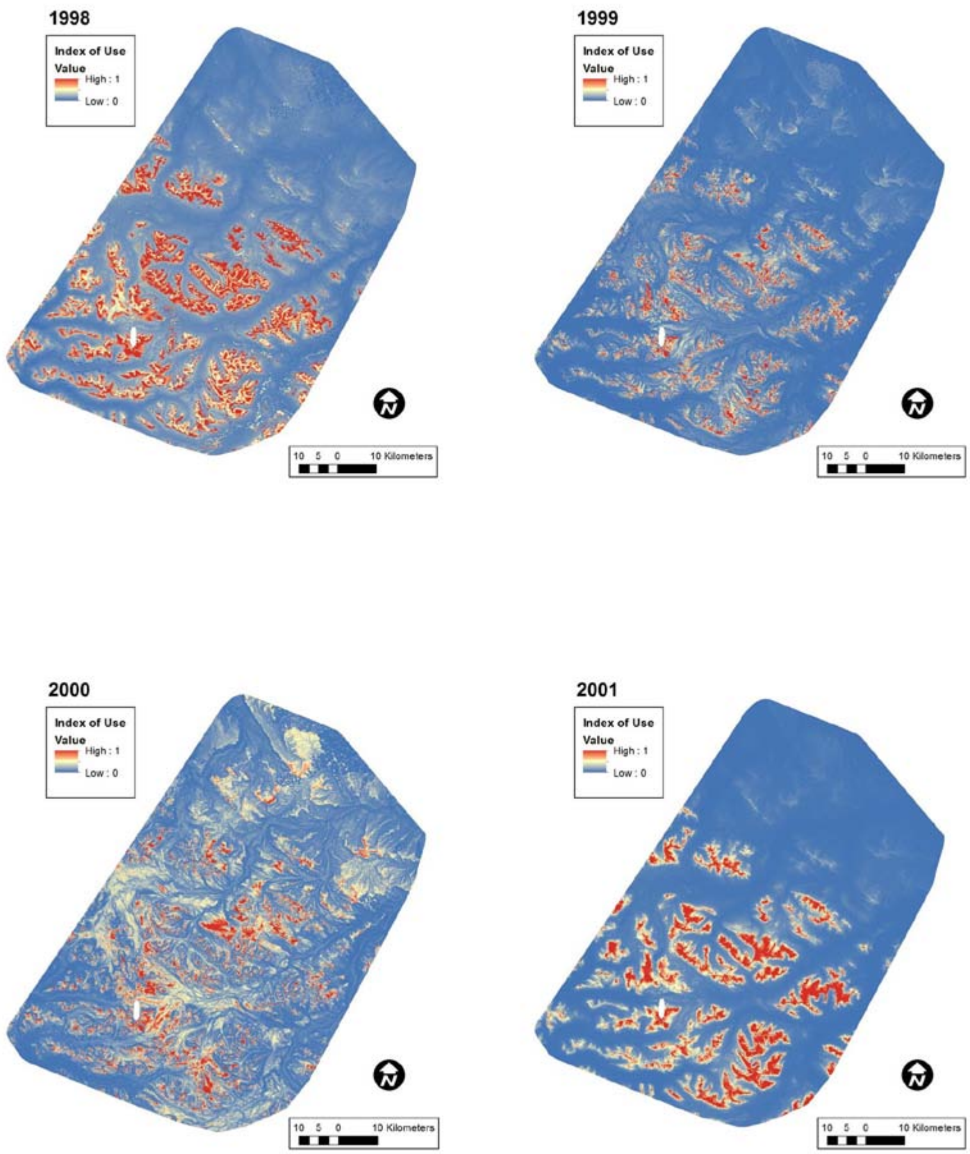
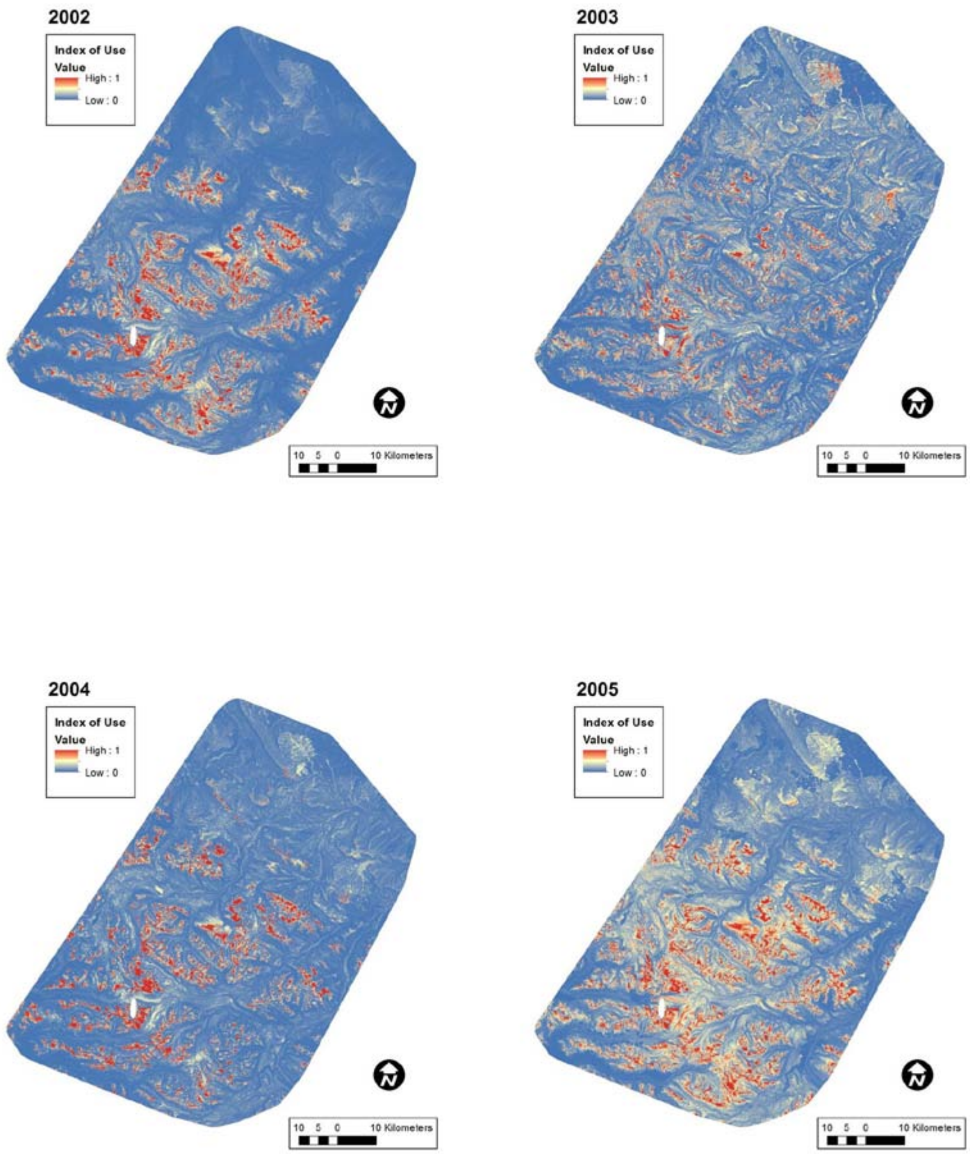


Figure A.2: Continued.



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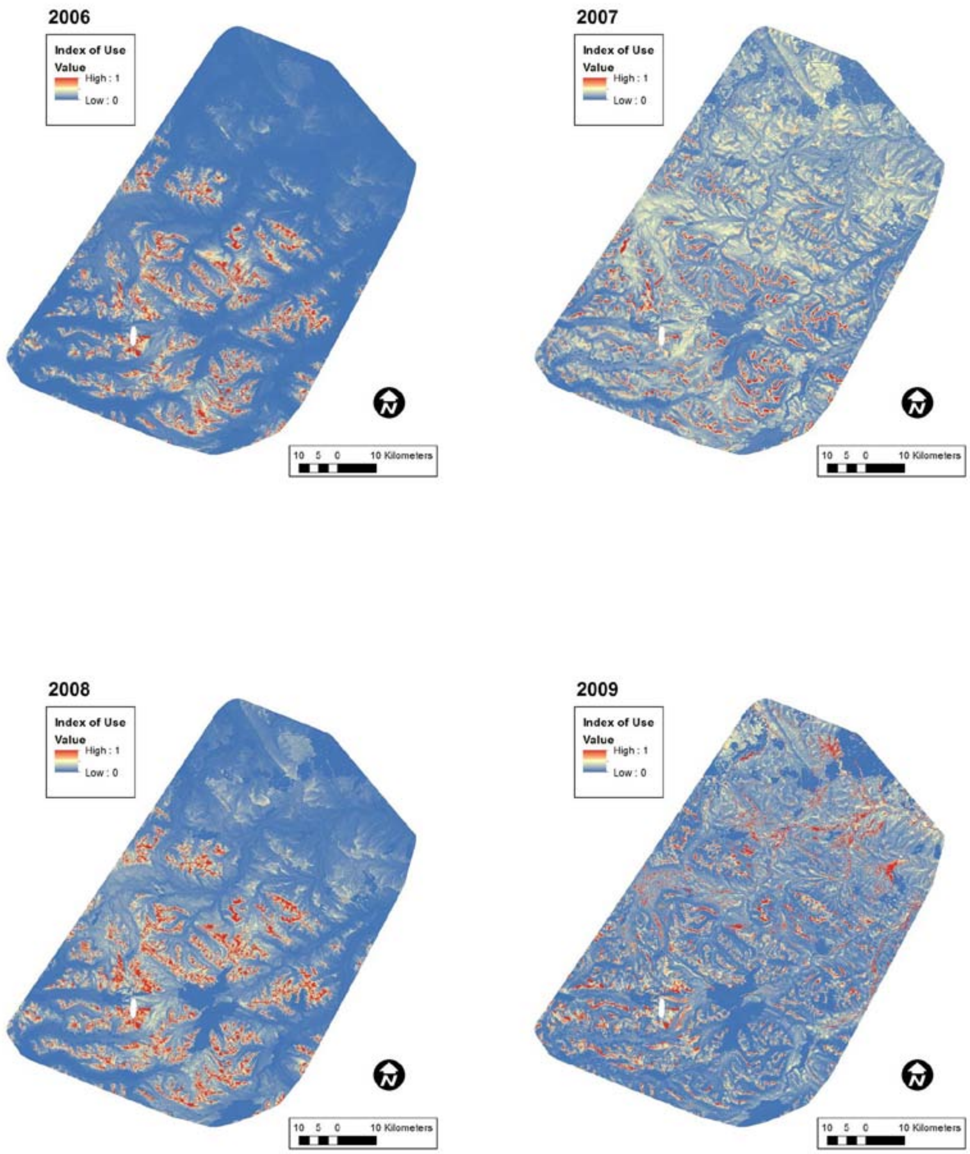

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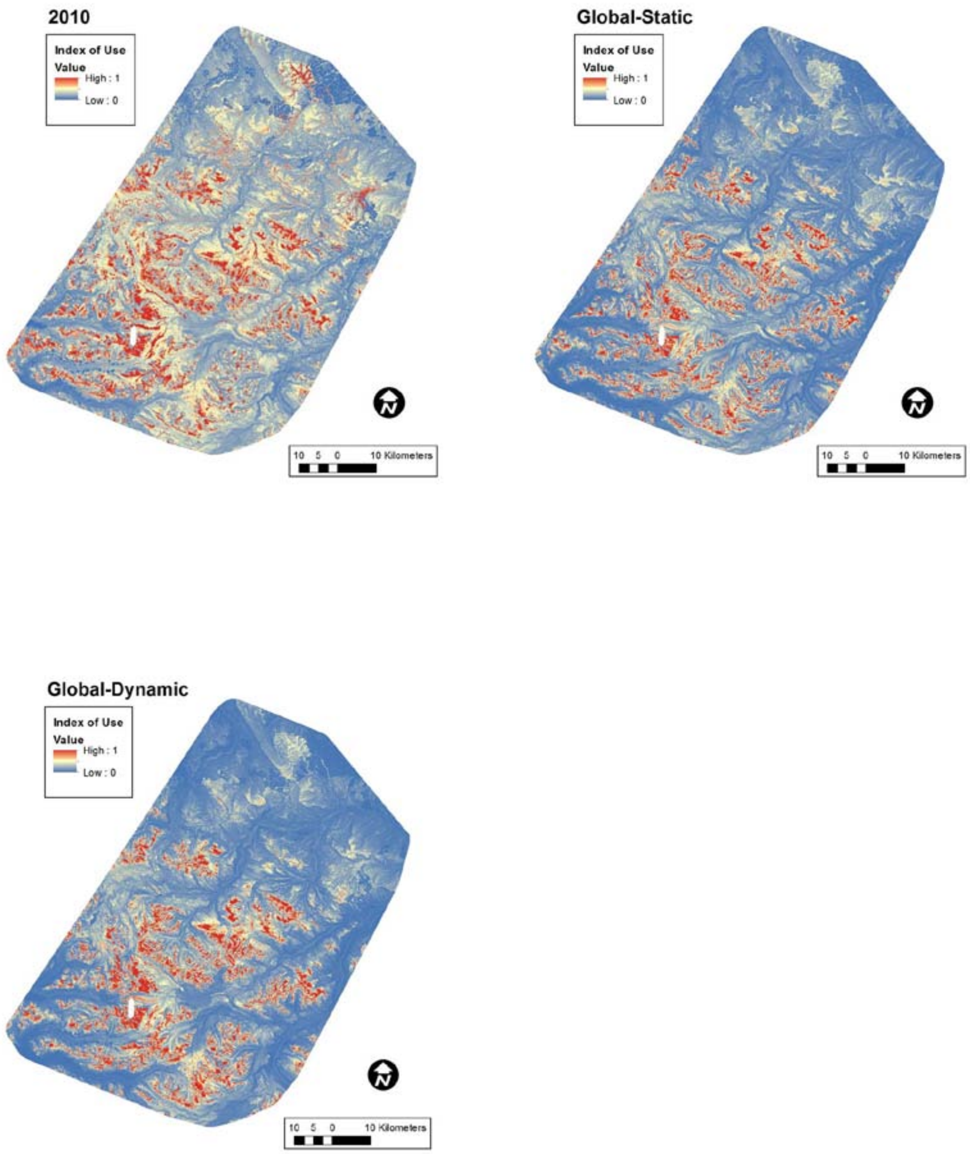
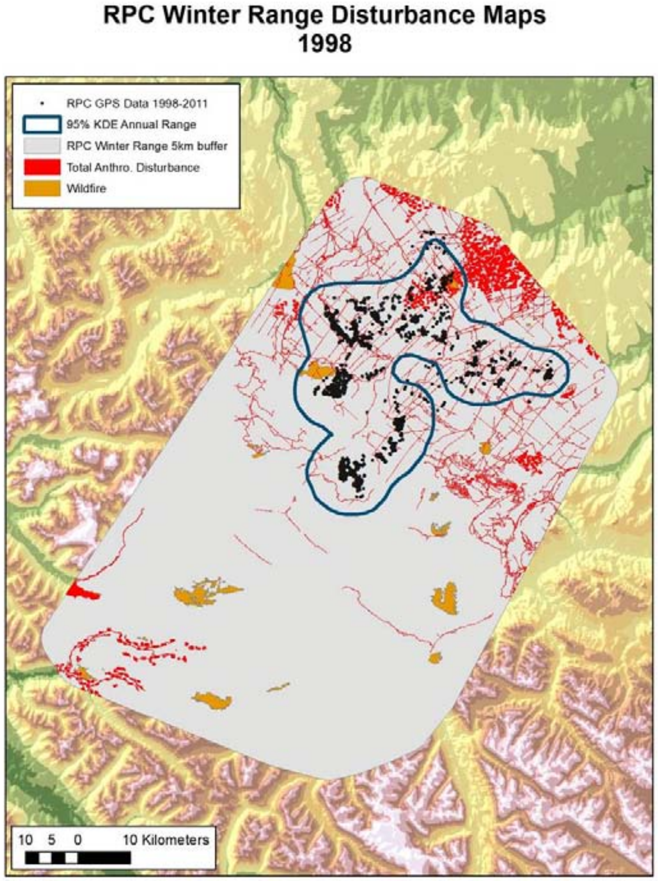


Figure A.3: Cumulative disturbance maps updated annually for the RPC population in west-central Alberta. The sequence of maps is from 1998 through to 2010 and a global map (all caribou locations and all disturbance). Annual ranges were calculated using 95% KDE.



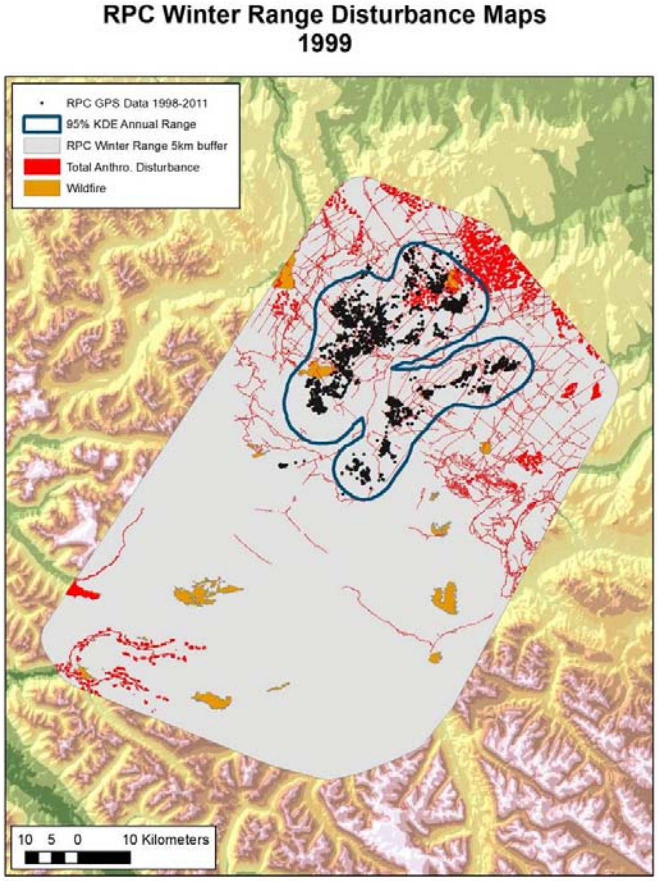
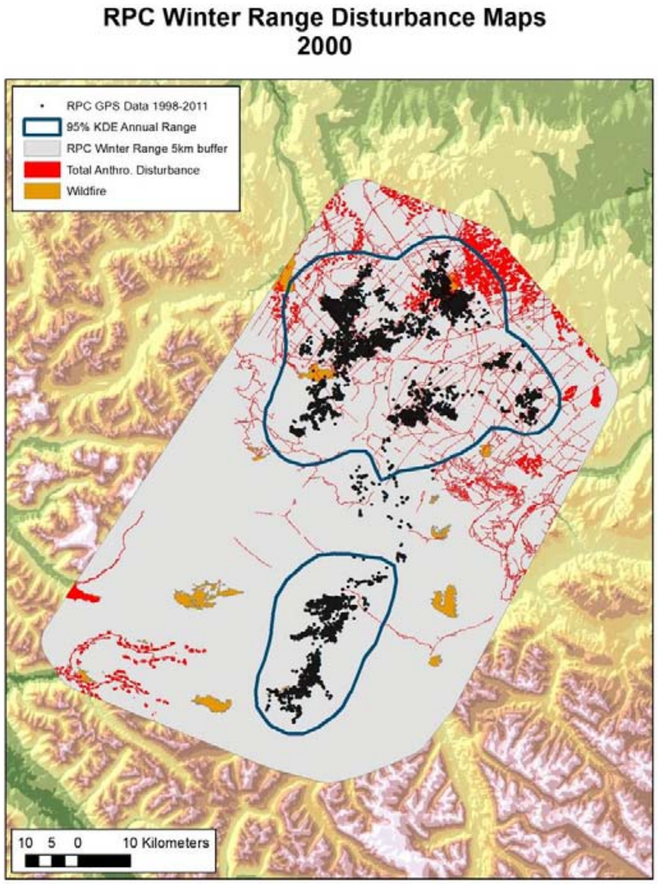
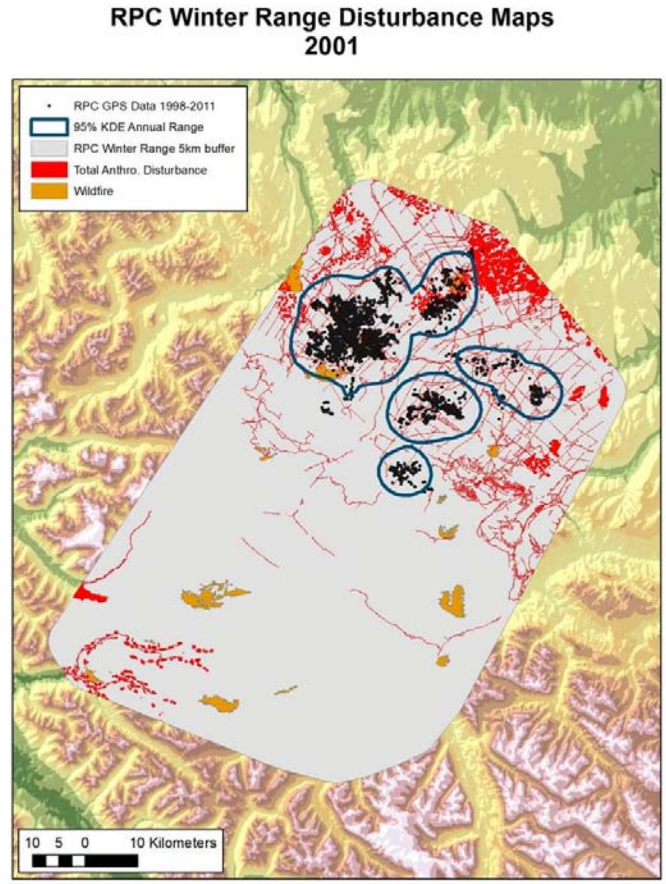


Figure A.3: Continued.





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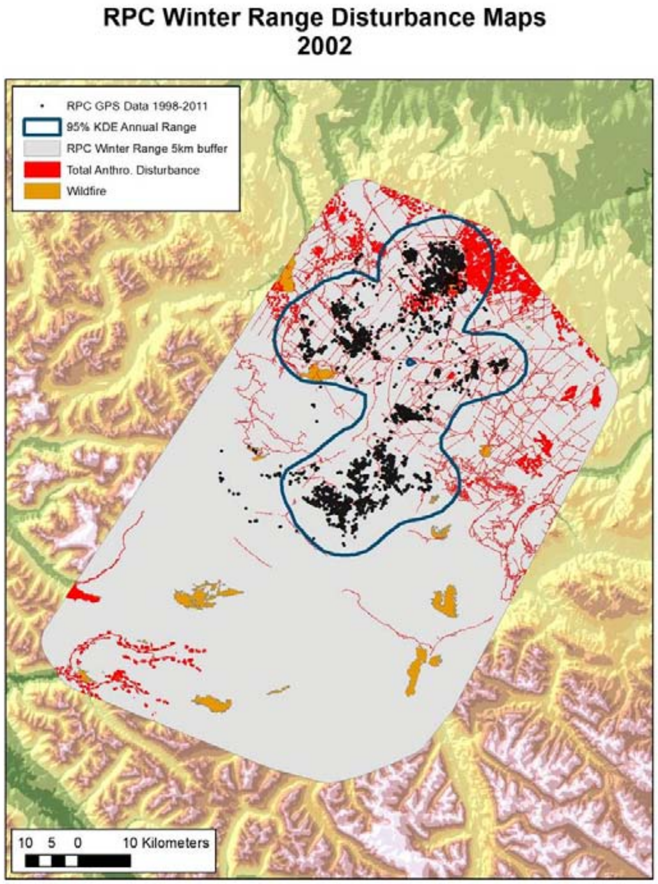

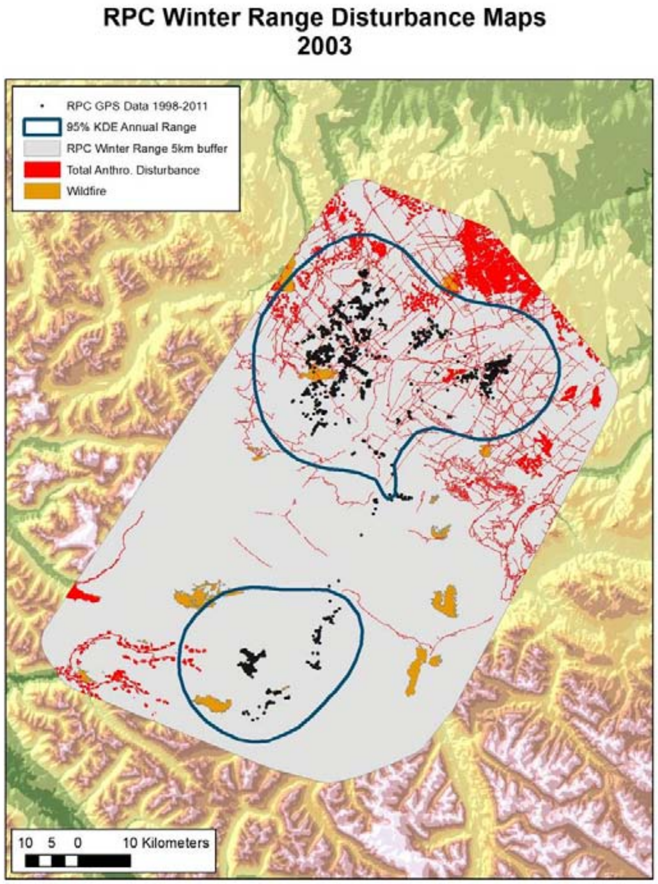

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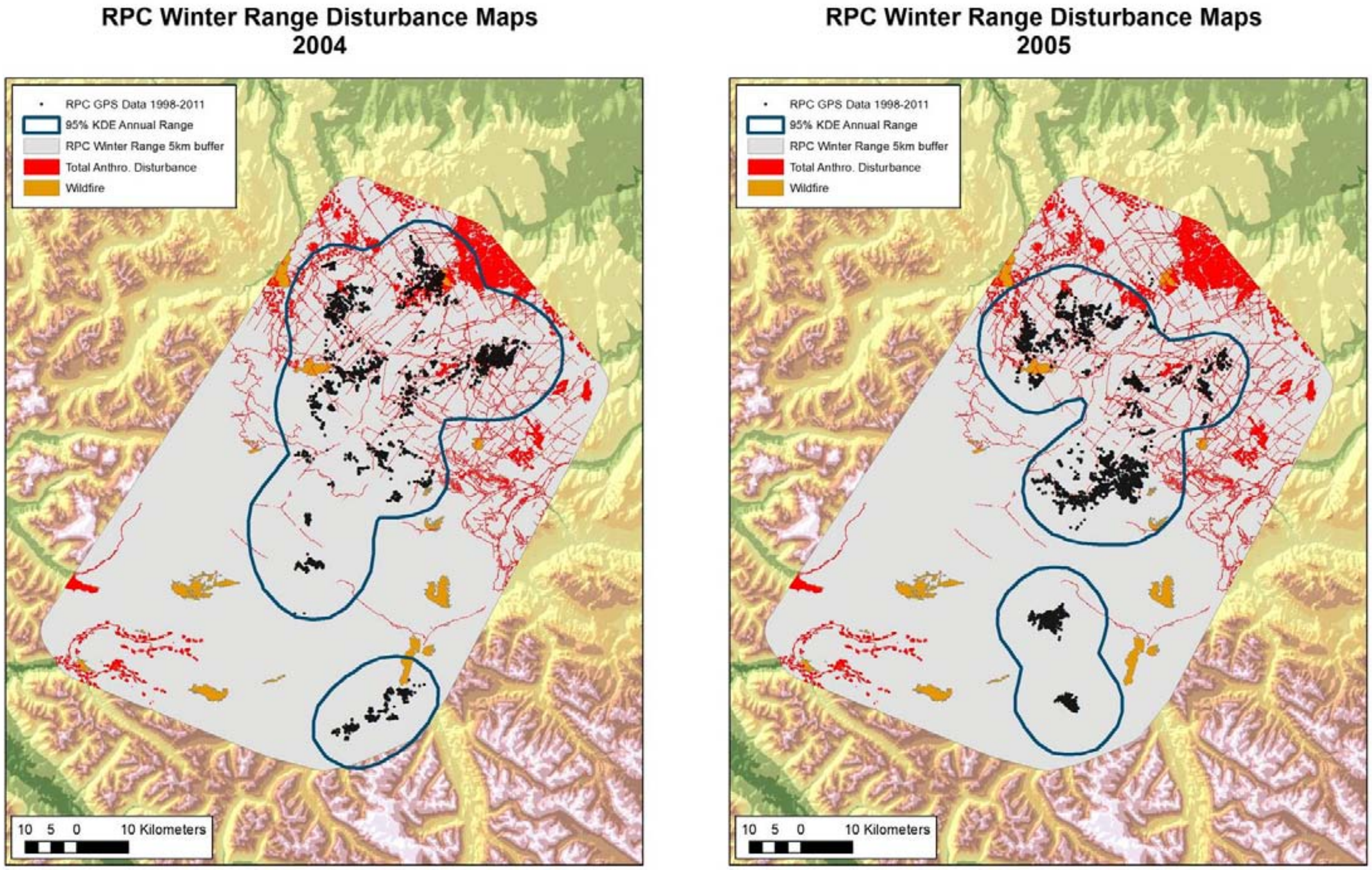
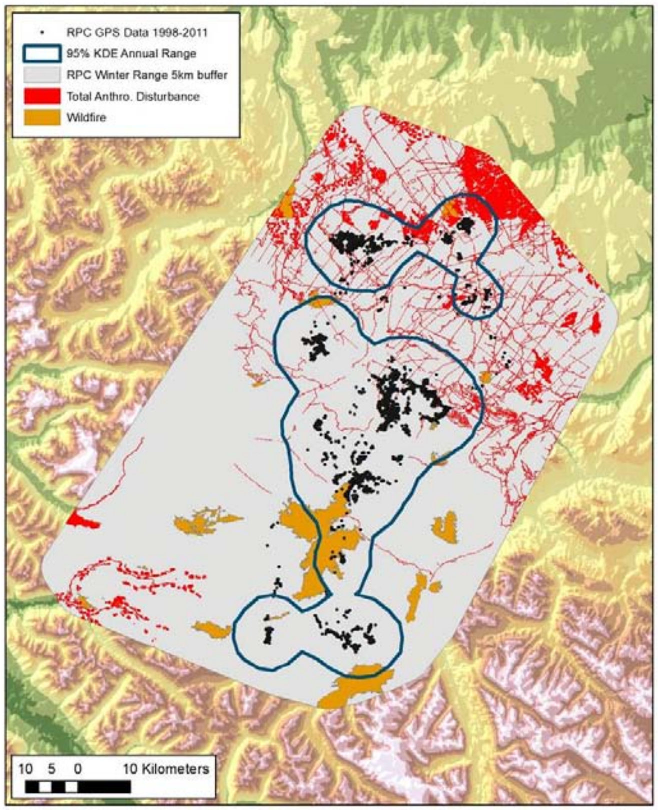


Figure A.3: Continued.

RPC Winter Range Disturbance Maps
2006



RPC Winter Range Disturbance Maps
2007

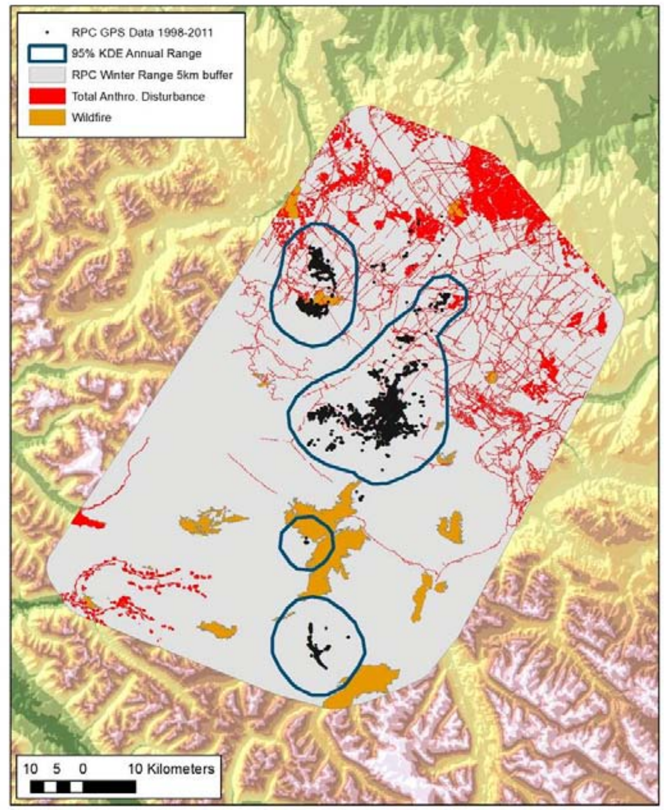
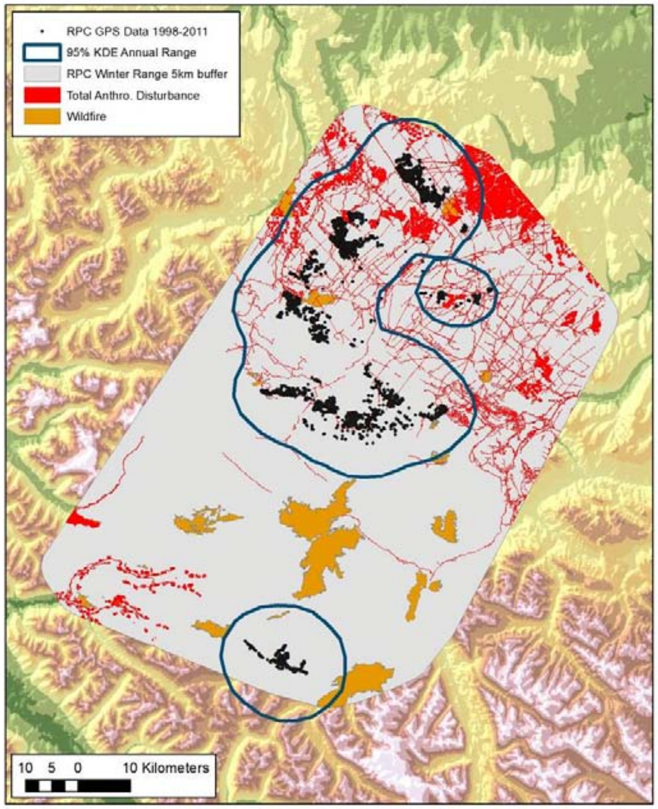


Figure A.3: Continued.

RPC Winter Range Disturbance Maps
2008



RPC Winter Range Disturbance Maps
2009

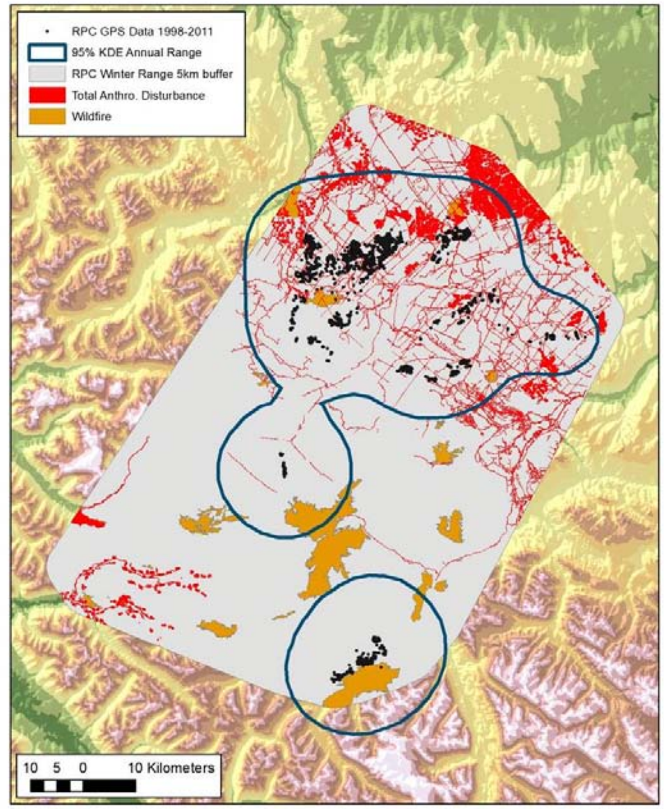


Figure A.3: Continued.

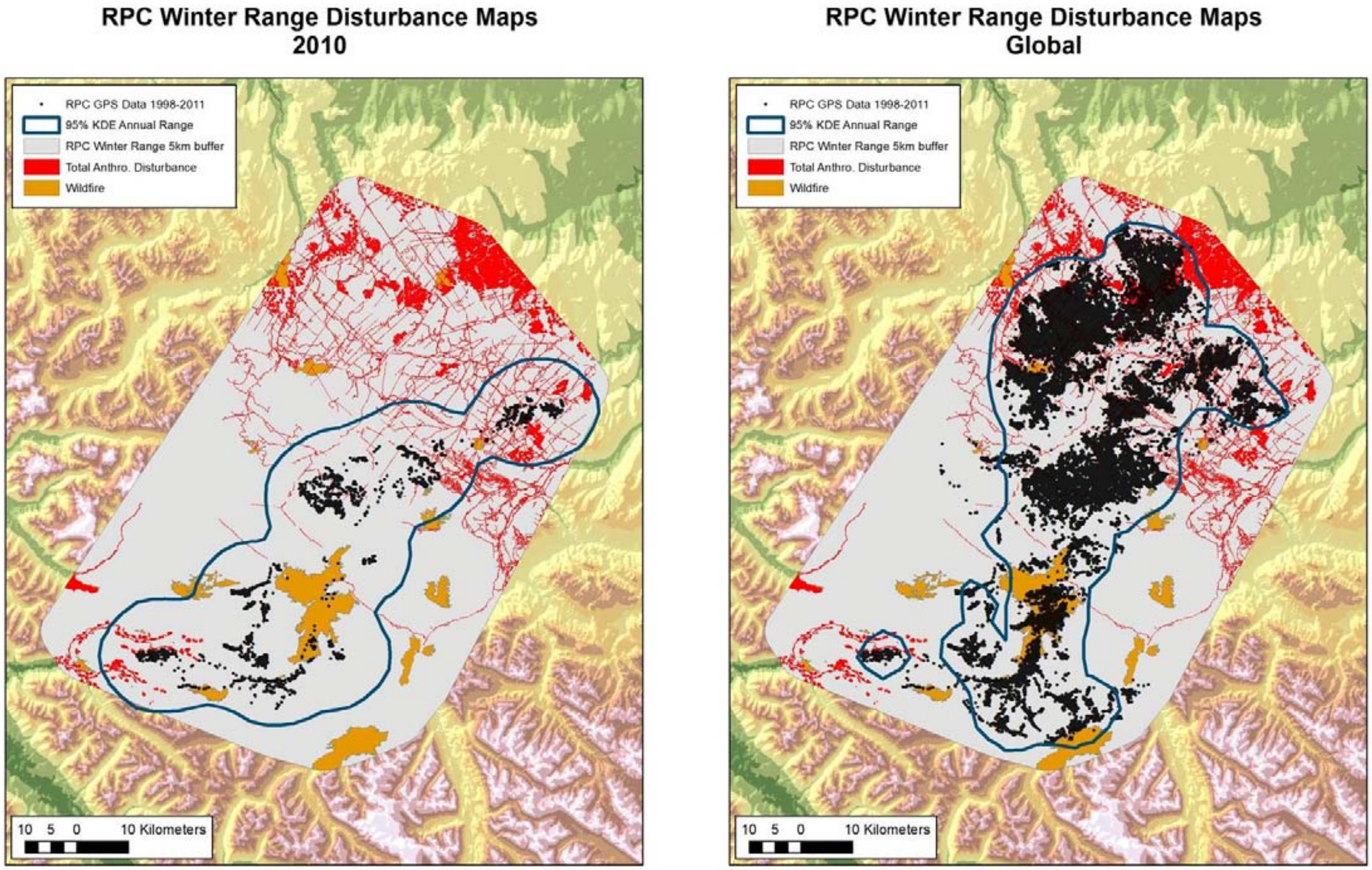


Figure A.4: Historical RPC winter range as defined by Edmonds and Bloomfield (1984: Figure 3, pg 37) used to guide the delineation of the RPC core winter range to examine spatial use of the RPC winter range from 1998 to 2011. The hatched areas represent primary the winter range from animals collared in 1980 to 1983.

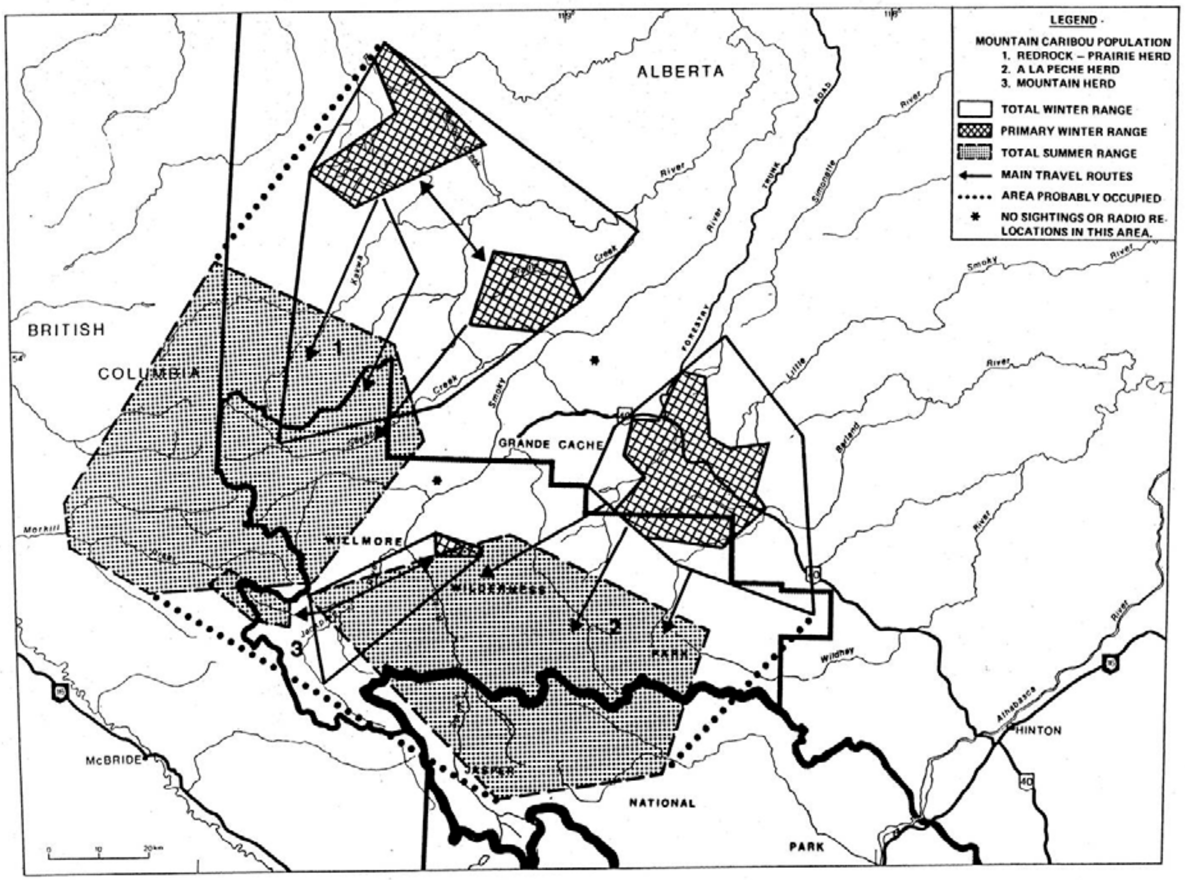


Figure 3. Mountain caribou range in the study area.

Figure A.5: Historical RPC winter range as defined by Brown and Hobson (1998: Figure 4, pg 25) used to guide the delineation of the RPC core winter range to examine spatial use of the RPC winter range from 1998 to 2011. The 100% MCP area represents winter (November - April) RPC caribou locations from 1981 to 1996.

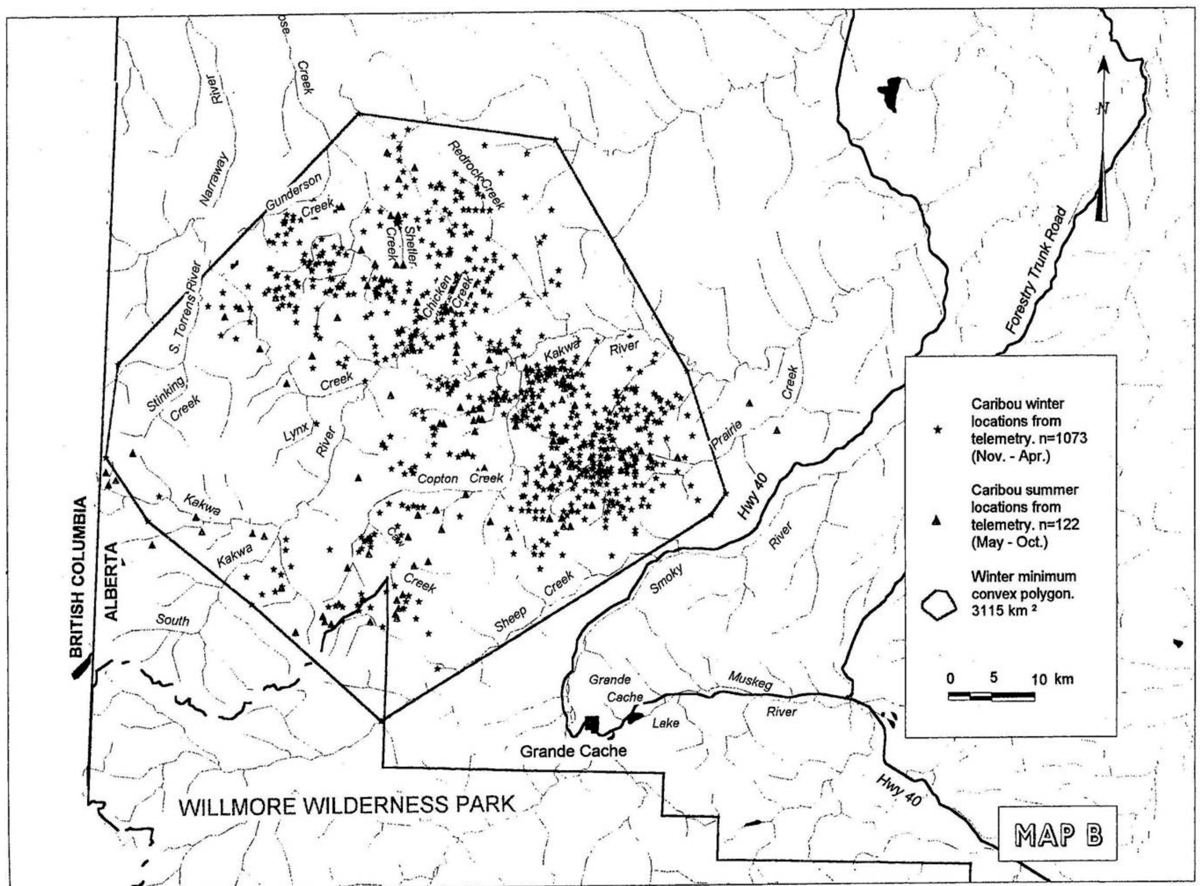


Fig. 4. Relocations of radio-collared caribou on the Redrock/Prairie Creek winter range, November 1981 to May 1996. Summer relocations (May to October) not made on the winter range are excluded.

Figure A.6: Estimated percent change in adult female population size for the RPC caribou population from 1997 to 2009. (Source: ASRD and ACA 2010; Figure 18(b), pg 45).