



6.4 Environmental Niche Analysis - Predicting potential occurrence of threatened boreal woodland caribou to support species recovery in Canada.

Introduction

The Boreal Caribou Critical Habitat Science Review has pursued four complementary analytical approaches to reflect the multi-scale, hierarchical interaction of species and their habitats; here we conducted an environmental niche analysis. We modeled the geographic extent of the environmental niche (fundamental and realized, e.g., abiotic and biotic) for boreal woodland caribou across its current extent of occurrence in Canada. While not directly incorporated into the prior analysis, the results presented here were expected to confirm the current national distribution, and contribute to the local population management in the action planning stage. For example, refined and validated niche models could inform management on priority areas for habitat restoration where the current local extent is not large enough, as well as guide monitoring programs throughout the extent of occurrence as part of the adaptive management framework. The population and distribution objectives in the National Recovery Strategy (Environment Canada 2007) are to maintain existing local populations of boreal caribou that are self-sustaining and achieve growth of populations not currently self-sustaining, throughout the current extent of occurrence. Delineation and management of these local populations are key to the recovery of boreal caribou (Environment Canada 2007).

The geographic distribution of a species is a function of its ecology and evolutionary history, determined by diverse factors operating at different spatial scales, including climate (Case and Taper 2000, Soberon 2007). We assume that a species will be present at a given point where three conditions are met: a) abiotic conditions (such as climate) are favourable, b) biotic conditions (other species) allow species to maintain populations, and c) the region is accessible to dispersal from adjacent populations (Soberon and Peterson 2005, Soberon 2007). These three conditions describe a species niche, one of the fundamental theories in ecology of how organisms use their habitats. Niche theory suggests that fitness or habitat suitability is not monotonically related to conditions or resources, but instead decreases from either side of an optimal condition (Hirzel et al. 2002). A geographic area, with the appropriate set of abiotic factors, free of competition from biotic factors for a species in which the species may theoretically occur, may be regarded as the geographical expression of the fundamental niche (Hutchinson 1957). In contrast, an area where the abiotic conditions are favourable but we also consider biotic interactions, such as competition and predation, may be considered the geographical representation of the realized niche (Hutchinson 1957). A region that has the right set of biotic and abiotic factors and is accessible to the species (via dispersal) is the potential geographic distribution of the species (MacArthur 1967, Soberon 2007).

The recent availability of species occurrence data over large regions, for example from breeding bird surveys or large-scale wildlife surveys, combined with the availability of large-



scale environmental climate and biotic data, has led to an increase in approaches to model the distribution of species (Soberon 2007). Species distribution models are one type of empirical model relating spatial observations of an organism to environmental predictor variables, using a variety of statistical techniques, from logistic regression to more complex computation approaches (Guisan and Zimmerman 2000). Guisan and Thuiller (2005) suggested that environmental predictors for species distribution models should be chosen to capture the three main types of influences on species distribution: i) limiting factors or regulators, defined as factors controlling a species ecophysiology (e.g., temperature, water, soil), ii) disturbances (natural or human), and iii) resource availability, defined as all compounds that can be assimilated by organisms (e.g., energy and water). Spatial patterns in relationships between species and their environments vary with scale, often in a hierarchical manner (Johnson 1980, Pearson et al. 2004). Environmental niche models are conceptually similar to other species distribution models commonly employed in ecology (resource selection functions (Boyce and McDonald 1999), bioclimatic envelopes (Hijmans and Graham 2006) etc.), but niche models are explicitly linked to niche theory and usually address distribution across broad regional scales (Anderson et al. 2002). Environmental niche models reconstruct species' ecological requirements (conditions or resources) and predict the geographical distribution of those requirements.

Ecological niche models (ENM) have been used to study issues in evolution (Peterson 2001), ecology (Anderson et al. 2002), and conservation (Peterson and Robins 2003). Their predictive models of species geographic distributions are important in a variety of conservation applications, such as conservation reserve design (Wilson et al. 2005), to predict the spread of invasive species (Peterson 2003), and to predict the effects of climatic change on species responses to future and past climates (Pearson and Dawson 2003, Hijmans and Graham 2006, Peterson et al. 2004). ENM models have been used to assess the distributional patterns of endangered species in many countries, including the United States (Godown and Peterson 2000), China (Chen and Peterson 2000), and eastern Mexico (Peterson et al. 2002). ENM models have also been used to incorporate multiple species and trophic interactions for example the implications for endangered spotted owls (*Strix occidentalis*) by invading barred owls (*S. varia*) facilitated by human disturbance in Washington and Oregon (Peterson and Robins 2003). Guisan et al. (2006) suggested that niche-based models may improve the sampling of rare species and Raxworthy et al. (2003) used ENM to target field surveys for under-studied reptiles and located previously undiscovered chameleon species in Madagascar.

As part of the science review process for boreal caribou, our goal was to support the identification of critical habitat by employing environmental niche analysis to understand the pattern of occupancy in the current extent of occurrence. First, we examined the potential distribution (fundamental niche) as a function of climate and topography for two 30-year time periods: 1930 to 1960 and 1971 to 2000. Boreal caribou have experienced a range contraction at the southern limit of their distribution; therefore we hypothesized that the potential distribution of woodland caribou has shifted northwards between these two periods. This analysis may help determine contributions of climate change in limiting habitat use by caribou. Second,



we predicted that the pattern of occupancy (realized niche) within the current extent of occurrence by biotic predictor variables, as derived from satellite imagery and other spatially explicit sources (e.g., Petrorrelli et al. 2005). Woodland caribou declines are hypothesized to result from indirect effects of anthropogenic disturbance across their range (McLoughlin et al. 2003, Laliberte and Ripple 2004, Vors et al. 2007, Wittmer 2007). Proximally, human disturbance is thought to increase primary prey densities, and hence densities of predators, like wolves and black bears, that cause caribou populations to decline through apparent competition (Seip 1992, James and Stuart-Smith 2000). Therefore, we hypothesized that the realized niche would be constrained by these biotic interactions indexed through spatial measures of human disturbance across the boreal forest. Finally, we discussed the potential contributions of environmental niche modelling to various aspects of the ongoing, adaptive process of identifying critical habitat as identified under the federal recovery plan for the species under the auspices of the Canadian Species at Risk Act.

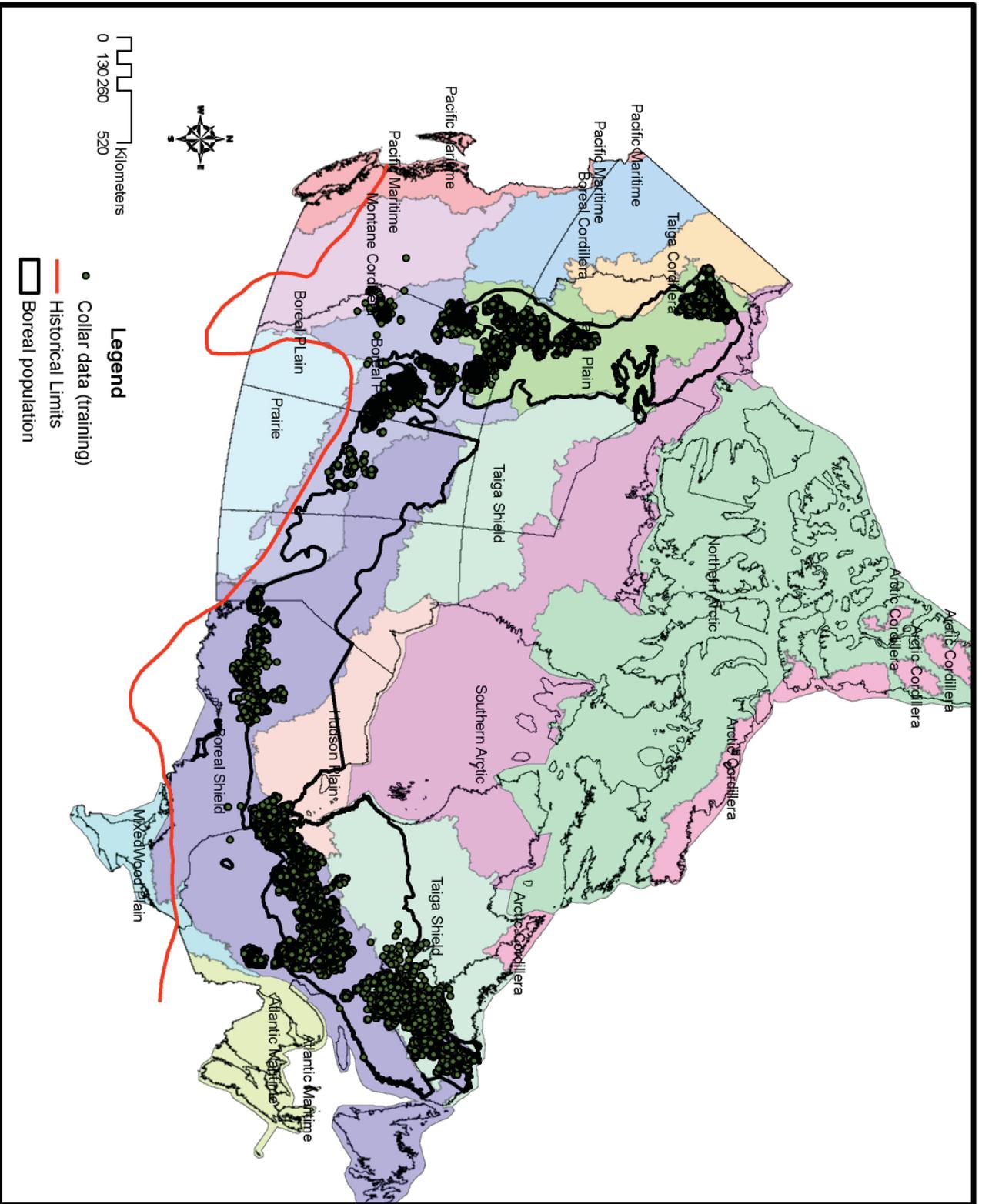
Methods

The boreal caribou is a forest-dwelling sedentary ecotype of woodland caribou with an extent of occurrence over approximately 2.4 million km², in eight provinces and territories, and occurring predominantly within five ecozones (Environment Canada 2007: Figure 1, Table 1). Often, boreal woodland caribou habitat is characterized as peatland complexes intermixed with mature to old pine, black spruce, and tamarack (e.g., Brown 2005). Forested peat complexes with abundant arboreal lichens and uplands dominated by mature conifers with dense ground lichens are typical of boreal caribou habitat, and are thought to provide for nutrient rich forage and as a refuge from higher predator densities associated with typical deer and moose habitat (e.g., deciduous/mixedwood, Thomas et al. 1996, McLoughlin 2003, 2005).

Occurrence Data

Geo-referenced boreal caribou observational location data were obtained from a variety of sources and consisted of various acquisition methods including: GPS (Global Positioning System) collar, ARGOS collar, VHF collar, aerial surveys, ground surveys, and incidental observations, ranging over time from the 1940's to 2007. The database included over 1 million records of caribou observations. Two different datasets were used for niche modelling, to train and to validate the models, respectively. For the former, collared (GPS, ARGOS, VHF) data were used, whereas non-collared (surveys) data were held back for independent validation of outputs (Fielding and Bell 1997, Boyce et al. 2002).

To reduce spatial and temporal autocorrelation and minimize bias introduced by collar type, training data were limited to one location, per animal, per day by random selection where multiple daily acquisitions were captured (White and Garrott 1990). Because occurrence data frequently included location error, entries with uncertainty greater than 1 km were excluded for the study, regardless of acquisition method. The training dataset for current analyses consisted of over 217,000 points from collared animals, but the distribution of locations was



Appendix 6.4 - Figure 1. Ecozone map of Canada showing the current extent of occurrence of Boreal Caribou and the training (collar) data. The black line represents the current extent of occurrence for the boreal ecotype from the National Recovery Strategy and the red line is the historical southern extent of woodland caribou (Environment Canada 2007).



not uniform throughout the geographic range of boreal caribou (Table 1, Figure 1). Therefore, we stratified sampling to obtain datasets representative of the species-habitat variability in different ecozones across the extent of occurrence (Callaghan 2008). For modelling purposes, we produced ten subsets consisting of 10,000 points randomly selected from the 200,000 locations at the same ratio as the proportion of boreal caribou range represented by that ecozone (Boyce et al. 2002, Araujo and New 2006). Niche models are sensitive to sample size and to biases in the geographic distribution of the data (Peterson and Cohoon 1999, Stockwell and Peterson 2002). Statistical sampling designs outlined above have been suggested to limit these biases, while increasing model performance (Araujo and Guisan 2006). Although this balanced the coverage for ecozones, there were still considerable gaps in the geographic distribution of the occurrence data (Figure 1).

Appendix 6.4 - Table 1. Percent of Boreal caribou extent of occurrence in each ecozone and breakdown of collar locations used for training subsets input data.

Ecozone	Percent of Extent of Occurrence	Percent of fixes	Number of fixes
Boreal Plain	13.5	21.4	46561
Boreal Shield	41.1	43.5	94893
Hudson Plain	7.7	1.7	3809
Montane Cordillera	0.4	0.6	1207
Southern Arctic	2.2	0.1	134
Taiga Cordillera	0.1	0.0	29
Taiga Plain	19.6	26.7	58115
Taiga Shield	15.3	6.1	13227

Environmental Covariates

To predict the geographic extent of the boreal caribou environmental niche, we used abiotic and biotic variables including climate surfaces, topography, and biotic variables derived from satellite and existing vector data. Climate covariates were created using an interpolation technique based on thin-plate-smoothing splines (Hutchinson 1995). Biologically meaningful climate parameters (35 bioclimatic) were derived from monthly temperature and precipitation data that were averaged over two 30-year time periods: 1930 to 1960 and 1971 to 2000. Data were provided by the Canadian Forest Service at 30 arcseconds (~1 km) and 300 arcseconds (~10 km) resolutions (see McKenney et al. 2006). Potential variables were selected based on hypotheses developed from literature reviews of caribou and other northern ungulates (Table 2). Climatic variables have been shown to affect population dynamics in many large-bodied, northern ungulates through direct and indirect mechanisms at a variety of scales (Weladji et al. 2002). Indirect effects include for example, late winter precipitation and spring temperatures and precipitation on forage quality and its quantity in summer, and conditions of the summer range have shown associated effects on body size and reproductive success (Finstad et al. 2000). However, winter weather severity also has direct effects on population dynamics. Years with high snowfall may lead to increased winter calf mortality (Fancy and



Whitten 1991), decreased body mass of calves (Cederlund et al. 1991) and lighter yearlings (Adams and Dale 1998). To reduce collinear predictor variables, we randomly sampled 10,000 grid cells from the entire country and derived Pearson’s correlation coefficients for 35 bioclimatic parameters and elevation. We excluded variables that had a coefficient of correlation >0.7 (Parra et al. 2004).

Appendix 5.4 - Table 2. Climate variables included in the abiotic environmental niche models together with elevation (from McKenney et al. 2006).

Variable	Hypothesis
Precipitation in driest period	High summer/fall forage availability – improved condition at breeding
Total precipitation for 3 months prior to start of growing season	Early green-up – improved calf survival
Growing degree days (gdd) above base temperature for 1st 6 weeks of growing season	Early green-up – improved calf survival
Precipitation of coldest quarter	Food limitation caused by crusting or snow depth
Gdd above base temperature 3 months prior to growing season	Snowy late winters lead to improved summer forage
Annual mean temperature	Range limit based on physiology
Maximum temperature of warmest period	Range limit based on physiology
Annual temperature range	Range limit based on physiology

Digital elevation models (DEM) were derived from the Shuttle Radar Topography Mission (SRTM) data and obtained from the WorldClim website (www.worldclim.org) at 1-km and 10-km grid cell resolution.

To model realized niche, we attempted to capture attributes related to competition (e.g., resource availability and predation) that may restrict the occupied niche or environmental space. To account for forage resources we included: MODIS derived cumulative annual fraction of Photosynthetically Active Radiation (fPAR) (Coops et al. 2007, Huete et al. , Zhao et al. 2005), minimum annual fPAR (Coops et al. 2007), landcover (Latifovich, unpub), and peatland presence (Tarnocai 2005). The fPAR data were derived from a physically-based model that describes the propagation of light in plant canopies (Tian 2000) together with MODIS spectral bands. The cumulative annual fPAR reflects the annual productivity of the site, whereas the minimum annual fPAR represents the minimum perennial cover of the site (Yang et al. 2006, Coops et al. 2007). Few studies incorporate information to account for predators or competitors directly in niche modelling, and those that have modelled the environmental niche of the predator or competitor and included them as a covariate (Peterson and Robbins 2003, Heikkinen et al. 2008). Few density data exist for the main predators of caribou across the boreal forest, yet predation by wolves and black bears is the most frequently identified limiting factor of caribou populations (Bergerud and Elliot 1989,



Johnson et al. 2004). However, the principal driving factor changing predator distributions at the southern limit of caribou range is hypothesized to be anthropogenic disturbance. Modern commercial forestry creates new early seral forest stands which benefit primary prey species, such as moose (*Alces alces*) and deer (*Odocoileus spp*), followed by wolves (Fuller 1981) resulting in increased predation rates on secondary prey such as caribou (Wittmer 2007). Human activities also include linear developments like roads, seismic exploration lines, pipelines, and utility corridors, all of which increase predation rates and efficiency of wolves preying on caribou (James and Stuart-Smith 2000, McKenzie 2006). Therefore, we approximated predation risk with: road density (calculated as the total distance of roads within 1-km pixel from the Updated Road Network (Geobase), the Statistics Canada Road Network (Statistics Canada) and the DMTI SpatialTM roads database GFWC 2007), disturbance (from GFWC anthropogenic footprint, GFWC 2007), mean forest patch size, number of forest patches, standard deviation forest patch size (derived from Earth Observation for Sustainable Development, calculated using (EOSD) gridded at 1 km, (Wulder et al. 2008).

Environmental Niche Modelling

Ecological niches of boreal caribou were modelled using Maximum Entropy (MaxEnt; Phillips et al. 2004, 2006). Maxent estimates the most uniform distribution (maximum entropy) of occurrence points across the study area, given the constraints that the expected value of each environmental variable under this estimated distribution matches its empirical average (Phillips et al. 2004, 2006). The raw output is a probability value (0-1) assigned to each map cell of the study area, which are then converted to a percentage of the cells with the highest probability value. This is termed the 'cumulative' output. Comparative studies using MaxEnt for species distribution modelling that used independent validation performance suggest that it is more accurate than other models (Elith et al. 2006, Hernandez et al. 2006) and does not require or incorporate known absences in the theoretical framework (Phillips et al. 2004). In MaxEnt, it is unnecessary to define the occupancy threshold *a priori*. In fact, the spatially explicit continuous probability output may be one of the most relevant advantages of MaxEnt for Critical Habitat Identification because it allows for the fine distinction of habitat suitability in different areas (Kirk 2007). We examined the continuous cumulative output to determine the potential to distinguish a continuum of habitat suitability in different areas.

For intrinsic model evaluation, the area under the receiver-operating characteristic (ROC) curve (AUC) provides a single measure of model performance, independent of any particular choice of threshold (Fielding and Bell 1997). The ROC curve is obtained by plotting sensitivity (fraction of all positive instances that are classified as positive or true positive rate) on the y-axis and 1-specificity (fraction of all negative instances that are classified as negative) for all possible thresholds. Since MaxEnt does not require or use absence data (negative), the program considers the problem of distinguishing presence from random, rather than presence from absence. Our ROC analysis used all the test localities as instances of presence and a sample of 10,000 random pixels drawn from the background as random instances (Phillips et al. 2006). A random prediction corresponds to an AUC of 0.5, the best discriminating model corresponds to an AUC of 1.0.



Model Scenarios

We produced environmental niche models for boreal caribou based on three independent environmental datasets to satisfy the objectives outlined above:

- 1) Potential distribution based on climate averages from 1971 to 2000 and elevation (current fundamental niche).
- 2) Potential distribution based on climate averages from 1930 to 1960 and elevation (historic fundamental niche).
- 3) Realized distribution based on biotic variables from recent satellite imagery (current realized niche).

If observed range contractions by caribou resulted, at least in part, from climate change, then we expected that the current fundamental niche should differ most from the historical fundamental niche at more southerly reaches and/or regions where caribou were present historically. Further, if biotic interactions exacerbated by anthropogenic disturbance account for range contractions, then we expected that the current realized niche should be smaller than the current fundamental niche.

All scenarios used the dataset derived from collared animals for training the models. Ten random subsets were run individually with MaxEnt (v3.1) and the cumulative distribution pixel values were averaged over the ten runs to produce a final map.

Results

Climate and Topography

The final models included the variables listed in Table 2. Mean AUC scores among sub-samples was 0.95, indicating that the model output was significantly better than random. Figures 2 and 3 show the mean cumulative Maxent output for climate variables and topography niche models, from 1930 to 1960 and 1971 to 2000, respectively. Outputs showed that areas of highest probability in both maps correspond to areas where collar data were available to train the model (Figure 1). Similarly, areas with no training presence are not strongly predicted in either time period (e.g., areas in northern Manitoba). Visual inspection along the southern extent of the distribution suggested that the fundamental niche in Ontario and Quebec has not changed significantly over the two time periods. In Alberta, however, the potential distribution may have receded northward. The earlier distribution map suggested presence to the disjointed Little Smokey population, whereas the map from the later time period did not (Figures 2 and 3).

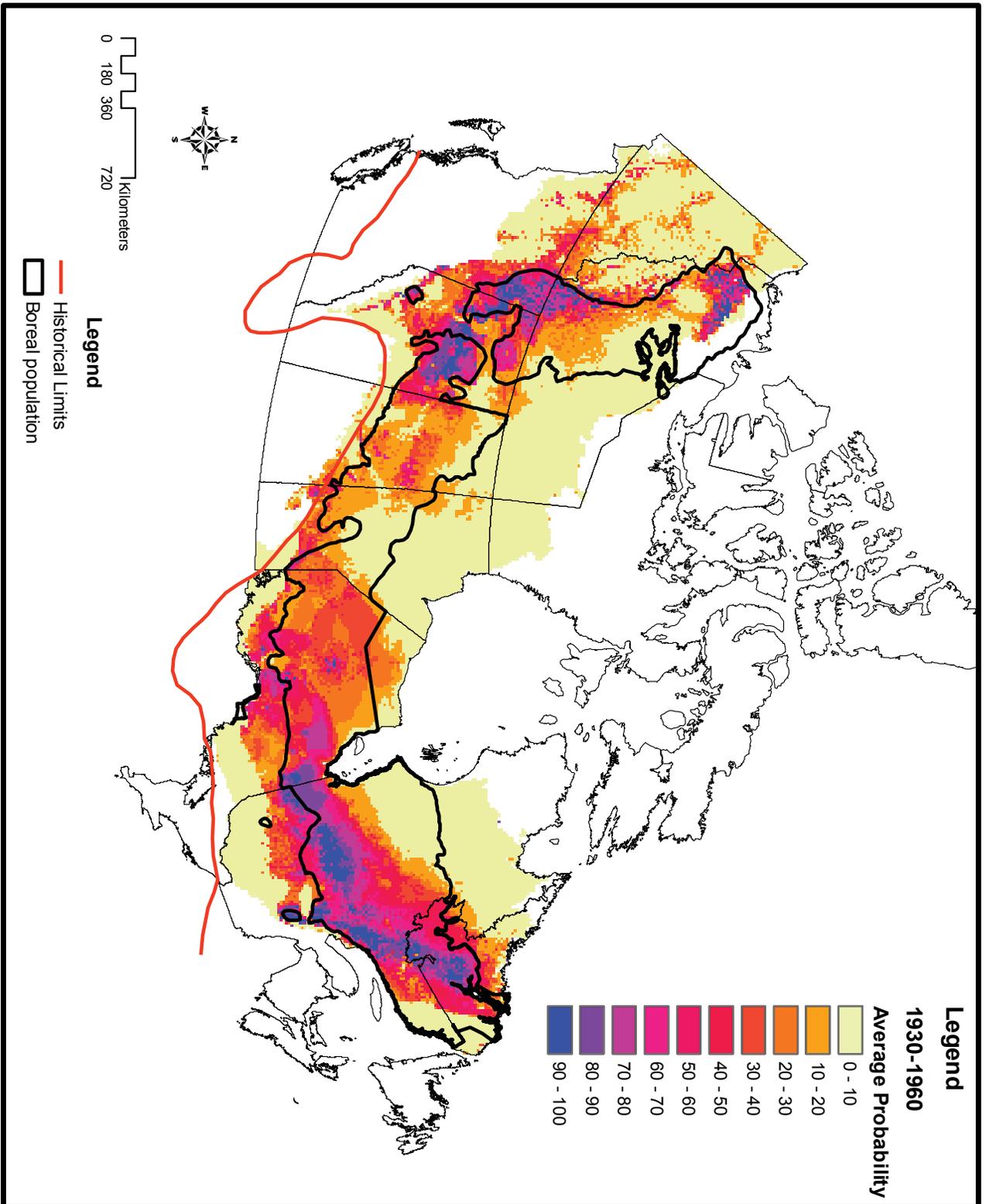


Appendix 6.4 - Table 3. Biotic covariates used in environmental niche models.

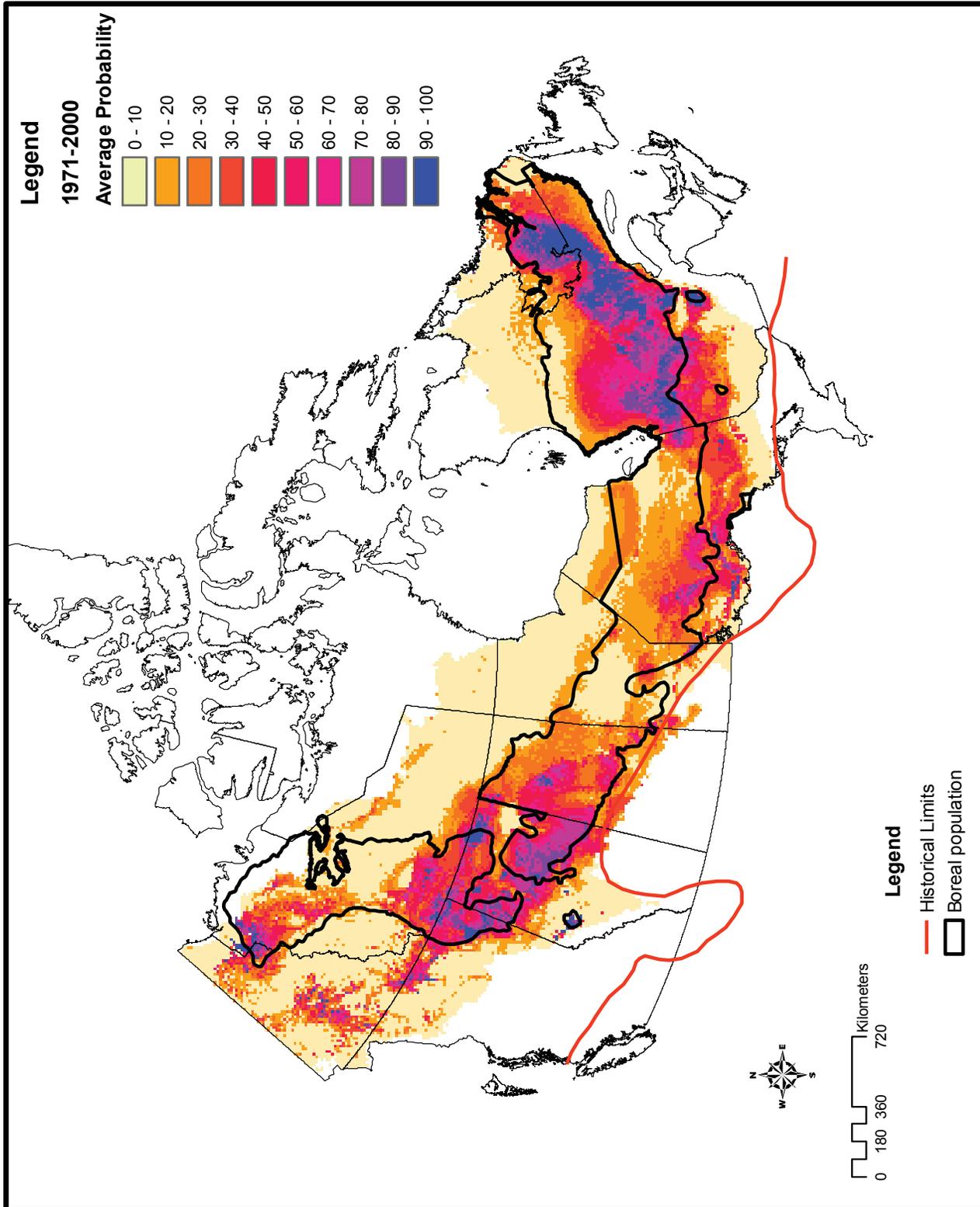
Cumulative Annual fPAR
Minimal Annual fPAR
Landcover
Peatland presence
Road Density
Anthropogenic disturbance
Mean forest patch size
Number of forest patches
Standard deviation of forest patch sizes
Elevation

Biotic Analysis

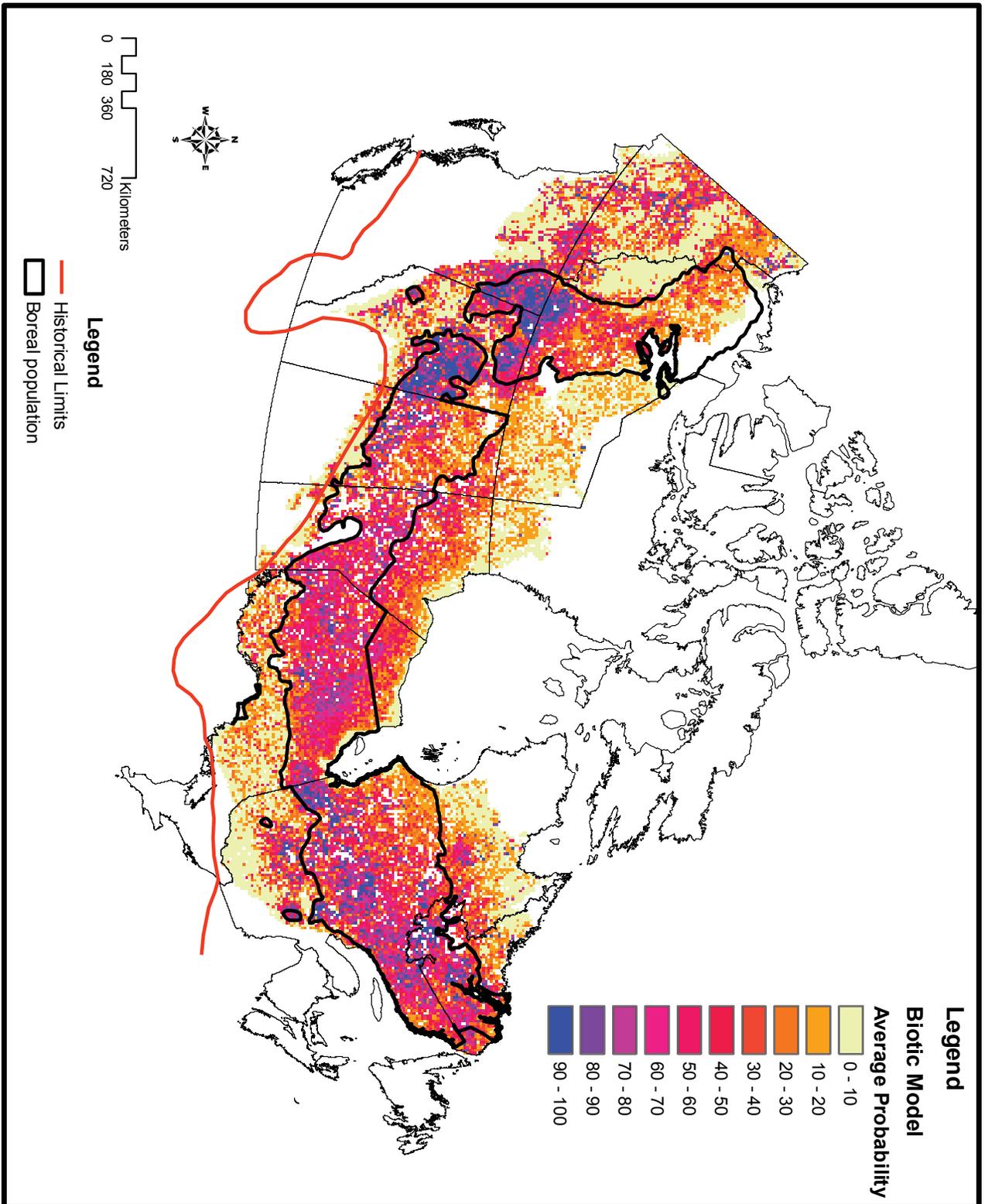
Covariates were screened for collinearity and variables included in the model are listed in Table 3. Mean AUC scores were 0.884 among the ten subsets. Figure 4 shows the cumulative MaxEnt output for the biotic models. Higher probabilities were associated with areas with high numbers of satellite collar fixes, but close examination of Alberta and British Columbia herds shows congruency with the ‘Local Population’ polygons in the National Recovery Strategy (Environment Canada 2007; Figure 5 a), where training data were not available. The model predicted a high probability of occurrence, consistent with the extent of occurrence across the range, with the exception of the distribution in northern Saskatchewan, northern NWT, and the northern part of the Quebec (Figures 4,5b).



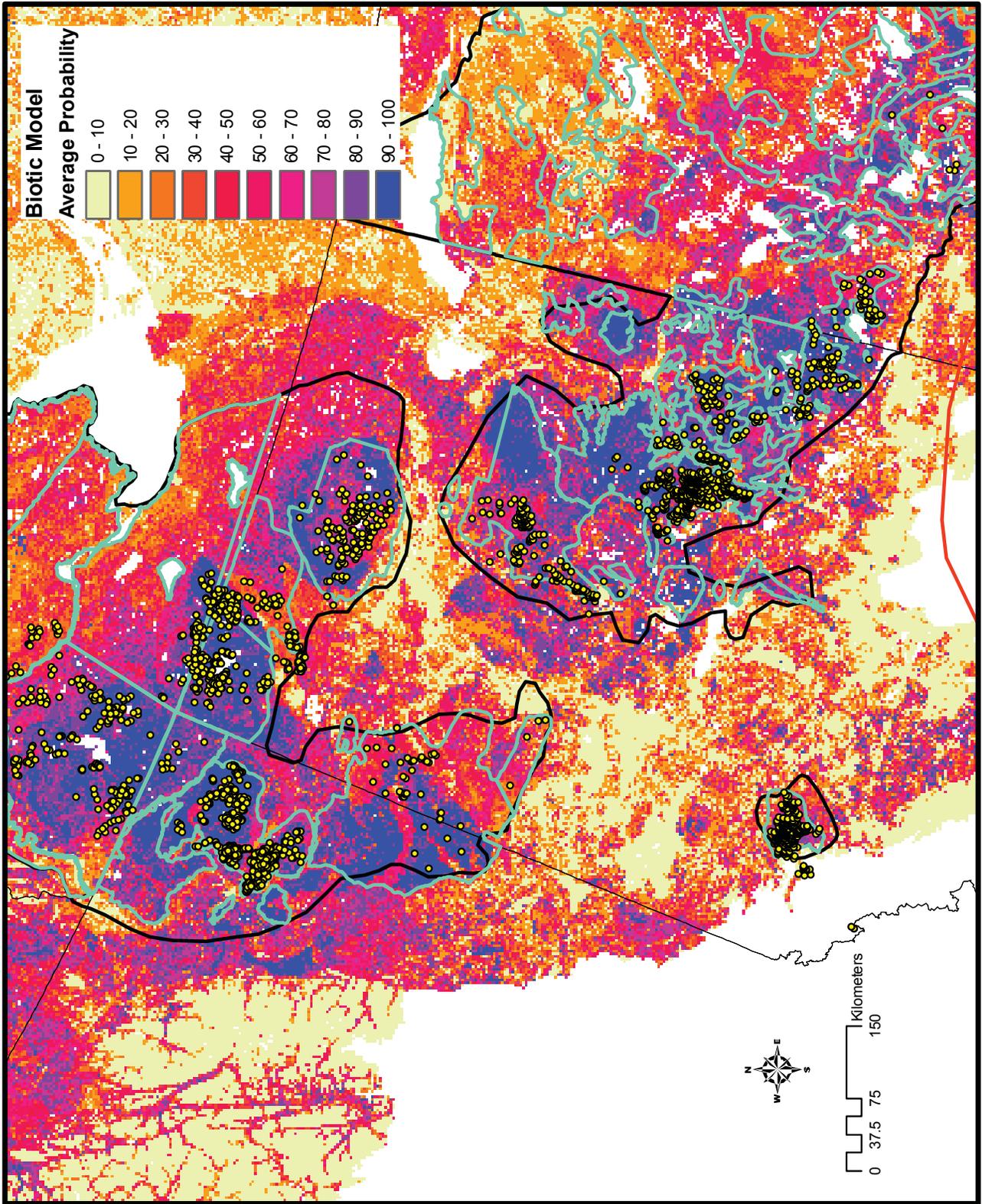
Appendix 6.4 - Figure 2. Cumulative output distributions based on climate variables from 1930-1960. Pixel values are averages from 10 sub-sample model runs.



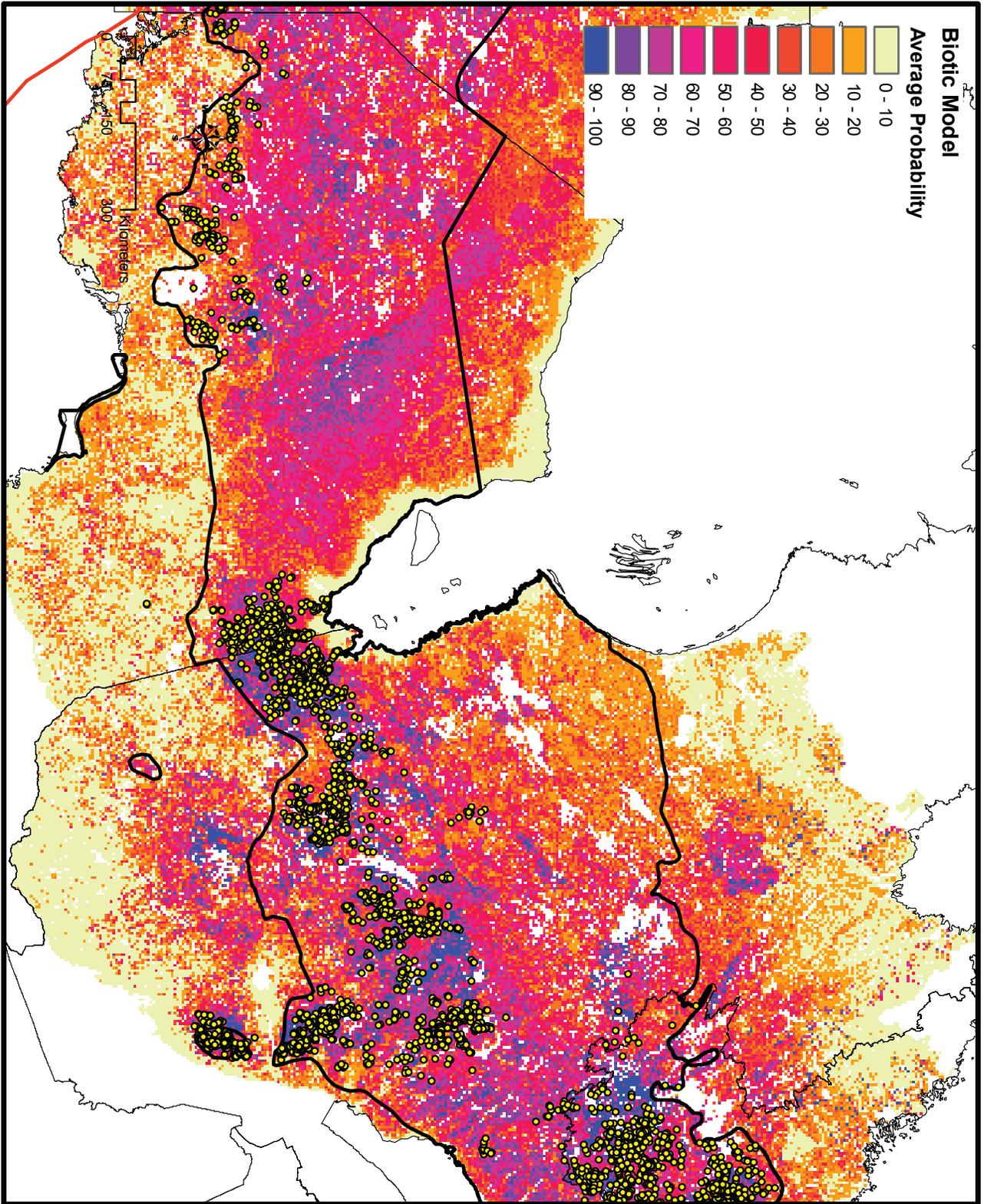
Appendix 6.4 - Figure 3. Cumulative output distributions based on climate variables from 1971-2000. Pixel values are averages from 10 sub-sample model runs.



Appendix 6.4 - Figure 4. Cumulative output from MaxEnt using biotic variables. Pixel values represent the average from 10 subset model runs.



Appendix 6.4 - Figure 5a. Biotic model output in western Canada. Light blue lines are the local population polygons from the respective jurisdictions and the yellow dots indicate collar location used for training data.



Appendix 6.4 - Figure 5b. Biotic model output in central Canada. Yellow dots indicate collar locations used for training data.



Discussion

We found support for our first hypothesis in part of the country but not everywhere. For example, the fundamental niche, or potential distribution, of woodland caribou may have contracted marginally along its southern frontier in Alberta and Saskatchewan. Thus, some minor range contraction may have occurred in these regions owing to climate change in the past 30 years. In Ontario and Quebec, however, the fundamental niche has remained relatively constant and, based on mid-20th century climate data, does not extend to the southern extent of the Boreal Shield Ecozone, as is suggested by the historical distribution of woodland caribou. Our study design called for training datasets to be compiled using radio-tagged animals owing to the large datasets available and the wide geographic distribution across the extent of occurrence, but these data did not exist for the entire time period. Improved estimates of the historical fundamental niche may come with inclusion of other types of locational data (e.g., not telemetry) consistent with the period, which may include animals outside (south) of the current distribution. It is possible that the more southerly habitats comprised a different biophysical fundamental niche space that is not captured in current distribution of animals.

Our second hypothesis that the realized niche is smaller than the fundamental niche was supported in some parts of the country. In Ontario, for example, Figure 3 shows continuous areas of potential habitat for caribou as far south as the entire north shore of Lake Superior, including Pukaskwa National Park. Figures 4 and 5b revealed that areas of potential continuous occupancy that should otherwise be suitable are restricted to some 200-300 km north of the lake, consistent with the current extent of occurrence. A remnant population of boreal caribou exists in Pukaskwa National Park, likely because conditions suitable for their survival continue to persist along the lakeshore, inside the Park. However, forest management of the landscape between the Park and the current more northerly extent of occurrence has eliminated other suitable habitats (Vors et al. 2007). Our results also suggested that some patches of potential habitat exist in this latter area and that movement of individuals between the present continuous extent and Pukaskwa Park may be possible. Refinement of these types of models may help to identify potential areas for connectivity and help determine priority areas for potential rehabilitation via landscape management.

Other studies have modelled population extirpations using niche models by combining climate variables and landcover data (Peterson et al. 2006). Climate, vegetation, and elevation datasets are often related (Hutchinson 1998). For example, in Canada, 'greener' areas get higher rainfall and also have higher temperatures (Ichii 2002). Elevation also shows a close relationship to temperature but the nature of this relationship is variable in space and time (Ichii 2002). Our analysis demonstrated correlation among many climate parameters used as predictor variables for caribou and the annual fraction of Photosynthetically Active Radiation (fPAR) from MODIS, as expected. Inclusion of climate parameters (at 1-km resolution) in the 'realized' niche models effectively 'washed out' the precision of the predictions. In the climate surfaces, pixel values are interpolated from weather station data, whereas satellite-derived data are collected such that a systematic measurement is taken for each pixel. Based on consistent and recent coverage by remote sensing, 1-km biotic variables should reflect spatial



and temporal variation at a higher resolution than the climate data and satellite based models will be more representative of current distribution boundaries (Parra et al. 2004). Within the range of a species, satellite-based models should have less over-prediction (commission error), or higher specificity, that is, higher probability of correctly predicting a cell as absent (Peterson et al. 2004, Parra et al. 2004). Further reduction of commission errors in the biotic models may come from exclusion of old locality records that reflect available habitat at a previous time, but which may have been recently altered. Our training dataset was limited to point locations from the last 20 years to be consistent with timelines used in other areas of the document (Environment Canada 2007), whereas the biotic variables were more recent (last 5 years). Industrial activities that are probably deleterious to caribou populations have increased drastically in some areas over the last 20 years (McKenzie 2006). Restricting location data to be temporally consistent may improve performance of the satellite-based models.

Somewhat unique to our realized niche models was the inclusion of data (disturbance, road density, fragmentation parameters) to account for top-down or predator interactions in limiting species distributions. Hutchinson's n-dimensional niche concept suggested that a species will occupy areas of the fundamental niche where the species is competitively dominant. However, interspecific competition also needs to be considered (Pulliam 2000). Evidence suggests that predation is a major factor in boreal caribou population dynamics and probability of persistence and thus should be considered when modelling caribou habitat occupancy (Sorenson 2008). Many recent satellite-based initiatives and worldwide efforts to maintain access to high quality space-based vegetation data ensure that the economic and timely availability of resource type information for modelling at broad geographic scales is secure (Yang et al. 2006). However deriving accurate and time-specific disturbance layers, such as linear feature density or other industrial activities at the scale required, is difficult and expensive. Improvement in the derivation and inferential capacity of these data and better relationships defining the spatial and temporal scale at which these top down predator interactions occur in caribou populations may improve the occupancy predictions.

A major limitation to any analysis, such as ours, is the geographic bias of locational data available to train the model (Peterson and Cahoon 1999, Johnson and Gillingham 2008, Phillips 2008). Our study design employed many protocols cited to improve model accuracy and reduce bias on model outcomes, including filtering of GPS collar data (Rettie and McLoughlin, Friar et al. 2004), random subsetting and multiple model runs (Araujo and New 2006), and ecological stratification (Reese et al. 2005, Araujo and Guisan 2006). Nonetheless, despite the large contributions of locational data from across the country, the extent of occurrence as outlined in Environment Canada (2007) is not completely sampled (Figure 1). The location of sampling areas highlights another important bias demonstrated, in theory and practice, to affect the outcome of niche modelling. Most studies have been done on caribou populations at the southern end of the range, while other studies have been conducted on low and/or declining populations (Environment Canada 2007). Niche theory and studies performed using environmental niche models suggest that to improve accuracy of predictions, known sink populations should not be included since this habitat may represent marginal niche



space (resources and conditions) for viable populations (Pulliam 2000, Soberon 2007). Sample selection bias due to sampling effort (accessibility) has been shown to dramatically reduce the predictive performance of presence-only models, such as MaxEnt (Phillips 2008). Improved sampling design to represent the entire geographic distribution and attempting to capture the entire niche space of boreal caribou would improve overall model performance and value of the outputs (Jimenez-Valverde and Lobo 2006).

In summary, preliminary results using environmental niche models to study the distribution of boreal caribou at broad scales are important to support Critical Habitat Identification. Species distribution models are increasingly used in conservation planning and management of rare or threatened species to understand the patterns and processes of occurrence on the landscape. The National Recovery Strategy delineates the extent of occurrence of boreal caribou and suggests that some portions of the shaded area (Figure 1) have higher probability of caribou occurrence than others (Environment Canada 2007). The strategy also considers local populations of boreal caribou to be the fundamental units of conservation and management for recovery and action planning. Further refinement and more rigorous validation of the models presented here would contribute to understanding the areas of occupancy and local population ranges within the larger extent of occurrence. The vital rates required for management and recovery of boreal caribou are difficult to obtain because of the large areas that the animals occupy and the low densities at which they exist and because the forested areas that they occupy are difficult to survey with traditional aerial techniques (Environment Canada 2007). Spatial predictions from niche-based distribution models may be used to stratify sampling to increase efficiency. The new data can then be used to improve the original model and performed repeatedly. Such an adaptive process would refine predictions and support management and recovery of local populations at a regional scale. A large range of techniques now exists to predict species distributions, and various studies have demonstrated the predictive capability and accuracy with various types of species and input data availability (e.g. Elith et al. 2006). Presence-only models, such as MaxEnt, may be the most appropriate for rare or threatened species, and caribou in particular, because absences are not likely actual absences but false negatives. Future analyses will focus on model comparisons and reducing data bias to accurately predict boreal caribou occupancy across its extent of occurrence.

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6.5 A National Meta-analysis of Boreal Caribou Demography and Range Disturbance

Preface

A key step in the critical habitat identification process is determining attributes of a caribou range that support or compromise population persistence (e.g. the ability of the range to support a self-sustaining population). This meta-analysis compiled demographic data from boreal caribou populations across Canada to evaluate the hypothesized relationship between caribou population parameters and levels of anthropogenic and/or natural (fire) disturbance on caribou ranges. Results from this work provide quantitative guidelines for one of the three assessment criteria (range disturbance) used in the evaluation of local populations for critical habitat identification.

Introduction

Woodland caribou (*Rangifer tarandus caribou*) are designated a species-at-risk nationally, and in most provinces and territories within their range, due to broad-scale range recession and population declines, in large part associated with human settlement and disturbance (Bergerud 1974, Mallory and Hillis 1998, Schaefer 2003, Vors et al. 2007). This species is closely associated with late-successional coniferous forests and peatlands (Rettie and Messier 2000). These forests are a source of lichens, which comprise the bulk of woodland caribou diet – particularly in winter – but lichen availability is generally not considered a limiting factor (Schaefer and Pruitt 1991, Joly et al. 2003, Courtois et al. 2007). More importantly, these forests provide refugia from predators and other ungulates (Bergerud and Elliott 1986). Many woodland caribou populations are in decline, and the proximate cause is thought to be increased predation. Logging and other disturbances that increase the amount of early seral-stage forest promote higher densities of prey species such as moose (*Alces alces*) and white-tailed deer (*Odocoileus virginianus*), which support higher predator densities, especially wolves (*Canis lupus*) (Bergerud and Elliott 1986, Seip 1992, Stuart-Smith et al. 1997, Racey and Armstrong 2000, Wittmer et al. 2005, 2007). In addition, linear disturbances (e.g. roads, seismic lines) that accompany industrial development in the boreal forest facilitate greater predator mobility and hunting efficiency (James and Stuart-Smith 2000, Dyer et al. 2001, McLoughlin et al. 2003, James et al. 2004). Boreal caribou, an ecotype of woodland caribou, are declining throughout much of their North American range (McLoughlin et al. 2003). Given the increasing levels of industrial development in previously pristine areas, preventing or mitigating further population declines is increasingly the focus of management efforts.

In this study, a simple question is posed: is there a clear relationship between caribou demography and anthropogenic and/or natural (fire) disturbance levels on caribou ranges across the distribution of boreal caribou in Canada? We expected that adult survival, calf recruitment and overall population growth would be negatively related to changes that create favorable habitat for moose and deer, in keeping with the logic that increased primary



prey increases predator density which contributes to caribou population decline. Caribou avoidance of industrial development (Bergerud 1974, Mallory and Hillis 1998, Dyer et al. 2001, Schaefer 2003) and recent burns (Schaefer and Pruitt 1991, Joly et al. 2003, Dunford et al. 2006) have been well documented; thus it is reasonable to postulate that these factors would negatively affect range condition with respect to the ability of an area to support a self-sustaining local population. Wittmer et al. (2007) found that the variation in adult female survival among 10 woodland caribou populations of the arboreal lichen-feeding ecotype was best explained by range condition. Further, in a review of 85 studies that examined impacts of human activity on caribou, Vistnes and Nellemann (2008) concluded that choice of spatial scale for examining impacts strongly influenced conclusions, recommending that accurate assessment required regional-scale studies. Finally, in a recent analysis of 6 boreal caribou populations in Alberta, Sorensen et al. (2008) demonstrated a negative relationship between range condition and population growth rate. Their 2 variable model, which included level of anthropogenic disturbance (%IND) and wildfire (%FIRE), explained 96% of the variation in caribou population growth rates. Hence, our selection of caribou range as the appropriate unit of analysis is justified.

The Sorensen et al. (2008) regression model represents a significant advance in our understanding of the effects of disturbance on caribou demography at the level of population ranges. However, the study was based on a small sample size and a limited range of values for anthropogenic disturbance (e.g. the minimum level of anthropogenic disturbance was 31.6%). As a result, while the data were sufficient to demonstrate significance in terms of a relationship between the dependent and independent variables, the model has limited scope for prediction beyond the geographic area and parameter space included in that study, and should be used cautiously within that region when predicting minimum levels at which negative effects on caribou population growth might occur. The objective here was to extend the Sorensen et al. (2008) analysis to populations of boreal caribou across Canada, in order to test whether the relationship documented was robust across a broader spectrum of range conditions, and guide evaluation of the ability of ranges to support self-sustaining populations. Original work on this study was initiated in 2006, as part of an independent effort to address this question. Augmentation and refinement of this effort was undertaken in conjunction with the Environment Canada scientific review of critical habitat for boreal caribou.

Methods

Data collection – caribou

Researchers and management agencies were approached to supply demographic information on woodland caribou populations that had been studied for a minimum of two years (the smallest interval included in Sorensen et al. 2008), and for which adult female survival (as determined by radio-telemetry monitoring) and/or calf recruitment (late winter calf/cow population surveys) had been measured. The intent was to assemble data that exhibited a broad range of variation with respect to geography and degree of anthropogenic change to population ranges. A tabular data survey with instructions was circulated to



potential contributors. Information on 25 boreal populations from 7 provinces and 1 territory was acquired (Figure 1). There was considerable variability in the intensity and duration of sampling, and availability of ancillary information.

Estimates of population condition

Of the 25 populations included in this study, data for assessing female survival and therefore estimating population growth rates were available for 15 (Table 1). Some populations had only a small number of female caribou collared and concomitant high variability in estimated survival. Therefore, to maximize the number of populations available for analysis, estimates of recruitment rates, which were available for all populations, were used as a surrogate of 'population condition'. Bergerud and Elliot (1986, 1998) demonstrated that recruitment was directly related to population rate-of-growth in caribou, as well as in other ungulates. Furthermore, recruitment may be a better short-term indicator of population condition in rapidly changing landscapes than either female survival or population growth rate, given that calves are more susceptible to predation than adults, and high adult survival could initially mask the negative effects of landscape change.

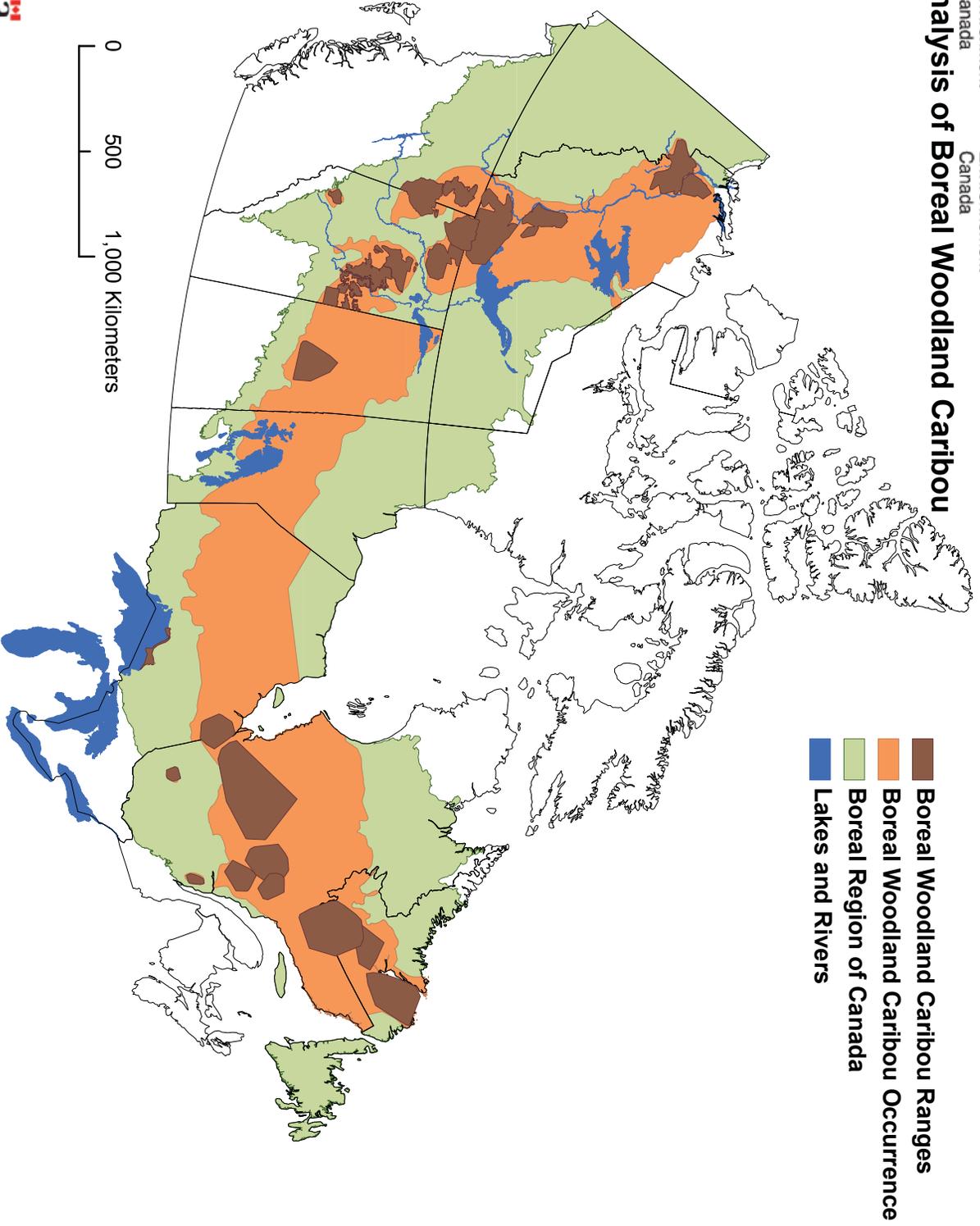
To test the relationship between recruitment and population growth, and the appropriateness of using recruitment as the response variable to range condition in the regression analysis, data from the subset of populations for which recruitment and survival were available were used to estimate population rate of change (λ) following Hatter and Bergerud (1991); see also McLoughlin et al. (2003) and Sorensen et al. (2008). However, because averages and not annual data were provided for some local populations, an arithmetic, rather than geometric mean (McLoughlin et al. 2003. Sorensen et al. 2008) was used to estimate average values for each parameter over the years of study included for each population (Table 1). Data for some populations were sub-sampled to be temporally consistent with available data on landscape change; in particular, to avoid inclusion of demographic data that potentially preceded the change. Also, some populations with long-term data exhibited trends suggesting that an average over the entire sampling interval was not representative of the current population condition. Where available, up to 4 years of most recent data, spanning a maximum sampling interval of 5 years, and with greatest temporal correspondence to the landscape change data, were used to estimate demographic parameters for analysis (Table 1). The 6 Alberta populations included in Sorensen et al. (2008) were also included in this study; however, the sampling intervals differed (1993-2001 vs. 2002-2006). Thus, it was possible to also evaluate the relationship between recruitment and population growth for a second subset of temporally non-overlapping data, based on Sorensen et al. (2008).

Delineation of population ranges

Range boundaries were provided by contributors for study populations, obtained from provincial or territorial sources for jurisdictionally-recognized population ranges, or generated from 100% minimum convex polygons (MCPs) of telemetry data provided by contributors. Delineation method is indicated in Table 1 and illustrated in Figure 1. Where a study population



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Meta-Analysis of Boreal Woodland Caribou



Appendix 6.5 - Figure 1. Location of 25 boreal caribou populations included in this study.



corresponded closely to a jurisdictionally recognized range (e.g. $\geq 90\%$ correspondence), the data were considered representative of the range, and the jurisdictional boundary was used for population delineation and characterization of range condition.

Characterization of range condition and model specification

Following Sorensen et al. (2008), the relationship between recruitment and range condition was evaluated by comparing three candidate models. Model 1 considered the percent of the range area burned within the past 50 years of the most recent recruitment data for each population. Fire data from the Canadian Large Fire Database, augmented by additional coverage for the Northwest Territories, that contained wildfires >200 ha (NRCan 2008, GNWT 2008) were used. Model 2 considered the percent of the range area affected by anthropogenic disturbance, based on GIS layers obtained from Global Forest Watch Canada (GFWC). GFWC have compiled the only available, nationally-consistent coverage of anthropogenic disturbance across forested regions of Canada. All visible linear and polygonal anthropogenic disturbances were digitized from Landsat images from the period 1985–2003, and combined with additional coverage of roads, reservoirs and mines from databases spanning the period 2002–2006. Linear disturbances included roads, railroads, seismic lines, pipelines, and utility corridors; polygonal features included recently anthropogenically-converted areas such as settlements, populated industrial areas, croplands (both new and abandoned), reservoirs, cutblocks, and mining activity. All features in the database were buffered by 500 m to create a “zone of influence”, and merged to create a non-overlapping coverage of all anthropogenic disturbances. Detailed methodology is available from Lee et al. (2006). Sorensen et al. (2008) used a 250-m buffer when quantifying human disturbance. However, we did not have access to the raw data used in the GFWC analysis, so could not select an alternate or varying buffer width. Nevertheless, in a review of reindeer and caribou response to human activity from regional-scale landscape studies, Vistnes and Nellemann (2008) report reduced use by caribou of areas within 5 km of infrastructure and human activity, thus the 500-m buffer is not unreasonable. Lastly, Model 3 considers the combined effect of fire and anthropogenic disturbance, herein termed total disturbance.

Characterization of total disturbance and modeling procedure

Sorensen et al. (2008) used a 2 variable model to characterize total disturbance (%FIRE and %IND); however, they found a relatively high correlation between these 2 variables (Pearson correlation of 0.69) which tends to produce least-squares estimates that are exaggerated in absolute value (Montgomery et al. 2001). Multi-collinearity between these 2 variables could also influence parameterization because of the likely non-linear relationship between the proportion of area disturbed and the level of spatial overlap. Specifically, at low levels of disturbance the spatial overlap is likely to be low whereas the likelihood of overlap should increase at higher levels of disturbance. Visual inspection of the data revealed such a pattern. Therefore, to describe total disturbance when testing the hypothesis of primary interest (e.g. the combined effects of fire and anthropogenic disturbance), the merged mapped of non-overlapping disturbances was used to derive a single measure of total disturbance. This



Appendix 6.5 - Table 1. Location, sampling duration, method of range delineation (J = jurisdiction; SA = study area), yearly ratio of calves per 100 cows (R), annual adult female survival (S), and rate of population change (λ) for 25 boreal caribou populations in Canada.

Local Population	Prov/Terr	Years Available	Sample	# Years	Years Used	Range	R	S	λ
Red Wine	NL	1981-1988, 1993-1997, 2001-2003	Y	3	2001-2003	J	45.4	n/a	n/a
Mealy Mountain	NL	1971, 1974-1975, 1977, 1981, 1985, 1987, 1994, 2002, 2005	Y	2	2002, 2005	J	50.3	89.0	1.19
Lac Joseph	NL	2000-2002, 2005, 2007, 2008	Y	4	2000-2002, 2005	J	34.3	n/a	n/a
Val-d'Or	QC	1987-1988, 1990-1991, 1995-2002, 2004-2005	Y	4	2001-2002, 2004-2005	J	15.3	87.0	0.94
Manicouagan	QC	1999-2001	N	3	1999-2001	J	50.5	75.0	1.00
Manouane	QC	1999-2001	N	3	1999-2001	J	28.1	86.0	1.00
Pimouacan	QC	1999-2001	N	3	1999-2001	J	40.6	82.0	1.03
Charlevoix	QC	2000-2001, 2004-2006	Y	4	2001, 2004-2006	J	35.0	n/a	n/a
Jamesie	QC	2002-2003	N	2	2002-2003	SA	27.4	n/a	n/a
James Bay	ON	1998-2000	N	3	1998-2000	SA	21.3	79.0	0.88
Pukaskwa	ON	1973-1991, 1997, 1999, 2001	Y	3	1997, 1999, 2001	J	40.3	n/a	n/a
Smoothstone- Mapawekka	SK	1993-1995	N	3	1993-1995	SA	28.0	84.0	0.98
Carbou Mountain	AB	1995-2007	Y	4	2003-2006	J	17.4	75.0	0.82
ESAR	AB	1994-1997, 1999-2007	Y	4	2003-2006	J	13.4	86.6	0.93
Red Earth	AB	1995-1997, 1999-2007	Y	4	2003-2006	J	13.6	81.9	0.88
WSAR	AB	1994-2007	Y	4	2003-2006	J	20.9	84.2	0.94
Little Smoky	AB	2000-2007	Y	4	2003-2006	J	12.3	82.2	0.88
Cold Lake	AB	1999-2002, 2004-2007	Y	4	2002, 2004-2006	J	12.6	83.8	0.89
Chinchaga	AB	2002-2007	Y	4	2003-2006	J	13.9	87.0	0.93
Snake-Sahaneh	BC	2004-2005	N	2	2004-2005	J	7.2	94.0	0.97
Cameron Hills	NWT	2006-2008	N	3	2006-2008	SA	16.4	n/a	n/a
Dehcho North	NWT	2006-2008	N	3	2006-2008	SA	20.7	n/a	n/a
Dehcho South	NWT	2006-2008	N	3	2006-2008	SA	32.3	n/a	n/a
GSA South	NWT	2004-2006	N	3	2004-2006	SA	28.9	n/a	n/a
GSA North	NWT	2005-2006	N	2	2005-2006	SA	45.4	n/a	n/a



method captured the required information from each variable while accounting for the spatial overlap, and increased the power of the test by reducing the number of variables in the model.

Linear regression and related diagnostics were used to test the relationship of recruitment to each measure of range condition specified by the three models. Similarly to Sorensen et al. (2008), herds were considered to be independent and Akaike’s Information Criteria (AIC) with correction for small sample sizes (AICc) was used to test between the three candidate models (Burnham and Anderson 1998).

Results

Estimates of population condition

Recruitment was positively correlated with population rate of change for both the subset of data evaluated here ($r=0.75$; $p<0.01$) and the Sorensen et al. (2008) data ($r=0.63$; $p<0.01$). Regression analysis yielded very similar constants and coefficients (Table 2). Recruitment was not correlated with adult female survival in either data set. Exploratory analysis of the subset of 15 populations further revealed recruitment to be more sensitive to % anthropogenic disturbance and % total disturbance than either adult female survival or population growth rate. Use of recruitment as an index of population condition for subsequent analyses of main models therefore seems reasonable.

Appendix 6.5 - Table 2. Regression statistics for analysis of mean annual recruitment versus population growth rate for a 15 population subset of data compiled for this study and 6 Alberta populations included in Sorensen et al (2008).

Data Source	R ²	β_0 intercept	SE	P	β_1 (X ₁)	SE	P
15 population subject	0.56	0.84	0.030	<0.001	0.005	0.001	0.001
Sorenson et al. (2008)	0.40	0.84	0.033	<0.001	0.007	0.002	<0.001

Regression diagnostics and data selection for main models

For the full data set, outliers were examined visually and tested for leverage and influence (leverage versus normalized residual squared plots) with DFBETA (STATA 8.0), which assesses how the coefficient is affected by deleting each of the observation values (values exceeding $2/\sqrt{n} = 0.4$ are of concern). Only Charlevoix had a DFBETA value above the model cut-off in Model 3 (Charlevoix DFBETA = 0.70). Given that was the only data point that significantly affected estimation of the regression coefficient, and that it was also the sole reintroduced population, it was excluded from further analyses.

There was no evidence of heteroscedasticity in the residuals of any of the models (White’s test and Breusch-Pagan test, STATA 8.0). Residuals from Models 1 and 2 met conditions of normality; however, residuals from Model 3 significantly deviated from normality (Shapiro-



Wilk test of normality, $P = 0.01$). Log transformations of the variable total disturbance were considered, as well as the addition of a squared term, to examine potential non-linear forms of the relationship. Neither of these options increased the fit of the model. Therefore, the linear form was retained due to ease of interpretation, and a lack of knowledge concerning the true form of the underlying distribution.

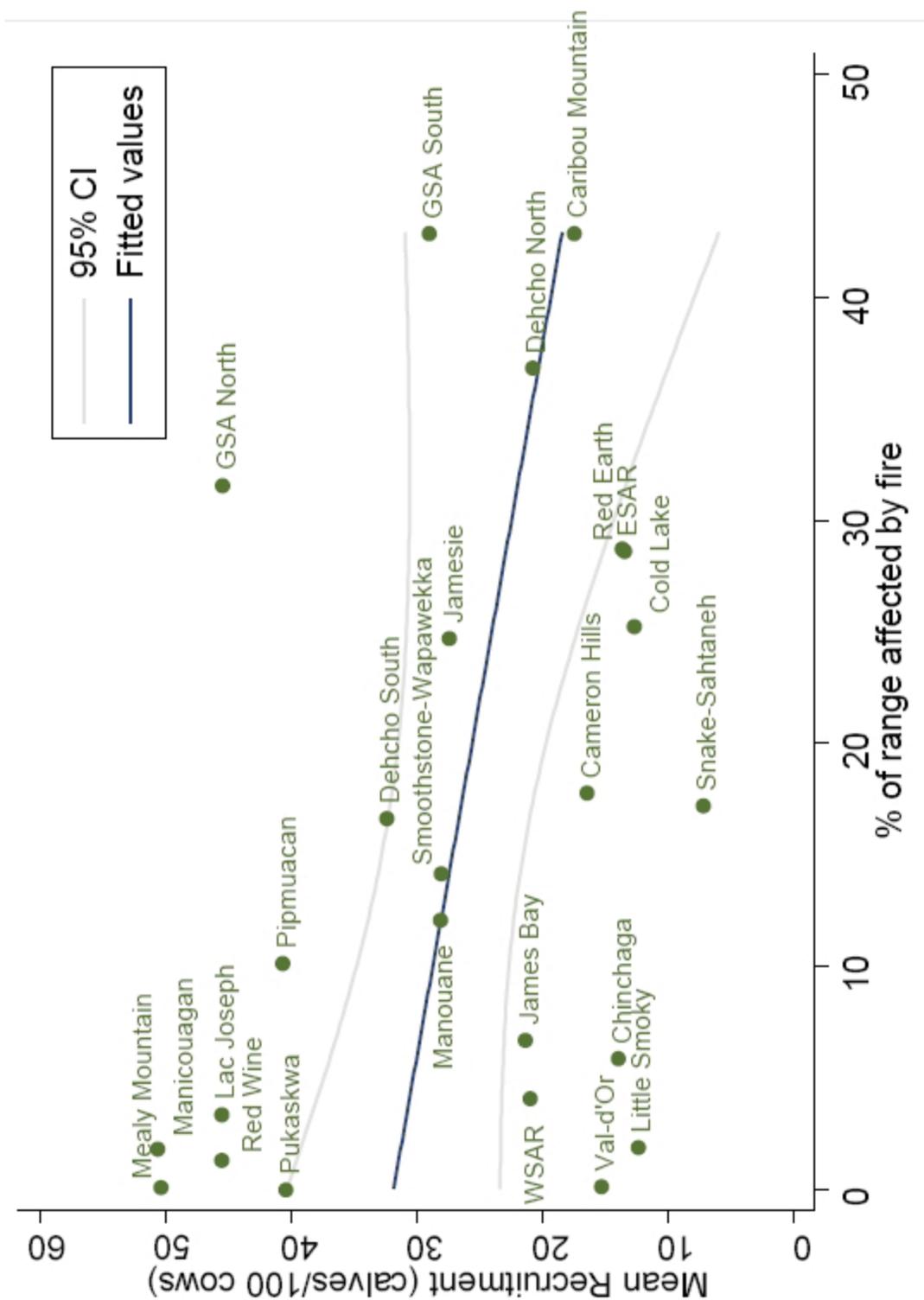
Regression results

There was no significant relationship between caribou recruitment rate and the percent area disturbed by fire alone ($F_{1,22} = 2.52$, $p = 0.13$; Model 1, Table 3; Figure 3). However, there were significant negative relationships between recruitment and the percent area affected by anthropogenic disturbance ($F_{1,22} = 20.21$, $p < 0.001$; Model 2, Table 3; Figure 4) and with the merged measure of total disturbance ($F_{1,22} = 34.59$, $p < 0.001$; Model 3, Table 3; Figure 5). Model 3, the measure of total disturbance, had the lowest AICc value and best fit with population recruitment rates (Table 3, Figure 5).

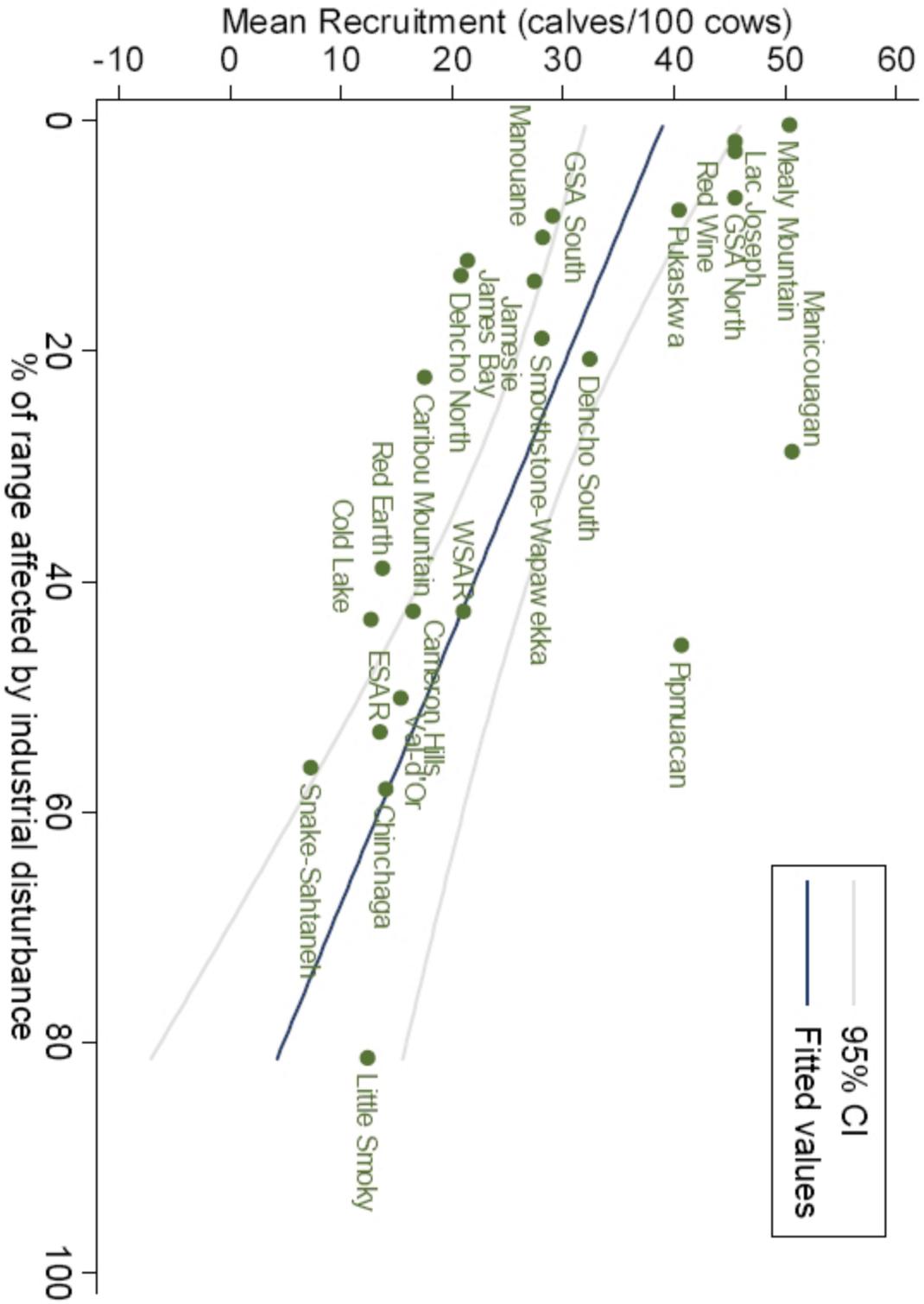
Appendix 6.5 - Table 3. Regression statistics for analysis of mean annual recruitment versus parameters of range disturbance for boreal caribou populations across Canada ($n=24$).

Model	R ²	β ₀ intercept	SE	P	β ₁ (X ₁)	SE	P	AIC _c
1 - % fire	0.10	31.86	4.10	<0.001	-0.31	0.20	0.13	54.81
2 - % anthropogenic	0.49	39.13	3.40	<0.001	-0.43	0.10	<0.001	49.15
3 - % total disturbance	0.61	46.37	3.75	<0.001	-0.49	0.08	<0.001	46.09

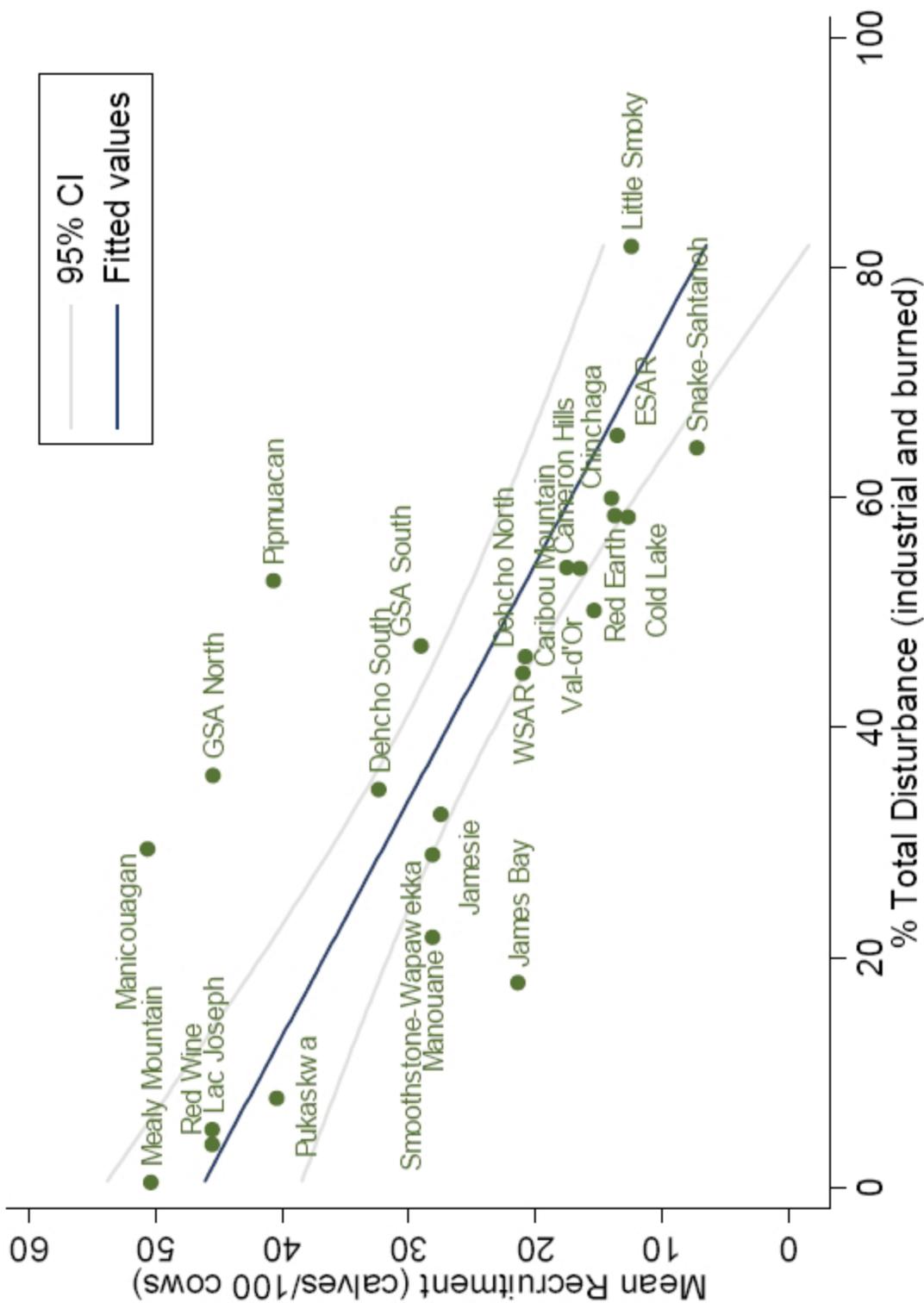
There was no clear pattern between the size of population ranges or study areas and the observed relationship between recruitment and total range disturbance (Figure 6).



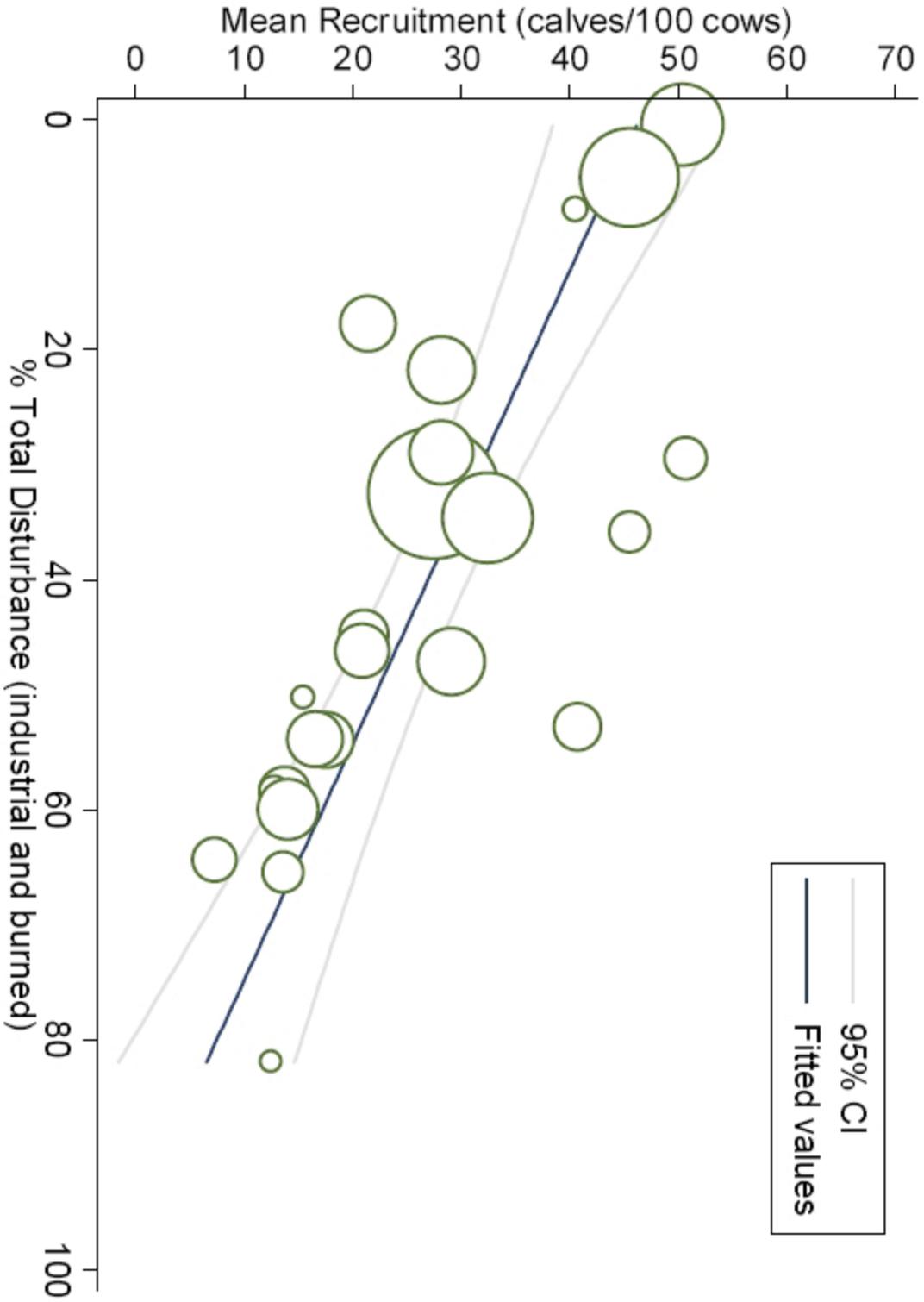
Appendix 6.5 - Figure 3. Linear regression of mean caribou recruitment versus the percent of range disturbed by fire within 50 years of the most recent demographic data (n = 24). The relationship is not significant (P=0.13).



Appendix 6.5 - Figure 4. Linear regression of mean caribou recruitment versus the percent of range affected by anthropogenic disturbance (n = 24).



Appendix 6.5 - Figure 5. Linear regression of mean caribou recruitment versus the percent of range affected by fire and anthropogenic disturbance, accounting for spatial overlap of the variables (n = 24).



Appendix 6.5 - Figure 6. Linear regression of mean caribou recruitment versus the percent of range disturbed by fire and anthropogenic disturbances, accounting for spatial overlap of the variables (n=24). The size of circles represents the relative size of individual ranges or study areas (see Table 1).



Discussion

This is the first analysis of caribou demography and range disturbance at the scale of the national distribution of boreal woodland caribou in Canada. We found a clear negative relationship between caribou recruitment, as measured by calf/cow ratios, and the level of disturbance within caribou ranges. Total disturbance (non-overlapping burn and anthropogenic disturbance) was the best predictor of boreal caribou recruitment rates. As in Sorensen et al. (2008), the extent of anthropogenic disturbance appeared to be the main driver of this relationship, also reflecting results from other studies where the level of anthropogenic disturbance influenced caribou distribution and persistence (Courtois et al. 2007, Schaefer and Mahoney 2007, Vors et al. 2007, Wittmer et al. 2007).

The relationship between recruitment rate and proportion of range disturbed by fire was less clear. The percent area burned within caribou ranges was not a significant predictor of recruitment rate by itself, but its merger with the anthropogenic disturbance layer did improve model fit. Similar to anthropogenic disturbances, fires affect the amount, composition and age structure of forest available to caribou, although the effect on configuration may be different; that is, disturbance by fire tends to be more aggregated and thus result in less fragmentation of remaining areas (e.g., Schmiegelow et al. 2004). Spatially, fires are represented as polygons of disturbance without consideration of severity; however, fires in boreal forests are highly variable, and often result in mosaics of burned and unburned patches within the mapped fire boundary (Smyth et al. 2005, Schmiegelow et al. 2006). This variability is likely to result in differential effects on habitat quality for caribou, dependent on their immediate effects on lichen and other forage, the post-disturbance trajectory of burned areas, and the indirect effects of disturbance by fire on habitat suitability and resultant numerical response by predators and apparent competitors. Nevertheless, the main question is how disturbance by fire differs from anthropogenic disturbances with respect to demographic response by caribou. In this regard, a conspicuous difference is the absence of linear features in naturally disturbed areas. As a result, fires are unlikely to elicit the functional response by predators attributed to increased travel and hunting efficiency in association with linear anthropogenic disturbances (James and Stuart-Smith 2000, James et al. 2004, Dyer et al. 2001, McLoughlin et al. 2003). They are many other aspects that could be examined, such as post-disturbance successional trajectories following fire or harvest, but comprehensive treatment is beyond the scope of the present exercise.

One methodological consideration is the 50-year window for quantifying disturbance by fire. The 50-year interval is consistent with Sorensen et al. (2008), and with the anticipated duration of effects on caribou from several studies (Klein 1982, Schaefer and Pruitt 1991, Dunford et al. 2006), but represents a discrete cut-off when extracting the disturbance data. For example, a large fire that burned 51 years before the last year for which demographic data were available would not have been included in the disturbance estimate for that local population range. Similarly, 49 year-old and 1 year-old fires were considered identical within a range, and no consideration was afforded across ranges to potential variability in the duration of impacts. Future analyses should consider a variable or moving window for measuring



this disturbance at the level of individual ranges, and given the large geographic extent over which the species is distributed, where possible incorporate information on variability in post-fire regeneration and recovery.

Measures of both anthropogenic and natural disturbance in this study were arguably conservative, due to a requirement to use nationally-standardized data sets. The Global Forest Watch Canada data were restricted to detection of features readily identified from mid-resolution satellite imagery (1:40,000–1:50,000 scale; overall pixel resolution of 28.5 m), and the Canadian Large Fire Database includes only fires >200 ha in size. Thus, narrow and small disturbances were not captured. Furthermore, the most recent anthropogenic disturbance data included were to 2005, and some features were current only to 2003. Effort was made to match demographic data to the disturbance layers; however, data availability was a constraint. In ranges experiencing high rates of change, the level of disturbance may have been underestimated, particularly when demographic data were very recent. Regardless, the strength of our analyses includes the standardization of disturbance measures across ranges, and the repeatability of the procedure. Finally, while our analyses revealed some fundamental relationships with a parsimonious explanation, our disturbance measures captured only a subset of the attributes that affect range condition, and a better understanding of additional range attributes could help explain variation in the observed relationships at a national scale. It is also important to note that our measures of disturbance accounted only for conspicuous changes to forest cover that could be derived from national-scale data and mapped. Some caribou ranges in Canada experience other forms of disturbance that may compromise population condition and/or affect range use. For example, low level aircraft traffic can affect caribou reproduction (Luick et al. 1996, Maier et al. 1998) and calf survival (Harrington and Veitch 1992). Over-hunting can also drive populations into decline (Bergerud 1967, 1974).

Of the models evaluated, total disturbance, expressed as proportional amount of range affected, was the best predictor of observed recruitment levels in caribou, explaining 61% of the variation in this parameter. An assumption implicit in the use of a simple model is that areas within population ranges or study areas that are not burned or impacted by anthropogenic features are equally good for caribou, which may or may not be the case. Exploring the variability in response across ranges, closer examination of the specific conditions on individual ranges, and consultation with biologists familiar with local circumstances, could help to identify reasons underlying populations falling outside the confidence intervals of the regression, and generate additional hypotheses about measures affecting range condition for evaluation in future analyses. An obvious additional attribute of disturbance that could be quantified using existing data is the spatial configuration of disturbances within caribou ranges, and their effect on measures of connectivity and patch size. There exists both theoretical and empirical evidence to suggest that, at the same level of disturbance, a more dispersed spatial pattern would lead to greater fragmentation of the range, greater interspersions of high quality caribou habitat with that suitable for other species, increased accessibility of the range by predators, and thus an overall decrease in available refuge areas for caribou, leading to negative effects on population condition.



The measure of population condition employed in this study was recruitment, for which the most extensive data set was available. Exploratory analyses revealed good correspondence between recruitment and population growth for a subset of the available data. However, recruitment was not correlated with female survival, as suggested for caribou populations in previous studies (e.g., Bergerud 1988). We had earlier hypothesized that a disjunct might exist. Future analyses should explore the relationship between recruitment and other population parameters through empirical and simulation studies. To be of greatest utility to management, demographic analyses should focus on the co-variation between vital rates and habitat variables (Boyce et al. 2005), in this case measures of range condition. There are several important outcomes from such work. First, it would increase understanding of the relationship between the components of population growth and their interaction with range condition, and identify uncertainties that could become the focus of future adaptive management experiments. Second, it would inform monitoring schemes for caribou, such that the data collected represent the most cost-efficient and effective measures of population condition. The development of long-term, standardized monitoring programs and protocols would produce consistent estimates that maximize the information available for future analyses.

Previous work suggests that population response may lag behind landscape change by up to several decades, due to the proximate factors responsible (Vors et al. 2007). Effects on caribou populations mediated by changes in competitors and predators can take some time to emerge, as numerical response by these species is not immediate. Our analyses did not address potential time lags in population response to changing range condition, as the Global Forest Watch Canada (GFWC) anthropogenic disturbance data could not be partitioned into time intervals. However, GFWC is presently completing a landscape change analysis, which quantifies anthropogenic changes over the time intervals 1990-2000, and 2001-2007. These data will facilitate investigation of caribou population dynamics relative to rates of change, as well as exploration of potential time lags in response.

A primary objective of the present study was to extend the Sorensen et al. (2008) analysis to a broader range of population and landscape conditions. The general model structure employed for each study was similar; however, different measures of both the independent and dependent variables were evaluated. Thus, it is not appropriate to quantitatively compare specific model outputs. Nevertheless, both studies posed the question: is there a relationship between human-caused disturbance and caribou population performance? The answer is affirmative. There is an increasing risk to caribou population persistence as the level of anthropogenic disturbance increases, and disturbance by fire interacts with this, such that the total disturbance on a caribou range must be considered when developing management guidelines. The results further suggest that it is possible to establish quantitative guidelines for disturbance thresholds relative to probability of population persistence, even though the mechanisms underlying the relationship may not be fully understood. Ultimately, the evaluation and management of habitat must be tied to demographic responses, like recruitment. Assembling and analyzing information from multiple populations – the product of many years of effort from many individuals - is one means to generate such vital knowledge.



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6.6 Non-Spatial Population Viability Analysis

Introduction

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) listed the boreal caribou ecotype as threatened in 2002 (Thomas and Gray 2002). Causes of the decline of boreal caribou populations include over-harvesting by humans and habitat alteration at the landscape scale that favours early seral stage forests and their associated prey species and predators (Environment Canada 2007). Key objectives of the national recovery strategy for boreal caribou are to prevent extirpation of local populations and to maintain or enhance habitat conditions to allow these populations to be self-sustaining (Environment Canada 2007). Concern about the long-term persistence of boreal caribou populations raises questions about the relative role of various vital rates and population size in maintaining populations of boreal caribou.

Deterministic and stochastic processes may cause populations to decline (Caughley 1994). Overharvesting, human-induced habitat loss and fragmentation, and predation are deterministic factors that may reduce population size (Diamond 1984, 1989). Once populations are small and isolated, they are vulnerable to demographic and environmental stochasticity, which may further reduce numbers and cause genetic isolation (Shaffer 1981, 1987, Lande 1988, 1993). The interaction of deterministic and stochastic factors may contribute further to endangerment, described as an extinction vortex (Gilpin and Soule 1986). Stochastic factors may cause small populations to become extinct, even if habitat conditions are adequate and deterministic causes of decline are removed (Shaffer 1981). Catastrophes (such as large forest fires) are considered to be an extreme form of environmental stochasticity that cause major reductions in populations and thus have important implications for any size of population (Lande 1993).

Habitat conditions directly affect the demographics of boreal caribou populations. Habitat alteration at the landscape scale, favouring early seral-stage forests and their associated prey species and predators, can result in declines in survival rate in boreal caribou (Wittmer et al. 2007). Reduced adult survival and recruitment increases the risk of extinction. Exploring how boreal caribou life history and vital rates influence population persistence in different habitat situations aids in our understanding of habitat conditions that may allow boreal caribou populations to be self-sustaining.

The Boreal Caribou Critical Habitat Science Review pursued four analytical approaches to support the critical habitat decision framework; here we report on one of these, a non-spatial population viability analysis (PVA). The objective of this work was to use non-spatial models to assess how population persistence is affected by aspects of boreal caribou life history and population structure, using the range of population vital rates and their variance that have been recorded for boreal caribou across their distribution. This work informs the Critical Habitat decision analysis by assessing population sizes required for persistence under various demographic conditions and by providing a tool to investigate the effects of altered vital rates on the population dynamics of boreal caribou.



Using a Leslie Matrix Model, we assessed the effects of variation in boreal caribou vital rates on population dynamics and persistence. Specifically, we explored the following questions:

- 1) What is the critical population size that will ensure persistence under environmental and demographic stochasticity and various combinations of adult and calf survival rates reported in the literature?
- 2) Of adult female survival, calf survival and their coefficients of variation (CV), which parameter has the greatest relative contribution to the probability of extinction?
- 3) How do recruitment rates affect the relative risk of extinction under various population sizes and adult female survival scenarios?

Methods

We used a two-stage, female-only Leslie matrix model with pre-calving census to model the population dynamics of boreal caribou. The model, BWCSim1.0 (Boreal Woodland Caribou Simulator; J. Tews unpubl.), was developed using Borland C++ Builder 5.0 Professional. The calculated intrinsic growth rate (λ) was based on a deterministic projection of the stage matrix (Caswell 2001). Density dependence was incorporated as a logistic Ricker equation assuming a maximum finite rate of population increase (λ) of $\lambda = 1.3$. Density dependent population growth is affected when abundance reaches the carrying capacity (K); below K vital rates of the stage matrix are unchanged. Fecundity was modelled as recruitment of female calves to yearlings per adult female and calculated at $t+1$ as: parturition rate * sex ratio * survival rate (0 -1 yrs).

We used stage-specific (calf, yearling, and adult) demographic data for boreal caribou available from published literature to populate the model (Table 1). We calculated the mean, minimum, and maximum values for female calf and female adult survival and corresponding coefficients of variation (CV; Table 1). From each study, we calculated each individual CVs using one of three approaches: 1) for studies that reported SE or 95% confidence intervals (CI) that were symmetrical around the estimate, we calculated CV as SE/Parameter Estimate; 2) for studies reporting 95% CI that had been calculated using bootstrapping or other techniques (making the back-calculation of CVs impossible), we divided the CI by 4 to obtain a rough estimate of SE and then calculated the CV as above; or 3) for studies that reported a CI that was asymmetrical or its upper bound was truncated to 1 (e.g., survival rates), we determined the difference between the mean and the upper or lower CI bounds, whichever had the highest value. We then estimated the CI as equal to twice that value and then calculated the corresponding CV.

A number of additional parameters were necessary to run the models (Table 2). We assumed that: adults represented 70% of the population, females represented 61% of adults, yearlings were 14% of the population, and calves were 16% of the population, and female adults and yearlings comprised 50% of the population¹, based on the means of values reported in the literature (Table 1). We set the proportion of calves that were female produced each year at 0.50 (Gustine et al. 2006) and, in the absence of published data for the proportion of female

¹ Adults in population = Total population (100%) – Yearlings (14%) - Calves (16%) = 70%; Adult females = 70% * 61% = 42.7% of population; Yearling females = 50% sex ratio * 14% = 7% of Population; Adult females + yearling females = 42.7% + 7% = 49.7 or \approx 50% of population.



yearlings, we also set this value at 0.50. The model generated a stable age distribution for the initial population (N_i) based on survival rates and N_i . We estimated that yearling females and adult females represented ~50% of population. Given that BWCSim1.0 predicts female abundance only (e.g., adults + yearlings), we doubled female abundance values predicted by the model to obtain total population sizes (including males, see footnote Table 1 for calculation). For all results, we reported total population size.

We set parturition rate for adults (>2 yrs old) at 0.76 based on the mean of values reported in the literature (Table 1). Caribou typically have their first calf at age 3, but earlier reproduction (as early as 2 yrs.) has been reported (Bergerud 1980). Consequently, we set the yearling parturition rate at 0. Although variations of parturition rate and calf sex ratio were not reported in the literature, we assigned a CV of 0.10 to both parameters under the assumption that they do vary.

We modelled simulated populations over 100 years, with 500 replicates. Carrying capacity was set at three times the initial female abundance ($3N_i$) to coincide with the widely accepted belief that boreal caribou populations occur at densities typically well below the carrying capacity of their habitat, likely because predation limits many North American caribou populations to levels below the density that food availability could sustain (Seip 1991, Bergerud 1996). BWCSim1.0 incorporates demographic stochasticity by using a random number generator to ascribe annual values for vital rates within the range of variation around mean values reported in the literature, thus simulating the variation in vital rates among individuals. Environmental stochasticity is simulated through the model replicates (e.g., generation of multiple Leslie matrices), which incorporate a range of survival and fecundity estimates derived from variation in vital rates.

BWCSim1.0 models the population demographics of single populations, whereby no immigration or emigration occurs between populations. Environmental catastrophes were not included in the model and there was neither maximum age nor maximum breeding age. To buffer against overly optimistic estimates of population persistence due to limitations of the model, we report quasi-extinction risk (risk of population dropping below 10 females) for critical population size assessment. For all other analysis, we reported predictions of extinction risk. The IUCN criterion for classifying species as Vulnerable (equivalent to COSEWIC's Threatened category) is a risk of extinction $\geq 10\%$ over 100 yrs (SSC 2001). We therefore set the threshold of acceptable risk of extinction at $< 10\%$ over 100 yrs.



Appendix 6.6 - Table 1. Mean, minimum, and maximum population parameter values for boreal caribou across Canada.

Jurisdiction	Year	% Adult Males	% Yearlings	% Calves	Adult Female Survival (S_{ad})	CV of Adult Female Survival ($S_{ad}CV$) ¹	Calf Survival (S_{calf})	CV Calf Survival ($S_{calf} CV$)	Parturition	Parturition CV	Study
QC	1999-2001	43.1%			0.75	0.11					Courtois et al. 2007
QC	1999-2001	37.3%			0.87	0.06					Courtois et al. 2007
QC	1999-2001	29.8%			0.82	0.07					Courtois et al. 2007
AB	1995	45.9%		9.0%	0.81						Stuart-Smith et al. 1997
AB	1976-78	46.0%		13.0%	0.85		0.25				Fuller and Keith 1981
NFLD	1995-97				0.88	0.09	0.46	0.46	1		Mahoney and Virgl 2003
NFLD	1994-97	43.2%									Mahoney and Virgl 2003
Sask	1993-96				0.80	0.12					Rettie and Messier 1998
Sask	1993-96				0.87	0.10					Rettie and Messier 1998
Sask	1993-96				0.79	0.13					Rettie and Messier 1998
Sask	1993-96				0.78	0.13					Rettie and Messier 1998
BC	2003-04				0.94	0.03			0.78		Culling et al.
AB	1993-2002				0.89	0.01					McLoughlin et al. 2003
AB	1993-2002				0.86	0.01					McLoughlin et al. 2003
AB	1995-2002				0.87	0.04					McLoughlin et al. 2003
AB	1995-2002				0.89	0.03					McLoughlin et al. 2003
AB	1998-2002				0.93	0.01					McLoughlin et al. 2003
AB	1998-2002				0.86	0.02					McLoughlin et al. 2003
Lab	1981-1988	38.9%		18.5%	0.80	0.07	0.38	0.12	0.74	0.10	Schaefer et al. 1999
Lab	1993-1997	28.6%		8.9%	0.70	0.07	0.17		0.71	0.09	Schaefer et al. 1999
ON	1976-1984	52.0%	15.7%	22.0%			0.67		0.81		Ferguson et al. 1998
AB	1999-2003	26.0%		10.9%	0.85	0.04	0.23				Smith 2004

¹ Coefficient of Variation



Appendix 6.6 - Table 1. Mean, minimum, and maximum population parameter values for boreal caribou across Canada.

Jurisdiction	Year	% Adult Males	% Yearlings	% Calves	Adult Female Survival (S_{ad})	CV of Adult Female Survival ($S_{ad}CV$) ¹	Calf Survival (S_{calf})	CV Calf Survival ($S_{calf} CV$)	Parturition	Parturition CV	Study
AB	1979-1984				0.75	0.14					Edmonds 1988
AB	1993-2001										Sorensen et al. 2008
AB	1993-2001										Sorensen et al. 2008
AB	1993-2001										Sorensen et al. 2008
AB	1993-2001										Sorensen et al. 2008
AB	1993-2001										Sorensen et al. 2008
AB	1993-2001										Sorensen et al. 2008
ON	2005										Vors 2006
ON	2005										Vors 2006
QC	1999				0.73	0.22					Courtois et al. 2005
QC	2000				0.82	0.14					Courtois et al. 2005
QC	2001				0.85	0.12					Courtois et al. 2005
QC	2002				0.79	0.15					Courtois et al. 2005
QC	2003				23.1%	0.06					Courtois et al. 2005
QC	2004				26.7%	0.87					Courtois et al. 2005
QC	2005				18.2%	0.93					Courtois et al. 2005
NFLD	1957-1967										Bergnerud 1971
NFLD	1957-1967										Bergnerud 1971
N. America		36.0%									Bergnerud 1971
	Min	26%	10%	9%	0.70	0%	0.17	12%	0.71	9%	
	Mean	39%	14%	16%	0.85	8%	0.38	38%	0.76	10%	
	Max	52%	16%	27%	0.94	22%	0.67	64%	0.81	10%	

¹ Coefficient of Variation



Appendix 6.6 - Table 2. Model parameters used for the non-spatial boreal caribou PVA

Parameter	Value/Range	Source
Stage classes	2 (Adult female, Yearling female, Calf female)	
Carrying Capacity	3 times initial female abundance (3N _i)	2,3,5,6,10,11,13,14
% Calf in population	16%	2,5
% Yearling in population	14%	2,3,5,6,7,10,11,13
% females among adults	61%	
% females among calves	50%	
% females among yearlings	50%	
Parturition rate	0.76	5,10,15,16
Recruitment (of female calves)	parturition - sex ratio - calf survival	
Yearling female fecundity	0	
CV Yearling Female Fecundity	0	
Adult female survival	0.70, 0.85, 0.94	1,3,4,5,6,8,9,10,11,12,13
CV adult female survival	1%, 8%, 22%	1,3,4,5,6,8,9,10,11,12,13
Yearling female survival	0.70, 0.85, 0.94	1,3,4,5,6,8,9,10,11,12,13
CV yearling female survival	1%, 8%, 22%	1,3,4,5,6,8,9,10,11,12,13
Calf Survival	0.17, 0.38, 0.67	1,3,4,5,6,8,9,10,11,12,13
CV calf survival	12%, 38%, 64%	1,3,4,5,6,8,9,10,11,12,13
1 Edmonds 1988; 2 Bergerud 1971; 3 Courtois et al. 2007; 4 Courtois et al. 2005; 5 Ferguson et al. 1988; 6 Fuller and Keith 1981; 7 Gustine et al. 2006; 8 Mahoney and Virgl 2003; 9 McLoughlin et al. 2003; 10 Schaefer et al. 1999; 11 Smith 2004; 12 Sorenson et al. (2008); 13 Stuart Smith et al. 1997; 14 Vors 2006; 15 Rettie and Messier 1998; 16 Culling et al. (no date)		



Critical Population Size Assessment

We modelled a combination of calf survival (S_{calf}) and adult female survival (S_{ad}) rates to assess the population size required to reduce the probability of quasi-extinction to <0.10 over 100 years. The values we used for low (L), medium (M), and high (H) survival and CV for calves and adult females, which were compiled from the mean and minimum and maximum of mean published values (Table 1). We assessed the following four combinations of vital rates:

- i) Low S_{calf} , high CV of S_{calf} , mean S_{ad} , and mean CV of S_{ad} (LHMM);
- ii) Mean S_{calf} , high CV of S_{calf} , mean S_{ad} and mean CV of S_{ad} (MHMM);
- iii) Mean S_{calf} ; High CV of S_{calf} ; Mean S_{ad} , High CV of S_{ad} (MHMH);
- iv) Low S_{calf} , high CV of S_{calf} , high S_{ad} and mean CV of S_{ad} (LHHM);
- v) 75th percentile of S_{calf} , CV of S_{calf} , S_{ad} , and CV of S_{ad} (75th percentile; Table 3).

We did not model a combination of high S_{calf} and low S_{ad} because we assumed this was unlikely to be observed in natural populations.

For each scenario, we increased initial female abundance until the risk of quasi-extinction was $<10\%$. The risk of quasi-extinction was calculated as the average number of years, over 500 replicates, for which abundance was equal to less than 10 female caribou over 100 yrs.).

Appendix 6.6 - Table 3. Scenario parameter values to assess population size thresholds of boreal caribou, based on calf and adult female survival (S) and variation (CV = coefficient of variation).

Scenario	Description of Scenario	Calf Survival (S_{calf})	CV Calf Survival S_{calf} CV	Adult Female Survival (S_{ad})	CV Adult Female Survival (S_{ad} CV)
LHMM	Low S_{calf} ; High CV of S_{calf} ; Mid S_{ad} , Mid CV of S_{ad}	0.17	64%	0.85	8%
LHHM	Low S_{calf} ; High CV of S_{calf} ; High S_{ad} , Mean CV of S_{ad}	0.17	64%	0.94	8%
MHMM	Mean S_{calf} ; High CV of S_{calf} ; Mid S_{ad} , mean CV of S_{ad}	0.38	64%	0.85	8%
MHMH	Mean S_{calf} ; High CV of S_{calf} ; Mean S_{ad} , High CV of S_{ad}	0.38	64%	0.85	22%
75 th Percentile	75 th P_ S_{calf} , 75 th P_ CV of S_{calf} ; 75 th P_ S_{ad} , 75 th P_ CV of S_{ad}	0.44	51%	0.88	15%



Population Trajectory Models

We modelled population trajectories using data from the only studies that reported both calf and adult female survival for four populations of boreal caribou (Table 1), including two study periods for a population in Labrador (for which vital rates differed substantially), for a total of five models (Table 4). We used mean survival rates and CVs of survival rates and population sizes reported in the studies. For the three studies that did not report variation in survival estimates, we used CVs compiled in Table 1 for the missing values. We assigned the Max CV (as reported in Table 1) to the missing S_{calf} CVs because because the corresponding S_{calf} rates for the missing values were below the overall mean of 0.38 and low survival estimates are associated with higher inter-annual variation and (Table 1). We used the mid-CV of 8% reported in Table 1 for the missing S_{ad} CV because the corresponding S_{ad} value was equal to the overall mean S_{ad} compiled in Table 1. All studies reported estimates of population size. We used 50% of these estimates as the initial female abundance to be modelled; given that we calculated female adults and yearlings represented ~50% of the total population. We used values reported in Table 2 for parturition, proportion of yearlings in population and calf sex ratio.

Appendix 6.6 - Table 4. Parameter estimates used to model populations of boreal caribou.

Study	Population	Population Size	N _i *	S _{ad}	S _{ad} CV	S _{calf}	S _{calf} CV
Fuller and Keith 1981	Birch Mountains, AB 1976 – 78	59	30	0.85	8%**	0.25	64%
Mahoney and Virgl 2003	Corner Brook Lakes, NF 1994 - 97	584	292	0.88	6%	0.45	17%
Schaefer et al. 1999	Red Wine Mountains, Labr. 1981 - 88	710	355	0.80	7%	0.38	12%
Schaefer et al. 1999	Red Wine Mountains, Labr. 1993 - 97	151	76	0.70	7%	0.17	64%
Smith 2004	Little Smokey, AB 1993 - 2003	80	40	0.85	4%	0.23	64%

* Initial female abundances (N_i) were set to 50% of population estimates reported in the studies.

** Data in italics denotes values assigned from range of mean values in Table 1.



Sensitivity Analysis

We conducted sensitivity analyses to determine the relative importance of adult female survival (S_{ad}), calf survival (S_{calf}), and their coefficients of variation (S_{ad} CV and S_{calf} CV) to risk of extinction, by modeling the range of mean values for each parameter that we compiled from the literature (Table 1). We varied one parameter at a time, while keeping the other parameters at mean values (Table 5). We then calculated the percent risk of extinction for each scenario as the average number of times the population reached 0 in 100 yrs over 500 replications. We ran models with three initial female abundances (N_i) at 100, 200 and 400 individuals to investigate the potential effect of population size on extinction risk.

Appendix 6.6 – Table 5. Scenario parameter values to assess the relative importance of population parameters to risk of extinction for boreal caribou.

Parameter Varied	S_{calf}	S_{calf} CV	S_{ad}	S_{ad} CV
S_{ad}	0.38	38%	0.70-0.94	8%
S_{calf}	0.17-0.67	38%	0.85	8%
S_{ad} CV	0.38	38%	0.85	1-22%
S_{calf} CV	0.38	12-64%	0.85	8%

Recruitment Analysis

We modelled the effect of recruitment on risk of extinction under a variety of female survival rates (0.80, 0.84, 0.88) and initial female abundances of 200, 400, 600, and 800 (corresponding to population sizes of 400, 800, 1200 and 1600 caribou; Table 5). We calculated corresponding calf survival rates from mean recruitment values taken from the National Meta-analysis of Boreal Caribou Demography and Range Disturbance (Table 6; see also Appendix 4.5). Given an assumed parturition rate of 0.76, calf survival was calculated as:

$$S_{calf} = (\text{mean recruitment} / 0.76) / 100$$



Appendix 6.6 - Table 6. Recruitment of boreal caribou and corresponding calf survival values.

Recruitment (calves/100 cows)	calves/cow	S_{calf}¹
7.15	0.072	0.09
12.30	0.123	0.16
12.60	0.126	0.17
13.40	0.134	0.18
13.60	0.136	0.18
13.90	0.139	0.18
15.25	0.153	0.20
16.38	0.164	0.22
17.40	0.174	0.23
20.71	0.207	0.27
20.90	0.209	0.28
21.30	0.213	0.28
27.35	0.274	0.36
28.00	0.280	0.37
28.05	0.281	0.37
28.94	0.289	0.38
32.28	0.323	0.42
40.33	0.403	0.53
40.58	0.406	0.53
45.37	0.454	0.60
45.40	0.454	0.60
45.40	0.454	0.60
50.25	0.503	0.66
50.54	0.505	0.67

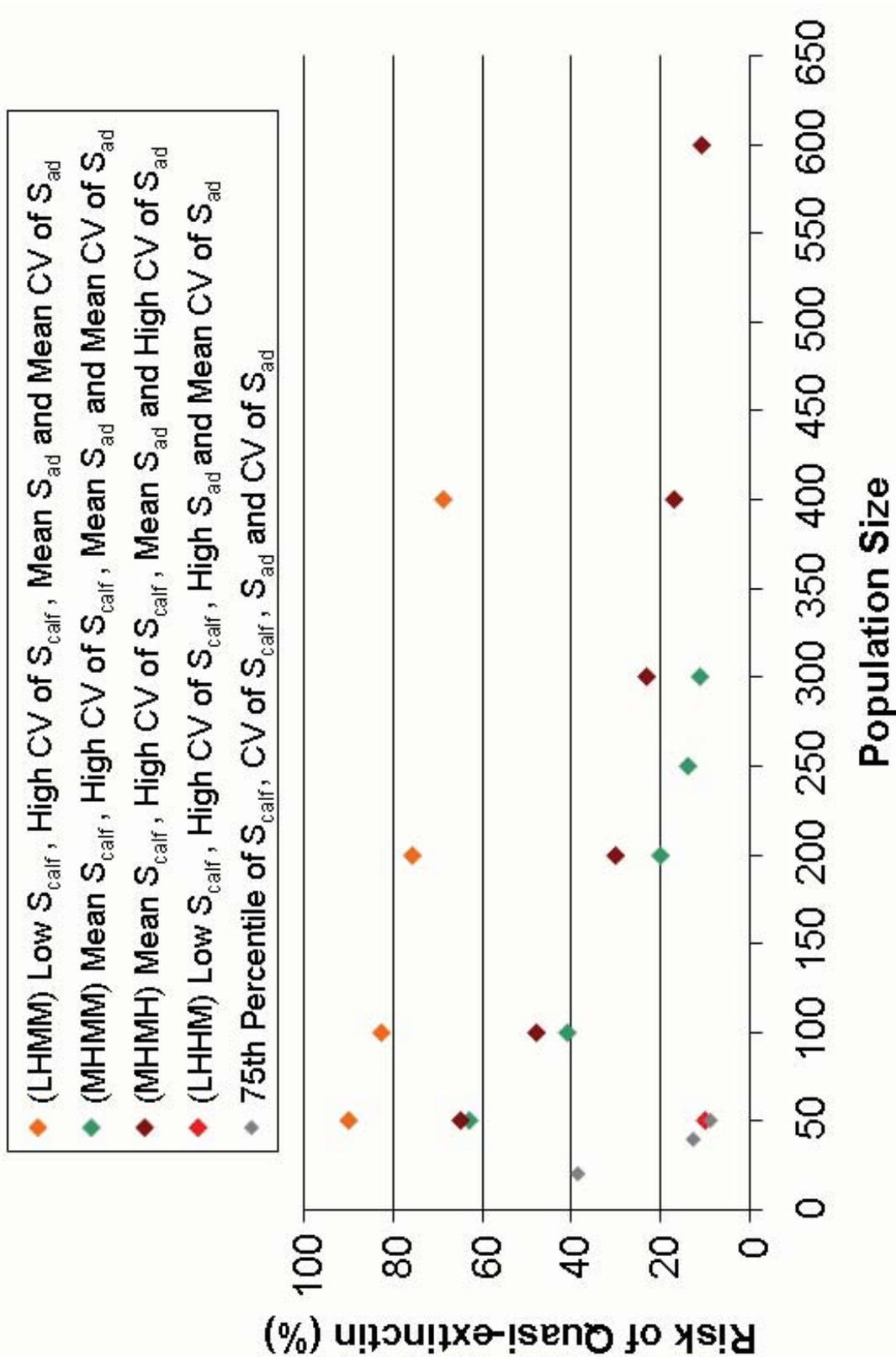
¹Calf survival calculated as $S_{calf} = \text{Recruitment} / \text{Parturition}$. Parturition rate assumed to be 0.76.



RESULTS

Critical Population Size Assessment

The results of the non-spatial PVA indicated that populations of boreal caribou with poor demographic conditions (e.g., low calf survival and moderate adult female survival) face a high risk of quasi-extinction regardless of population size (Figure 1; LHMM). Populations with medium calf survival (high CV) and medium adult female survival (mean CV) required a minimum of 300 individuals to reduce the risk of quasi-extinction to <10% (Figure 1; MHMM). Under the same mean survival rates but using high CVs for calf and adult female survival, a population size of 600 was required to offset the risk of quasi-extinction. Under conditions of low calf survival (high CV but high adult female survival and mean CV), however, a population of 50 animals had a quasi-extinction risk <10%, suggesting that high adult female survival compensated for low calf survival (Figure 1; LHHM). Under good demographic conditions (e.g., relatively high adult female and calf survival corresponding to 75th percentile of survival rates and CVs), a population size of 50 had a 10% chance of quasi extinction over 100 yrs (Figure 1; 75th Percentile).



Appendix 6.6 - Figure 1. The effect of population size on risk of quasi-extinction for models combining various rates of calf survival, CV of calf survival, adult female survival, and CV of adult female survival. Quasi-extinction is defined as the risk of the population decreasing below 10 females over 100 yrs. See Table 3 for a description of the models.

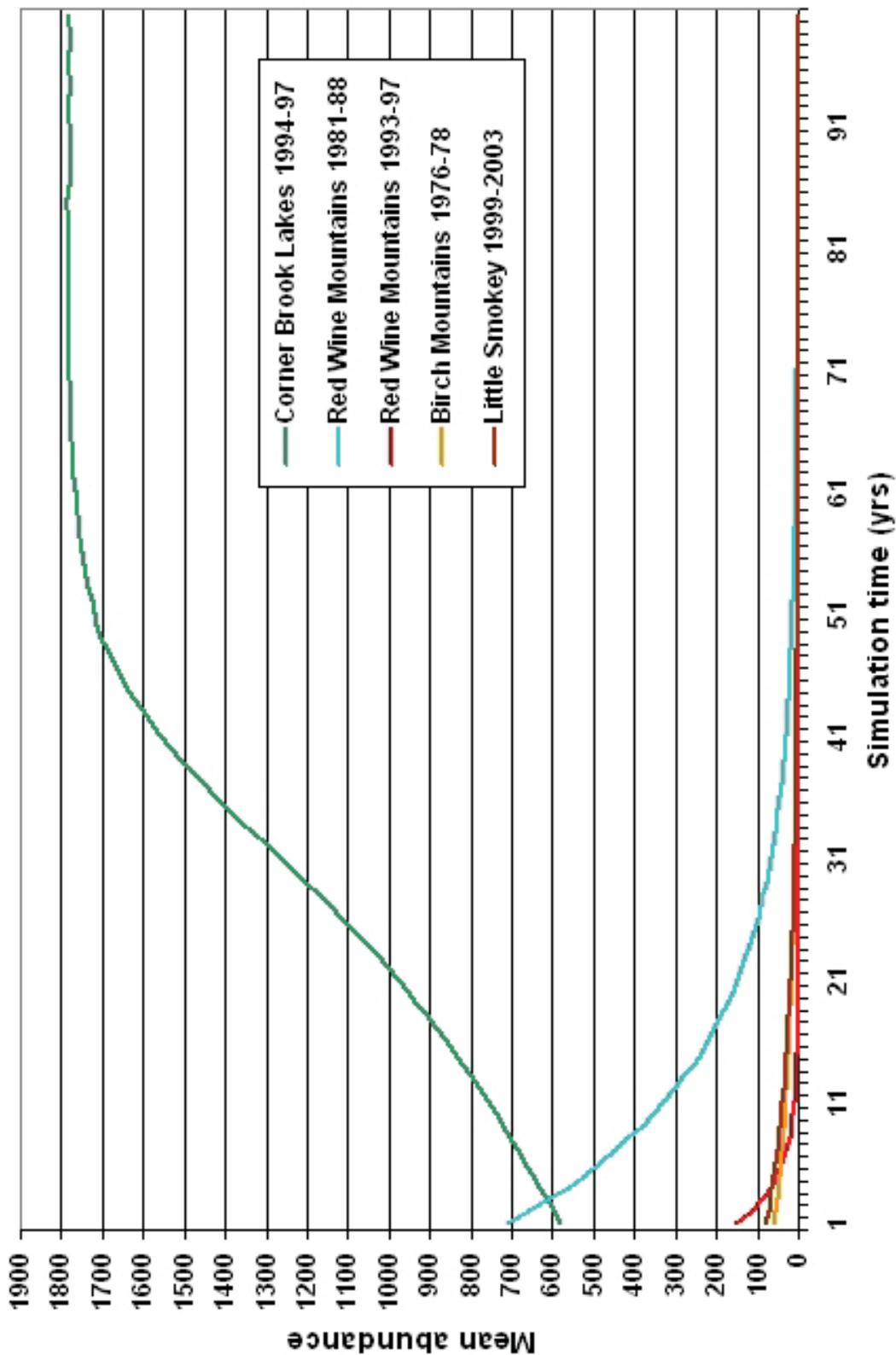


Population Trajectory Models

All populations, except the Corner Brook Lakes population in Newfoundland (Mahoney and Virgl 2003), went extinct within 100 yrs, although the time to extinction varied among studies (Figure 2). The three populations with the poorest demographic conditions (Red Wine Mountains late period, Birch Mountains, and Little Smokey) declined to the quasi-extinction threshold of 10 females within 20 years, while the Red Wine Mountains early population declined at a slower rate. The risk of extinction $P(e)$ and quasi-extinction $P(qe)$ for all populations except the Corner Brook Lakes population was $>10\%$ (Table 7).

Appendix 6.6 - Table 7: Probabilities of extinction, $P(e)$, and quasi-extinctions, $P(qe)$, over 100 yrs for four boreal caribou study populations

Study Population	$P(e)$	$P(qe)$
Birch Mountains 1976-78	0.52	0.82
Corner Brook Lakes 1994-97	0.00	0.00
Red Wine Mountains 1981-88	0.30	0.55
Red Wine Mountains 1993-97	0.83	0.93
Little Smokey 1999-2003	0.53	0.80



Appendix 6.6 - Figure 2. Predicted changes in the mean total abundance of four populations of boreal caribou over 100 years, based on population size, mean adult female survival, calf survival, and their variance (CVs) reported in a demographic study for each population. We used mean values from 15 boreal caribou population studies to estimate parturition, proportion of yearlings in population and calf sex ratio.



Sensitivity Analysis

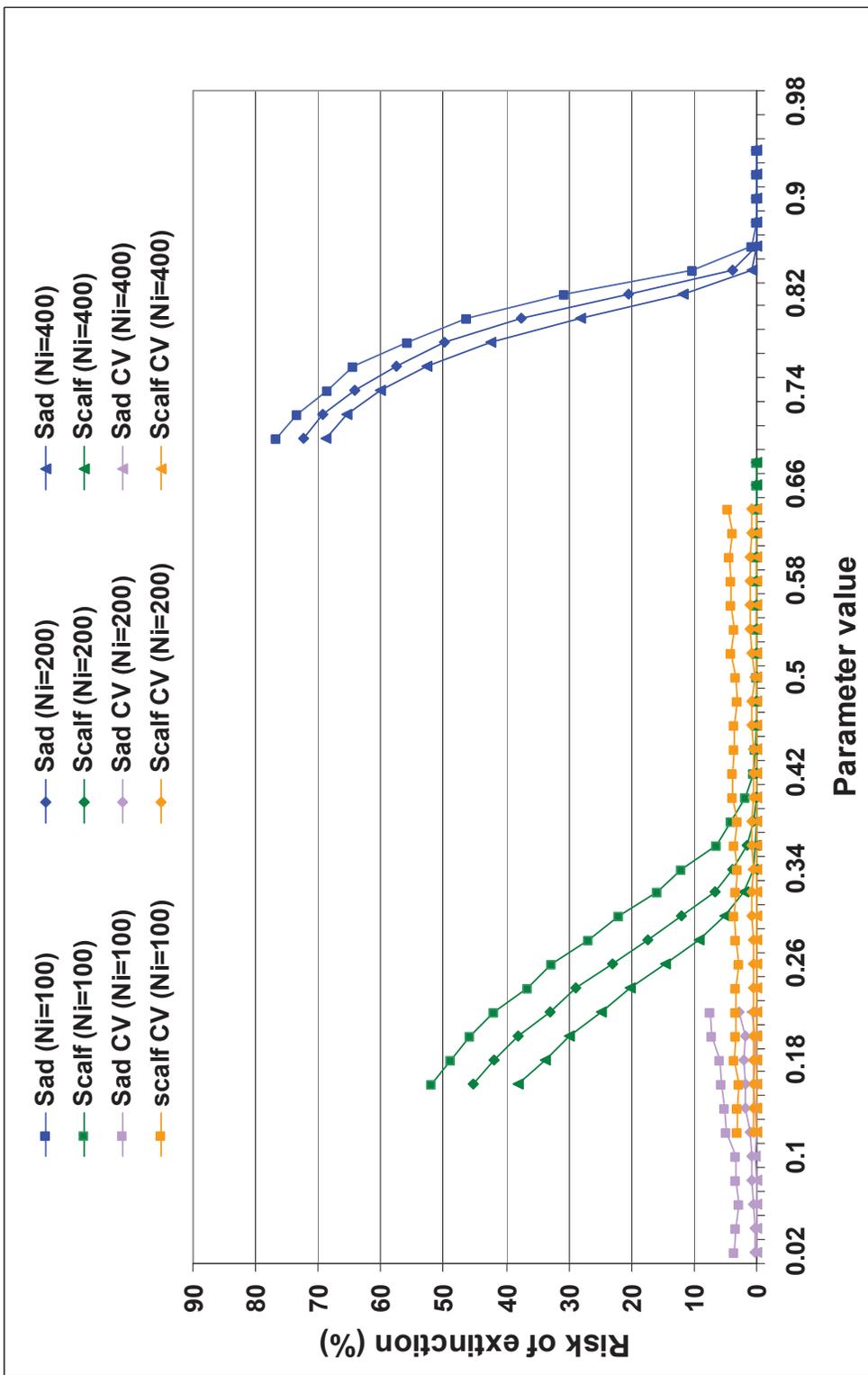
Of the vital rates that we tested, adult female survival and calf survival had the largest effect on probability of extinction (Figure 3). The CV of S_{ad} and the CV of S_{calf} had minor effects on probability of extinction, depending on the size of the population that was modelled (Figure 3). Relative to survival rates, population size in the range that we modelled (100 – 400 initial adult and yearling females) had little effect on the risk of extinction (Figure 3).

The cumulative percent change in risk of extinction was much greater with increasing adult female survival than it was with increasing calf survival (Figure 4). A change from mean to low S_{ad} increased the probability of extinction by 72%, while a change from mean to low S_{calf} increased the probability of extinction by 42%. In contrast, a change from mean to low CV of S_{calf} or S_{ad} did not change the probability of extinction more than 5%. Relative to survival rates, population size in the range that we modelled (100 – 400 initial adult and yearling females) had little effect on the risk of extinction (Figure 3).

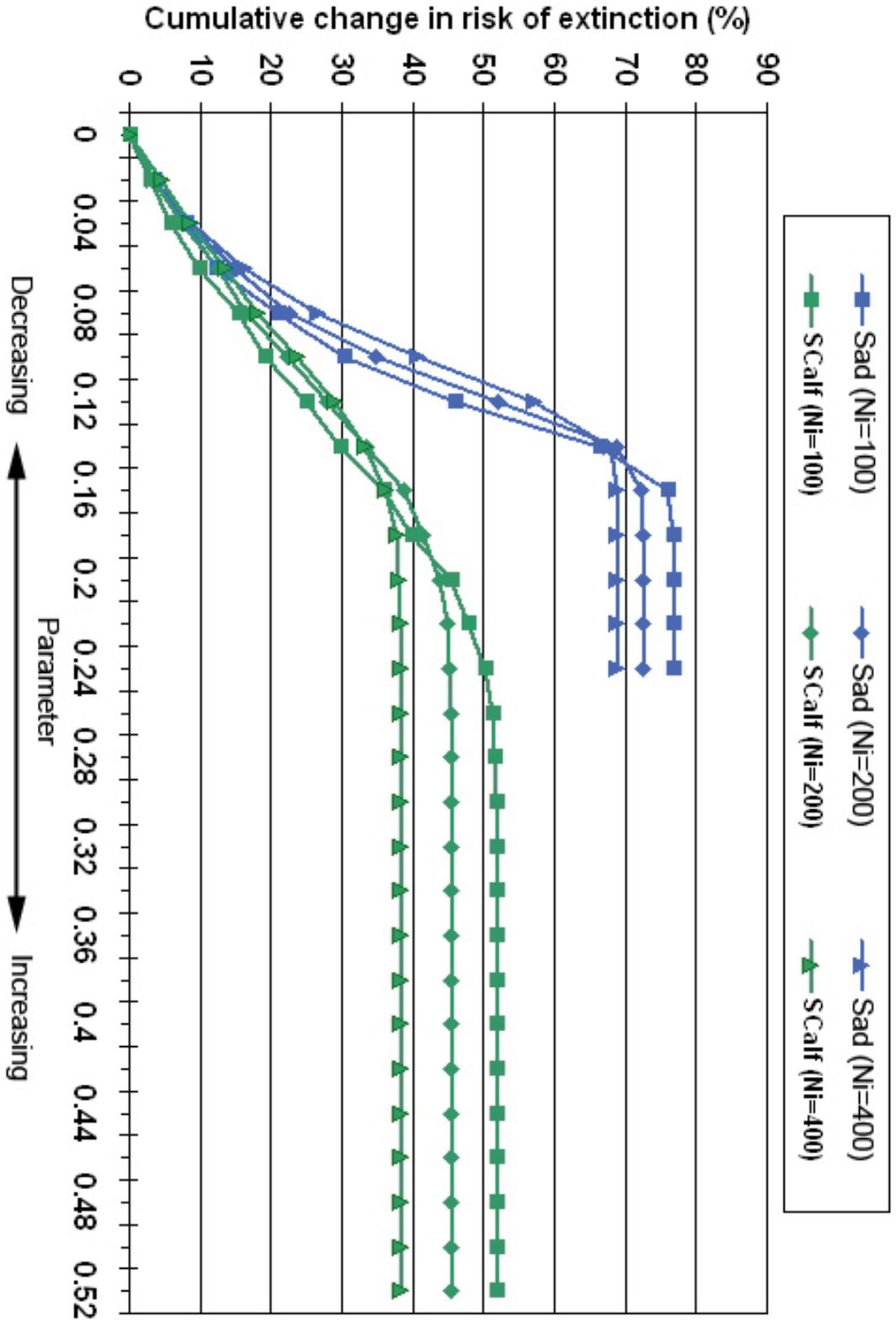
Although the range in modelled adult female survival was smaller (0.70 – 0.94) than the range in modelled calf survival (0.17 – 0.67), the cumulative change in risk of extinction was much higher for adult female survival (78%; Figure 4) than for calf survival (52%), suggesting the importance of adult female survival in boreal caribou population dynamics.

Recruitment

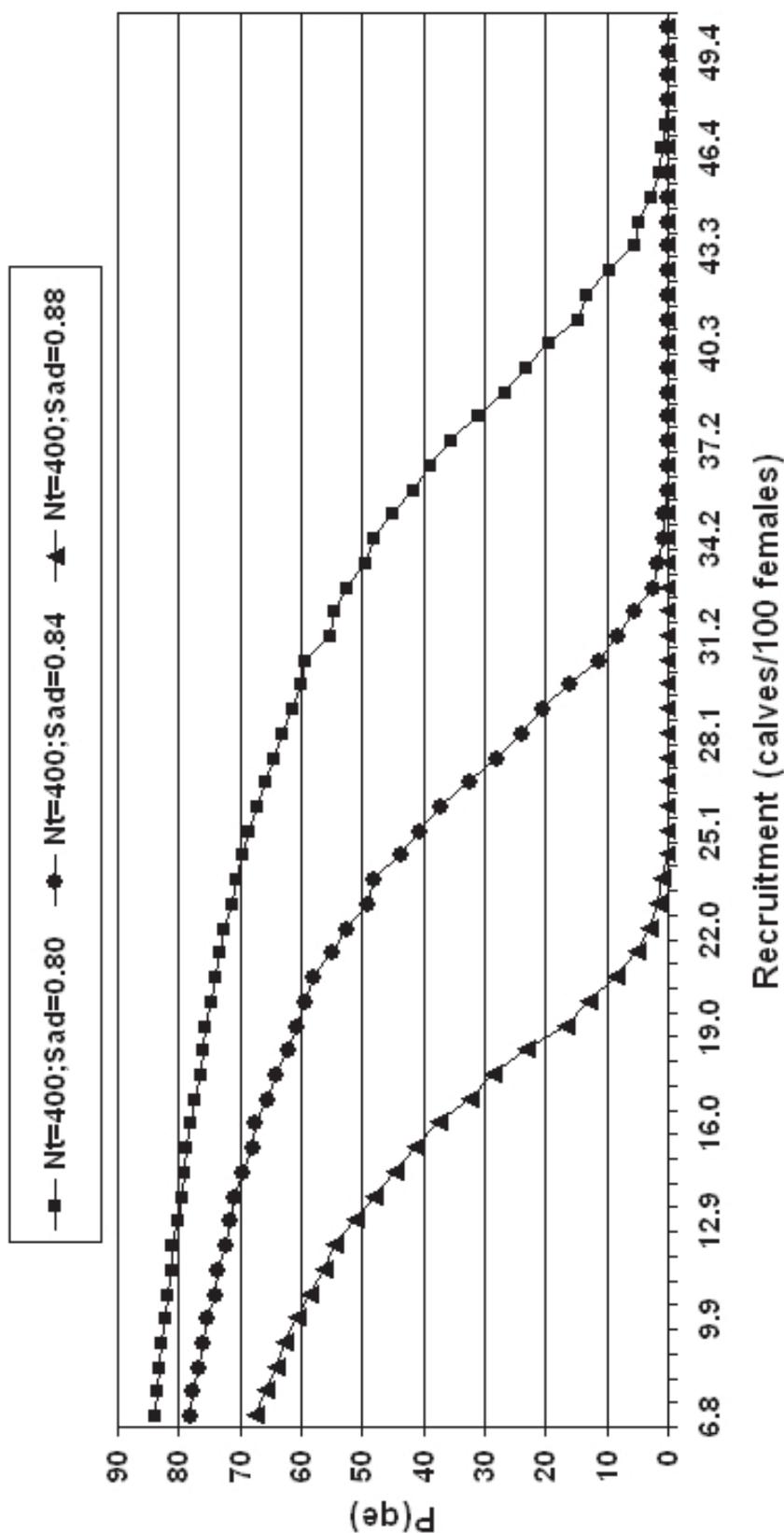
The probability of extinction decreased with increasing recruitment rates (Figure 5). Under conditions of relatively high adult female survival (0.88), populations of 400 individuals required a recruitment rate of 20 calves/100 cows to reduce the risk of quasi-extinction to <10% (Figure 1). Under lower adult female survival (0.80 – 0.84), populations of 400 individuals required a recruitment rate of 30 – 43 calves/100 cows to reduce the risk of quasi-extinction to <10% (Figure 5).



Appendix 6.6 - Figure 3. Change in risk of extinction (%) per unit increase in rates of the following parameters: adult female survival (S_{ad}) and female calf survival (S_{calf}), and unit increase of coefficients of variation (CV) of adult female survival (S_{ad} CV) and female calf survival (S_{calf} CV), under various population sizes (Ni; 100, 200, 400 females).



Appendix 6.6 - Figure 4. Cumulative change in risk of extinction due to change in adult females survival and calf survival for boreal caribou. Parameter represents adult female survival (S_{ad}) or female calf survival (S_{calf}).



Appendix 6.6 - Figure 5. Effect of boreal caribou recruitment (calves per 100 females) on the probability of quasi-extinction as determined from a non-spatial population viability analysis. Quasi-extinction risk was measured as the mean number of years (across 500 replicates) the population reaches 10 females over 100 years (based on an initial population size of 400 females).



Discussion

Our models suggested that populations of boreal caribou with poor demographic conditions (e.g., low calf survival and moderate adult female survival) face a high risk of quasi-extinction at any population size. Under moderate demographic conditions (mean calf survival and mean adult female survival), population size plays an important factor in reducing risk of quasi-extinction. Under good demographic conditions (e.g., relatively high calf and adult female survival or high adult female survival and mean calf survival), when other factors that may increase the risk of extinction are absent, small populations of 50 individuals could persist for long periods of time. Of the 57 local populations of boreal caribou in Canada that are considered to be threatened, 46% are small (less than 300 animals), 28% are considered to be declining, and 19% have both conditions. Our models indicated that small, declining boreal caribou populations are in immediate need of enhanced management to improve their chance of persistence.

Our results indicated that adult female survival strongly influences boreal caribou population trajectory and that high adult female survival can buffer the effects of poor calf recruitment. This conclusion is supported by field studies that have demonstrated the strong influence of adult female survival on ungulate demographics (Nelson and Peek 1984, Eberhardt 1985, Hern et al. 1990, Walsh et al. 1995, Crête et al. 1995, Arthur et al. 2003, Wittmer et al. 2005). Our results also demonstrated the influence of calf survival on boreal caribou population trajectory, similar to Bergerud (1971), who showed a strong correlation between calf survival and population growth. Raithel et al. (2007) found that, despite calf survival having relatively low elasticity, the variation in calf survival explained most of the variation in lambda in an elk population.

Our models indicated that, given demographic conditions reported in the literature for four populations of boreal caribou, three have a high risk of extinction. Under relatively poor demographic conditions (e.g., relatively low adult female and calf survival), no population size can eliminate the risk of extinction, although larger populations would take longer to become extinct. The population experiencing good demographic conditions, on the island of Newfoundland (Mahoney and Virgl 2003), exists in the absence of wolves, a predator whose functional and numerical response increases with habitat disturbance (Seip 1991). It is unrealistic to expect that vital rates of boreal caribou remain unchanged over 100 years. For example, adult female survival in the Red Wine Mountains population increased from an average of 0.70 during 1993 - 1997 to 0.90 during 2000 - 2005 (Unpublished data, Wildlife Division, Government of Newfoundland and Labrador). This population has therefore not met the prediction of extinction in the PVA as a result of increasing adult female survival. Nonetheless, our results illustrated that moderate to low adult and calf survival rates increase the risk of extinction and that populations with poor demographic conditions decline rapidly regardless of their population size. Positive change in vital rates, however, particularly of adult female survivorship, can significantly change the outcome of the PVA predictions. Thus, models need to be re-evaluated as new data and new knowledge become available.



Our results demonstrated that the probability of extinction in boreal caribou populations decreases with increasing recruitment rates. Bergerud (1992) reported that 27.7 calves/100 cows yielded a finite rate of population increase (λ) value of 1, based on 32 population survey years of both barren-ground and woodland caribou. Our results indicated that this threshold can vary, depending on survival of adult females.

In our model, density dependence is incorporated as a logistic Ricker equation (scramble competition), assuming a maximum population growth rate (λ) of $R_{max}=1.3$. Population growth is affected when abundance reaches the carrying capacity K ; below K vital rates of the stage matrix are unchanged (e.g., no density dependence). Although this suggests a ceiling form of density dependence, any form of density dependence below K would otherwise increase extinction risk and suggest an unrealistically high risk of extinction.

Linking carrying capacity with population size (e.g., $K=3N_i$) likely introduced some density dependent bias, especially for large populations, that may result in an overestimation of growth rates for large populations. The importance of the CV of survival increases as population abundance approaches K because a high CV causes greater fluctuations in abundance and thus causes the population to approach or overshoot K more rapidly, when density dependence effects occur.

The primary limitations of our model were that no maximum age or maximum age of breeding were incorporated. These limitations resulted in optimistic projections of extinction risk and likely over-emphasized the importance of adult female survival to risk of extinction and under-estimated the critical population size. The addition of a multi-age matrix model with a maximum age and senescence components would address these issues and produce more realistic estimates of extinction risk relative to population size.

Future modeling efforts should investigate the relationship between the age structure of the initial population on population size and trend over time. Insight into the degree to which a population skewed toward females is able to moderate a decline due to the greater proportion of reproducing individuals and how the proportion of yearlings to adults can influence trends would help inform conservation management of boreal caribou. An investigation of the correlation between adult and calf survival would help elucidate the relative importance of these factors and inform the development of management strategies that affect these vital rates.

The recovery strategy for boreal caribou states the “need for large areas of boreal forest with adequate amounts of suitable habitat and low predation rates is a consistent requirement for the conservation of the boreal population of woodland caribou across Canada” (Environment Canada 2007). Given that population vital rates are affected by habitat alteration that favours alternate prey and their predators, the non-spatial PVA provided insight into the effects of a range of demographic conditions on population persistence and the recovery goal of self-sustaining boreal caribou populations.



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6.7 Spatial Population Viability Analysis Case Study

Carlos Carroll, Ph.D.

Introduction

The overarching goal of the national recovery strategy for boreal caribou is to conserve and recover boreal caribou populations and their habitat; that is, prevent extirpation of local populations and maintain or enhance habitat condition to allow these populations to be self-sustaining (EC 2007). Thus the link between population viability and habitat amount and condition is an explicit part of the recovery goal. The question of “How much and what configuration of **habitat** is enough to achieve the goal of self-sustaining (viable) **populations?**” links the process of delineation of critical habitat designation with an analytical approach or suite of methods known as population viability analysis (PVA).

Population viability analysis often involves the use of analytical models to provide quantitative estimates of extinction times and probabilities. Most recent review papers on PVA have judged these metrics less than robust to model and data uncertainty (McCarthy et al. 2003). This type of PVA has also been criticized for limited relevance to real-world conservation planning contexts, due to its emphasis on “small population paradigm” factors (e.g., inbreeding depression) rather than more pressing “declining population paradigm” factors (e.g., habitat loss) (Caughley 1994). Here we use a broader definition of PVA that includes a range of methodologies to integrate existing knowledge and models of varying complexity in a structured way. The most valuable output of such PVA is often a better understanding of how trends in species distribution at larger spatial and longer temporal scales are linked to landscape change (development) trends, in a way that is difficult to assess without some form of modeling. This allows PVA to be used as a tool to rank alternative management scenarios rather than assign absolute persistence probabilities. There are significant challenges to application of such PVA modeling to boreal caribou, such as the species’ relatively complex local population dynamics. A variety of specific analytic methods can be used, with the most appropriate method for boreal caribou depending on factors such as the spatial scale of the question and the nature of available input data. The critical habitat science review has pursued four complementary analytical approaches: environmental niche analysis, range-wide meta-analysis of demography-habitat relationships, non-spatial (heuristic) PVA, and spatial PVA. Here we review initial results from the spatial PVA. The timeline of the critical habitat science review did not allow for completion of a full PVA study. These results instead serve as a proof of concept to assess the relevance of spatial PVA to the boreal caribou critical habitat analysis and recovery process. Although spatial PVA modeling methods are more complex, time-consuming, and require greater levels of input data than other methods, their potential to inform critical habitat designation and planning may justify their use as a complement to other, less data-intensive decision support tools. The major questions addressed in this report include:



- Adequacy of spatial (habitat) data – What type and quality of spatial data are required for a PVA?
- Adequacy of demographic data – Is possible to estimate demographic rates in different habitats with a level of accuracy sufficient for a PVA?
- Relevance of results – Do results from a spatial PVA inform recovery planning in ways not possible with other methods?
- Integration with other tools – How are results from a spatial PVA best integrated in a decision support context with results from the other analysis methods used in the critical habitat science review (environmental niche analysis, meta-analysis, and heuristic PVA)?

Motivation

The boreal caribou critical habitat science review participants chose spatial PVA as one of four methodologies to evaluate during the science review process. In common with the environmental niche analysis, the PVA incorporates spatial data. Spatial models are essential supports to critical habitat analysis in that they provide a broad-scale summary of landscape condition. In contrast to the environmental niche analysis, which addresses habitat primarily at the broadest spatial scale, the spatial PVA focuses on aspects of habitat such as forest type and distance from roads that act at an intermediate spatial scale corresponding to the extent of the local population.

Given this scale of analysis, several approaches of varying levels of complexity could be implemented. The same habitat data used as input to the spatial PVA could also be appropriately used to develop a “static” habitat model (e.g., habitat suitability index (HSI) or resource selection function (RSF)). However, even if such static models are used in place of a dynamic population model, a PVA-type process may be useful to help structure range-wide meta-analysis of habitat data and consideration of how the habitat relationships translate up spatial scales from habitat patches to landscapes and from short-term temporal fluctuations to long-term trends and persistence thresholds.

The model used here (“HexSim”; (Schumaker et al. 2004; Schumaker in prep.)) is a spatially-explicit population model (SEPM; also termed an individual-based model) in which habitat quality affects individuals that are followed as they age, give birth, disperse, and die over time. Individuals may hold exclusive territories or live in social groups. To justify its additional complexity, a spatial PVA must provide insights not possible with a static habitat model. One benefit of a SEPM is that it can help incorporate landscape processes into conservation planning and thus facilitate evaluation of the effects of alternate future scenarios. Planners must consider multiple future landscape scenarios due to uncertainties as to the effects of climate change, inherent uncertainty in ecosystem processes such as fire, and alternate options for management processes that transform habitat.

Previous research applying SEPMs to threatened species recovery planning found that the models gave insights beyond those provided by static habitat models because they could assess area and connectivity effects (e.g., inter-population dynamics and source-sink



dynamics) that strongly affected persistence of the species considered (Carroll et al. 2006). This may also be the case for boreal caribou. Alternatively, a caribou SEPM could provide similar conclusions to a simpler model such as an HSI and thus the simpler model would be preferred. Or a caribou SEPM could potentially offer new insights but require spatial data or demographic parameters that are largely unavailable. Each of these three outcomes is likely true in different regions, and a case study such as the one described here can help planners assess when and where SEPM are an appropriate decision support tool. Even if the data in a particular portion of caribou range are inadequate for deriving SEPM-based predictions regarding quantitative persistence thresholds, SEPM may still be useful in a heuristic sense in offering insights as to emergent processes and effects of landscape condition and structure on caribou persistence.

Caribou SEPM can be expected to be more complex than those for species such as the spotted owl, where individuals defend exclusive territories. Because boreal caribou occur in social groups, local population dynamics should be added to the SEPM. Movement between seasonal habitats should be added to the model for local populations where this occurs. In addition, multi-species SEPM that can capture the interaction between predators and caribou, and indirectly with alternate prey species such as moose, should be possible and may reveal important insights. However, it is important to keep in mind a key guideline: what is the simplest model that effectively supports conservation planning, and what real-world complexities can be ignored in the model without qualitatively compromising results in terms of the questions at hand?

The spatial scale of the case studies presented here was determined somewhat opportunistically by the extent of the available spatial habitat data. Ideally, as was the case here, the spatial data used would encompass the larger landscape, rather than only areas currently occupied by caribou. This extent allows addressing questions such as “How does habitat condition in the larger landscape support or not support caribou occurrence?” But unlike methods that assess summary statistics on aggregate habitat amount within a local population’s range (e.g., the proportion of the landscape within a set buffer distance from roads), a SEPM also focuses on finer-scale habitat pattern and composition. At this scale, the model addresses “How does the arrangement of habitat patches within the extent of a local population influence its persistence and demography?”, e.g., by influencing within-range movement and consequent exposure to predation.

Relationship with other components of science review

The four components of the critical habitat science review form a spatial and analytical hierarchy of methods. Their output shows less generality and more complexity (or “biological realism”) as one descends the hierarchy. Environmental niche analysis and range-wide meta-analysis can be seen as top-level methods, followed by the heuristic PVA, and finally the spatial PVA. Results from top-level analyses reveal overarching constraints on processes examined at lower levels. This perspective allows a synthesis of the four components. Lower-level results suggest factors missing from the top-level analyses, and in turn the top-level



analyses suggest the extent to which conclusions from e.g., the spatial PVA results may lack generality to some portions of range.

Environmental niche analysis (ENA) characterizes the distribution of boreal caribou by examining which abiotic factors (climate and topography) characterize the distribution of observed locations. These models may be especially relevant in predicting potential effects of climate change. In a second stage of ENA, broad-scale biotic variables (land cover and human impact levels) are added to further refine the models. However, these variables, because they are the lowest common denominator of detail available range-wide, lack the fine-scale habitat data possible in the spatial PVA. The second range-wide approach is a meta-analysis of relationships between demography and habitat. Both of these approaches, in contrast to the spatial PVA, can produce broadly general conclusions as to what abiotic and biotic conditions permit boreal caribou occurrence and persistence. However, neither approaches are mechanistic, in that they do not address the biotic mechanisms by which e.g., climate limits distribution. The heuristic PVA, in contrast, uses non-spatial models to assess how population persistence is affected by aspects of boreal caribou life history and population structure (e.g., age structure, age-specific survival and fecundity, environmental stochasticity, breeding structure, and density dependence). Because such a non-spatial PVA has far fewer parameters and computational demands than a SEPM, the heuristic PVA can more exhaustively explore the plausible parameter space for population dynamics and assess sensitivity of model results to chosen parameters. The spatial PVA explores only a subset of this parameter space but adds consideration of landscape structure and individual movement.

The spatial PVA is linked to the meta-analysis component, in that results of the meta-analysis can be used to inform, and to some extent validate, PVA results. The PVA can help in interpreting results of the meta-analysis in that the PVA may offer heuristic insights as to the mechanisms by which the ability of an area to support caribou scales up spatially from the patch to landscape. Additionally, spatial PVA tools allow simulation of longer-term trends and scenarios to extrapolate the relationships drawn from the meta-analysis to future landscapes.

Comparison of the heuristic and spatial PVA results helps assess 1) to what degree the spatial PVA model's behaviour is an artefact of particular assumptions as to parameters, 2) whether spatial effects produce qualitatively different results in terms of predictions of population persistence. An integrated assessment using the four approaches might begin with general conclusions as to what climatic conditions and broad-scale habitat characteristics are associated with boreal caribou occurrence (ENA) and persistence (meta-analysis), and refine these conclusions by assessment of what life history characteristics (heuristic PVA) and spatial population dynamics (minimum area requirements or dispersal limitation) may explain these patterns and further limit distribution and persistence.



Methods

Spatially-explicit population models (SEPM), like static HSI models, use input data on habitat factors that affect survival and fecundity of the species of concern. But SEPM then integrate additional information on characteristics such as demographic rates and dispersal behaviour. For example, social carnivores often require larger territories than solitary species of similar size, and may thus be more vulnerable to landscape fragmentation in a SEPM (Carroll et al. 2006). Unlike a simpler HSI model, a SEPM can provide insights on the effects of population size and connectivity on viability and identifying the locations of population sources and sinks.

HexSim, the SEPM used here, links the survival and fecundity of individual animals or groups to GIS data on mortality risk and habitat productivity (Schumaker et al. 2004, Schumaker in prep.). Individual territories or group ranges are allocated by intersecting the GIS data with an array of hexagonal cells. The different habitat types in the GIS maps are assigned weights based on the relative levels of fecundity and survival expected in those habitat classes. Base survival and reproductive rates, derived from published field studies, are then supplied to the model as a population projection matrix. The model scales these base matrix values based on the mean of the habitat weights within each hexagon, with lower means translating into lower survival rates or reproductive output. Each individual in the population is tracked through a yearly cycle of survival, fecundity, and dispersal events. Environmental stochasticity can be incorporated by drawing each year's base population matrix from a randomized set of matrices whose elements were drawn from a beta (survival) or normal (fecundity) distribution. Adult organisms are classified as either territorial or floaters. Floaters must always search for available breeding sites or existing groups to join. Movement decisions can be parameterized in a variety of ways, with varying proportions of randomness, correlation (tendency to continue in the direction of the last step), and attraction to higher quality habitat (Schumaker et al. 2004). Because it is difficult to parameterize movement rules directly from field data (but see Fryxell and Shuter 2008), it is important to assess the sensitivity of model results to a range of plausible movement parameters.

SEPM can produce a wide range of output in the form of both spatial data (maps) and summary statistics (e.g., population time series). This output can be used to assess an area in terms of the probability of occurrence of the species (similar to the output of a HSI model), the area's demographic role (source or sink) as well as give population-level predictions of long-term persistence or extirpation.

Because absolute estimates of risk from a SEPM are suspect due to uncertainty in data and models, SEPM output should instead be used to rank candidate recovery strategies in terms of viability (or extinction risk) and distribution (range expansion or contraction).



Spatial Data

Two case study areas were selected opportunistically for the SEPM analysis based on data availability. The first study area is located in northeastern Alberta on lands with forest tenure held by Alberta Pacific Forest Industries (ALPAC). This area encompasses the extent of the ESAR (eastside of Athabasca River) and WSAR (westside of Athabasca River) caribou herds (local populations). The area is predominantly a mixture of peatland and upland habitats with the predominant resource industries being timber extraction and oil and gas development. The second case study area is located in southeastern Manitoba, and encompasses the extent of the Owl Lake herd. The predominant resource industry in this area is timber extraction. Data for this study area was provided by the Eastern Manitoba Woodland Caribou Advisory Committee (EMWCAC). While the two case study areas obviously do not represent the full spectrum of landscape contexts encountered across the range of boreal caribou, they do show contrasts in habitat use and type of threats to population persistence. For example, a large expansion of linear features related to the energy sector is ongoing in the Alberta study area. The Manitoba case study allows examination of effects of timber harvest scenarios (as well as lower rates of expansion of linear features) on population persistence. Use of two contrasting case study areas allows more general assessment of what minimum level of habitat data (vegetation and linear features) is required for SEPM analysis.

In Alberta, data from the Alberta Vegetation Inventory (AVI) was classified into high, medium, and low quality caribou habitat. High quality habitat was defined as pure stands of black spruce, pure stands of larch, and mixed stands of black spruce and larch. Medium quality habitat was defined as black spruce and larch dominated-stands mixed with tree species other than larch and black spruce.

Low quality habitat was defined as all remaining areas. A second habitat layer was created from data on linear features. Areas within 250m of a roads or seismic lines were considered reduced in habitat suitability based on previous research (Dyer et al. 1999). The spatial data for the Manitoba study area was received later than the Alberta data and time constraints permitted only initial evaluation of its suitability for SEPM modeling. It is anticipated that spatial data predicting summer and winter habitat suitability (HSI model) will be the key input to the SEPM. Data on linear features (roads and transmission) lines are also available and may be buffered as in the Alberta case study.

Parameters

Survival rates were parameterized for the Alberta study area based on an expert workshop held with a subset of the Science Advisory Group (SAG) in Vancouver, BC, February 11-12, 2008. Rates were set to vary by habitat type and age class. Survival rates in high and medium quality habitat varied based on the proportion, averaged over a 10 km² moving window, of the area within 250m of linear disturbance. The equation for adult annual survival rate [S_a] in high and medium quality habitat was $S_a = 0.98 - (\text{proportion within buffer} * 23)$ (Figure 1). The equation for calf annual survival rate [S_c] in high and medium quality habitat was:



$S_c = 0.50$ – (proportion within buffer * 40). Adult annual survival rate in poor quality (upland) habitat was set to 0.65 irrespective of proportion of linear disturbance buffer. Calf annual survival rate in poor quality (upland) habitat was set to zero irrespective of proportion of linear disturbance buffer. Fecundity rate was set constant across habitats as 0.5 female offspring/female/year. A range of values for the parameter for maximum movement distance have been assessed. The base value used in the simulations shown here is 112 km (total path length, not total net displacement). All of the parameters used above would be subject to further review, revision, and sensitivity analysis in the course of a complete PVA study in order to produce a credible decision support tool.

Results

This initial report focuses on qualitative patterns in the results because it is expected that quantitative predictions would change as initial exploratory simulations are subject to review and sensitivity analysis in a complete PVA study. In the initial simulations, areas of high predicted occupancy are relatively widespread across the Alberta study area when linear disturbance effects are not considered (Figure 2a). This may be conceptualized as representing a landscape state closer to historic (pre-development) condition. These areas are much reduced in extent under the simulations where survival rates are affected by linear disturbance buffer zones (Figure 2b). This may be conceptualized as assessing the current landscape condition. The ESAR herd is affected more heavily by linear disturbance than is the WSAR herd. According to our data, 63.0% of the ESAR range is within 250 meters of linear disturbance, versus 44.93% of the WSAR range. A comparison between the HexSim simulations with and without linear disturbance shows a decline in occupancy probability of 76.7% for the ESAR herd, versus 58.7% for the WSAR herd. Although neither local population has a high likelihood of extirpation (given no further habitat loss) in these initial simulations, more realistic assessment of persistence probabilities should await simulations that better incorporate group dynamics.

Occupancy rates shown above are drawn from the final decade of 200 year simulations, averaged over 10 simulation runs. Although the simulations are 200 years in length, the landscape does not change in the current analysis. Therefore, predictions show the equilibrium “carrying capacity” of the current landscape, not the future persistence probabilities of the population given landscape change. Both stochastic landscape change, such as driven by fire, and deterministic habitat trends, such as increases in linear disturbance, would alter current equilibrium carrying capacity. These aspects could be explored in future simulations.

Despite a static landscape, population levels show wide variation around carrying capacity. A plot of five population time series drawn from the Alberta simulations with linear disturbance (Figure 2b) is shown in Figure 3. Relatively large population fluctuations (~20%) over periods of several decades are evident although the longer-term trend is stable. These fluctuations are driven by both demographic stochasticity and habitat pattern. The potential of caribou life history structure and demographic stochasticity in relative small populations to cause long-term fluctuations should be evident in a non-spatial (heuristic) PVA model. However, a spatial



model such as a SEPM allows habitat fragmentation and dispersal limitation to accentuate small population effects and increase the magnitude of fluctuations. The larger population inhabiting the “historic” landscape (Figure 2a) shows fluctuations of smaller magnitude due to both larger population size and lower levels of landscape fragmentation. The model output emphasizes that it is inherently challenging to interpret data from population monitoring programs for long-lived vertebrates, and SEPM simulations could be instructive in designing monitoring programs for more intact landscapes. However, deterministic habitat changes in the Alberta study area over the short-term will likely swamp the effects of demographic stochasticity.

Although HexSim simulations for the Manitoba study area were not possible within the timeframe of this study, the input habitat layers appear suitable for use in HexSim simulations. Figure 4 shows predictions from the EMWCAC HSI model (averaged over 100 km² moving window) for the Manitoba study area for a) caribou summer habitat, and b) winter habitat, overlaid with linear features. Although HexSim does allow habitat value to change seasonally, there is relatively low contrast between winter and summer HSI values (correlation = 0.944). Although here the HSI values are averaged over a moving window to graphically display large-scale landscape pattern, the unaltered HSI values would be used as input to HexSim. Although density of linear features is much lower than in the Alberta study area, there is enough separation between blocks of high-quality habitat to suggest that a spatial model that incorporates effects of landscape structure may be informative.

Discussion

The HexSim model has been previously used in population viability analyses for species where individuals hold exclusive territories (Carroll et al. 2003, Schumaker et al. 2004). Boreal caribou are the first species with group, rather than individual, movement dynamics to which HexSim has been applied. The complexity of adapting the HexSim model to caribou life history and group dynamics has slowed initial progress in developing realistic simulations. However, despite these challenges, further effort invested in model development with HexSim is worthwhile due to the potential for HexSim to provide unique insights into the relationship between habitat and viability of boreal caribou populations.

Concurrently with the national critical habitat science review process, a spatial PVA of Ontario boreal caribou populations has been developed (Fryxell and Shuter 2008). This work extends previous caribou simulation models (e.g., Lessard 2005) in several areas, notably by parameterizing movement paths from statistical analysis of detailed movement data rather than by conceptual models (e.g., attraction to high quality habitat). The model of Fryxell and Shuter (2008) is not fully spatial or individual-based as demographic rates experienced by caribou are based on an analytical wolf-moose-caribou predator-prey model. The model is highly suited for exploration of the general types of demographic parameters and landscape conditions that support caribou persistence and thus falls into an intermediate level of complexity between the heuristic non-spatial PVA and the HexSim model. In contrast, the strengths of the HexSim model are that it is fully individual-based, and thus can evaluate



relationships that emerge from spatial interactions between caribou, their predators (e.g., wolves), and alternate prey species (e.g., moose). A “canned” software application such as HexSim inevitably lacks the flexibility of a program developed specifically for a single species, but as a consequence offers the potential for greater standardization and comparability between study areas and between species than is possible with a custom-built program such as used in Fryxell and Shuter (2008).

Although a conclusive evaluation of the potential for SEPM as a decision support tool in the boreal caribou conservation planning process is not possible in this report, initial results do shed light on the four questions outlined in the introduction (adequacy of habitat and demography data, relevance compared to and integration with results from other methods). The spatial (habitat) data from the two case study areas appear adequate for conducting PVA simulations. However, although the habitat suitability model based on vegetation type and linear features generally matches observed caribou distribution in the Alberta study area, there are contrasts in some areas (high habitat quality with no herds observed) that needed to be further evaluated. The demographic data available for the Alberta study area also appear adequate for HexSim parameterization, as estimates of adult and calf survival by major habitat class in disturbed and undisturbed habitats can be derived from field data. Suggested methods for integrating spatial PVA results with those from the environmental niche analysis, meta-analysis, and heuristic PVA have been described above. Although it is not yet possible to conclusively evaluate whether SEPM tools will inform recovery planning in ways not possible with other methods, the potential benefits justify further model development as described above.

The boreal caribou conservation planning process has at least three stages: 1) the now-completed critical habitat science review, 2) assessment of what constitutes effective protection, to be completed over the coming months, and 3) longer-term conservation planning efforts at the provincial and federal level. In the shorter term of the first two stages of planning, it seems clear based on the challenges encountered so far in parameterizing the caribou HexSim model that the SEPM approach is best developed as a heuristic tool for illuminating area and connectivity effects in representative case study areas. This is due to limitations on available habitat data, but also as a strategy to concentrate effort on refinement of the SEPM model before application to a large number of study areas. Although initial predictions can be developed from a SEPM at a relatively early stage in the modeling process, they should not be used in a decision support context until exhaustive sensitivity analysis has been completed. In the interim, static habitat (HSI or RSF) models (e.g., Sorenson et al. 2008) should be developed and used to track amount and quality of habitat at local and range-wide scales, and perhaps refined through consideration of landscape structure (core area size, etc.) in addition to habitat amount. These static models are a foundation for and complementary to SEPM model development.

Over the longer term (stage 3), SEPM seems a promising approach for addressing issues that have arisen during the critical habitat science review. This is because SEPM output directly addresses the relative risk to population persistence of alternate conservation strategies and



thus what constitutes effective protection. By evaluating persistence under scenarios where habitat is maintained, enhanced, or decreased, SEPM output supports placing populations within a framework of range adequacy and resiliency as developed in the critical habitat science review process. SEPM are also currently the best tool for rigorously assessing the importance of intra- and interpopulation connectivity for persistence of boreal caribou, as in cases where large-scale industrial development may fragment habitat of formerly continuous populations.

The next steps in SEPM development for the two case study areas described here fall into several categories. Initially, the focus will be on parameter refinement and sensitivity analysis under the current static landscape. Availability of seasonal HSI models as in Manitoba will allow the SEPM to incorporate seasonal ranges and movement between them. More complex population dynamics (e.g., Allee effects) will be incorporated in the simulations. Once a satisfactory parameter set for current landscapes has been developed, simulations will incorporate future scenarios, including threats from development and climate change, and simulation of landscape dynamics due to forest succession and fire. The value of the SEPM analysis will be enhanced by continued interaction and integration of the spatial PVA with the other three facets of the science review (environmental niche analysis, meta-analysis, and non-spatial PVA).

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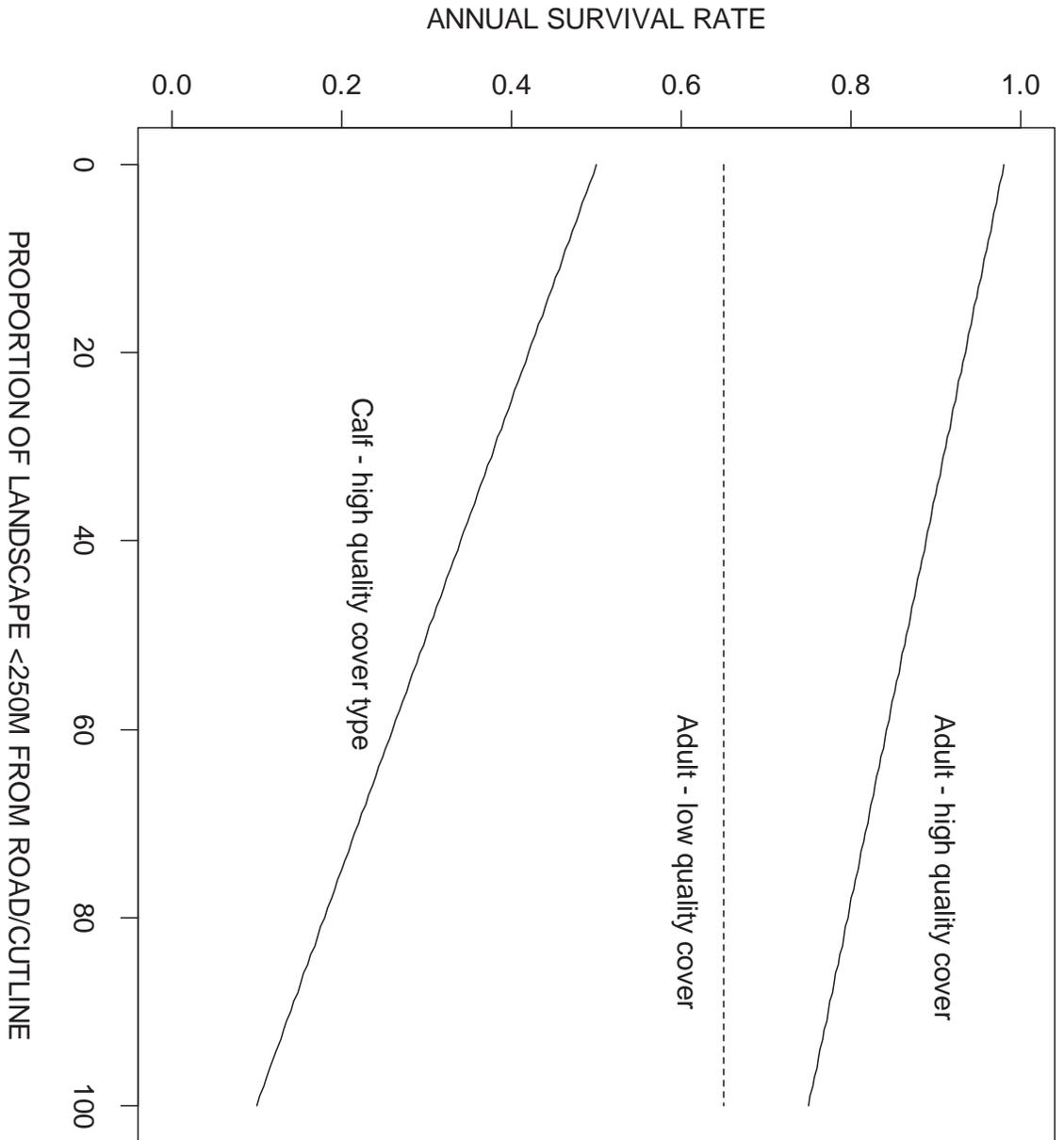
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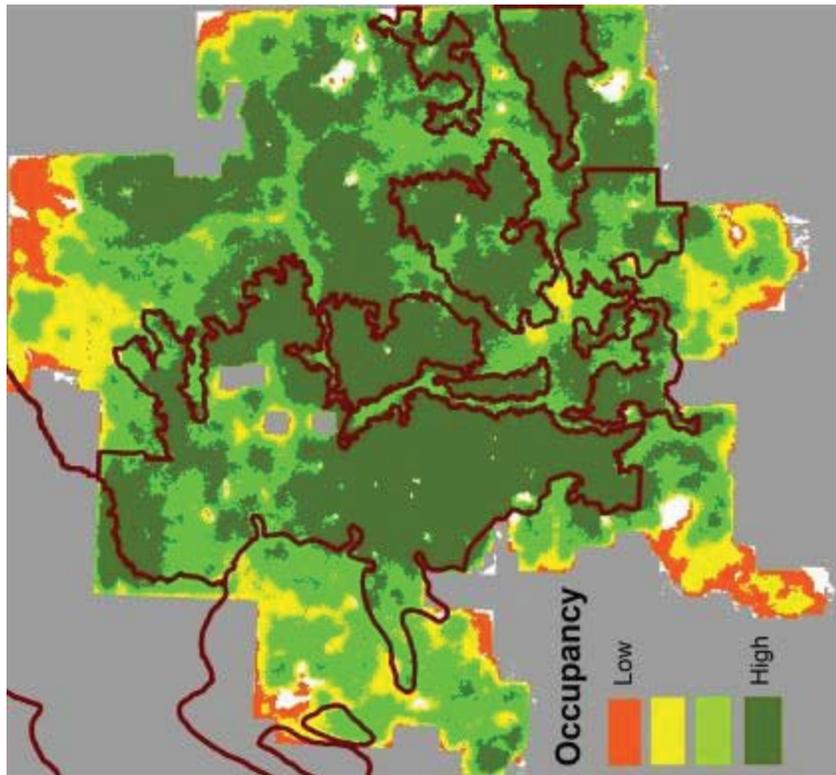
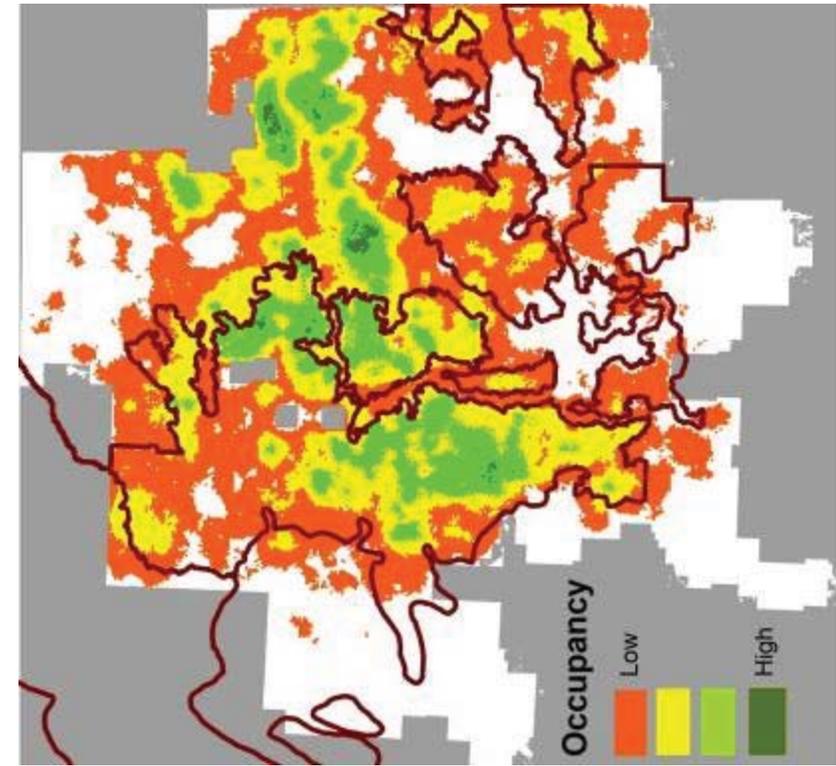
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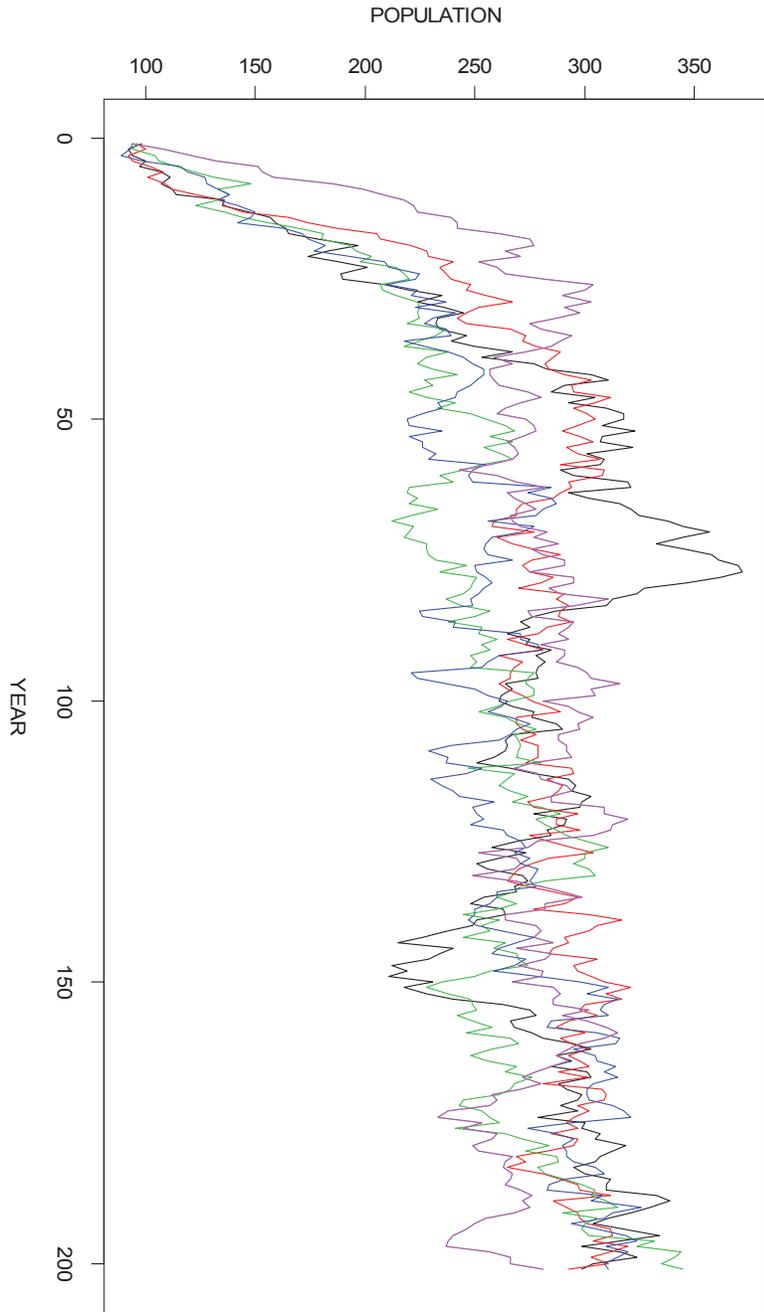
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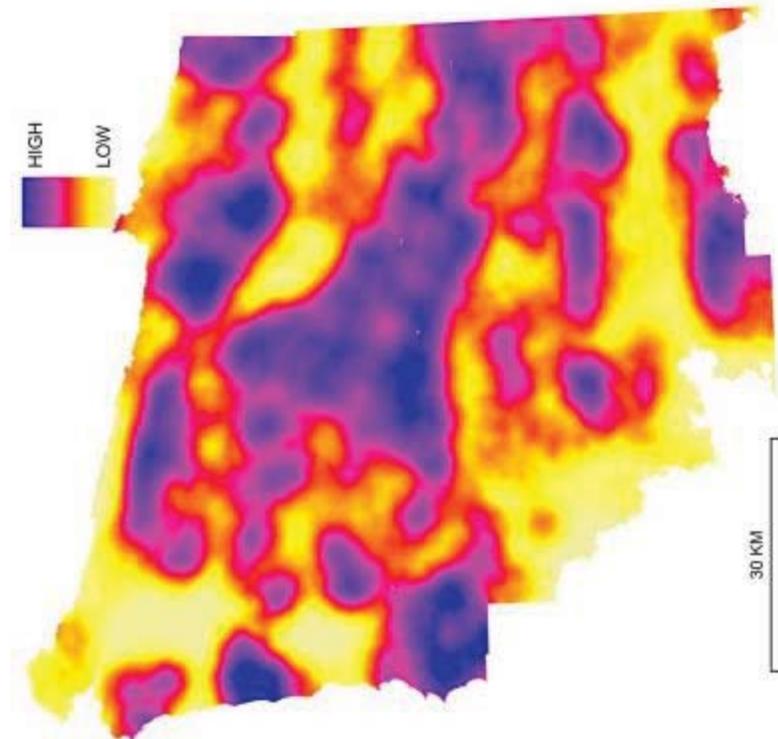
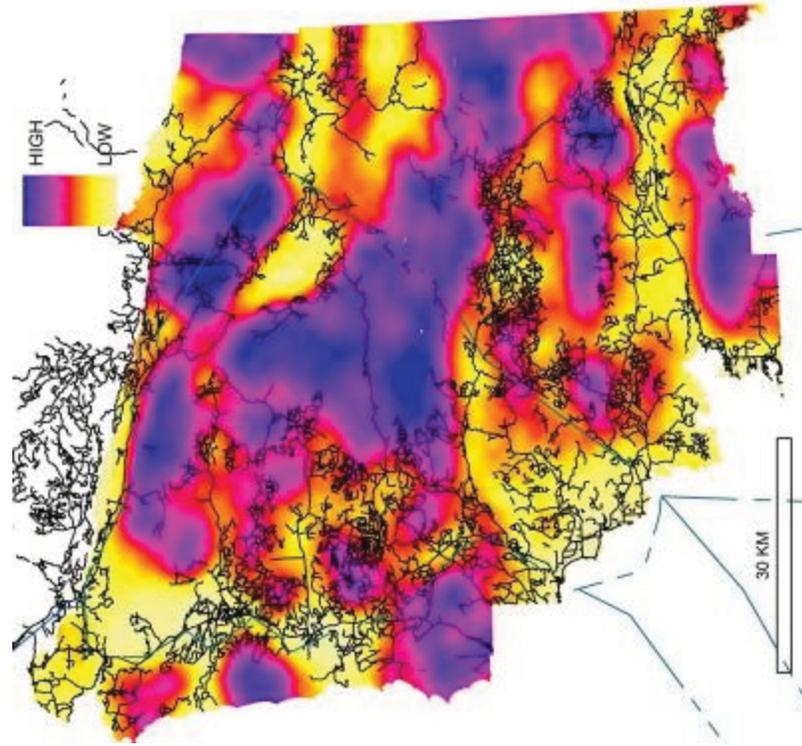
Appendix 6.7 - Figure 1. Initial parameterization for the Alberta study area of the relationship between caribou survival rate and proportion of landscape within 250 meters of linear disturbance features (roads and seismic lines).



Appendix 6.7 - Figure 2. Results from initial HexSim simulations for the north/eastern Alberta case study area under a) habitat condition without effect of linear disturbance, b) habitat condition with effect of current linear disturbance levels.



Appendix 6.7 - Figure 3. Population time series from HexSim simulations for the north/eastern Alberta study area showing long term fluctuations around stable trend. These fluctuations are driven by a combination of demographic stochasticity and dispersal limitation related to habitat pattern.



Appendix 6.7 - Figure 4. Predictions from the EMWCAC HSI model (averaged over a 100 km² moving window) for the Manitoba study area for a) caribou summer habitat, and b) winter habitat, overlaid with linear features.



6.8 Conditional Probability Table

The conditional probability table for the joint distribution of criteria states, with integrated prior probability assignments as referenced in Section 2.6.5. SSfR is the probability of a local population being self-sustaining, given present range and population conditions.

Trend	Size	Disturbance	SSfR	Range Assessment
Declining 0.1	Very Small 0.1	Very High 0.1	0.1	R _{NSS}
		High 0.3	0.2	R _{NSS}
		Moderate 0.5	0.2	R _{NSS}
		Low 0.7	0.3	R _{NSS}
		Very Low 0.9	0.4	R _{NSS}
Declining 0.1	Small 0.3	Very High 0.1	0.2	R _{NSS}
		High 0.3	0.2	R _{NSS}
		Moderate 0.5	0.3	R _{NSS}
		Low 0.7	0.4	R _{NSS}
		Very Low 0.9	0.4	R _{NSS}
Declining 0.1	Above Critical 0.5	Very High 0.1	0.2	R _{NSS}
		High 0.3	0.3	R _{NSS}
		Moderate 0.5	0.4	R _{NSS}
		Low 0.7	0.4	R _{NSS}
		Very Low 0.9	0.5	R _{SS} /R _{NSS}
Stable 0.7	Very Small 0.1	Very High 0.1	0.3	R _{NSS}
		High 0.3	0.4	R _{NSS}
		Moderate 0.5	0.4	R _{NSS}
		Low 0.7	0.5	R _{SS} /R _{NSS}
		Very Low 0.9	0.6	R _{SS}
Stable 0.7	Small 0.3	Very High 0.1	0.4	R _{NSS}
		High 0.3	0.4	R _{NSS}
		Moderate 0.5	0.5	R _{SS} /R _{NSS}
		Low 0.7	0.6	R _{SS}
		Very Low 0.9	0.6	R _{SS}
Stable 0.7	Above Critical 0.9	Very High 0.1	0.6	R _{SS}
		High 0.3	0.6	R _{SS}
		Moderate 0.5	0.7	R _{SS}
		Low 0.7	0.8	R _{SS}
		Very Low 0.9	0.8	R _{SS}
Increasing 0.9	Very Small 0.1	Very High 0.1	0.4	R _{NSS}
		High 0.3	0.4	R _{NSS}
		Moderate 0.5	0.5	R _{SS} /R _{NSS}
		Low 0.7	0.6	R _{SS}
		Very Low 0.9	0.6	R _{SS}
Increasing 0.9	Small 0.3	Very High 0.1	0.4	R _{NSS}



Trend	Size	Disturbance	SS/R	Range Assessment
		High 0.3	0.5	R _{SS} /R _{NSS}
		Moderate 0.5	0.6	R _{SS}
		Low 0.7	0.6	R _{SS}
		Very Low 0.9	0.7	R _{SS}
Increasing 0.9	Above Critical 0.9	Very High 0.1	0.6	R _{SS}
		High 0.3	0.7	R _{SS}
		Moderate 0.5	0.8	R _{SS}
		Low 0.7	0.8	R _{SS}
		Very Low 0.9	0.9	R _{SS}
Unknown 0.5	Very Small 0.1	Very High 0.1	0.2	R _{NSS}
		High 0.3	0.3	R _{NSS}
		Moderate 0.5	0.4	R _{NSS}
		Low 0.7	0.4	R _{NSS}
		Very Low 0.9	0.5	R _{SS} /R _{NSS}
Unknown 0.5	Small 0.3	Very High 0.1	0.3	R _{NSS}
		High 0.3	0.4	R _{NSS}
		Moderate 0.5	0.4	R _{NSS}
		Low 0.7	0.5	R _{SS} /R _{NSS}
		Very Low 0.9	0.6	R _{SS}
Unknown 0.5	Above Critical 0.5	Very High 0.1	0.4	R _{NSS}
		High 0.3	0.4	R _{NSS}
		Moderate 0.5	0.5	R _{SS} /R _{NSS}
		Low 0.7	0.6	R _{SS}
		Very Low 0.9	0.6	R _{SS}



6.9 Estimates of Numbers and Trends for the Boreal Population of Woodland Caribou Provided By Jurisdictions

Note: Caribou local population estimates in the following chart may not fully account for the movement of caribou between jurisdictions within trans-boundary ranges (e.g., some caribou that cross provincial/territorial borders may be represented more than once). Also, some of the local population size estimates and trend data are based primarily on professional judgement and limited data and not on rigorously collected field data.

Local Population refers to the 39 recognized discrete local populations; Unit of analysis refers to the remaining units of which 6 units in NWT are the results of sub-dividing a large area of relatively continuous habitat considered to be occupied by one large population into units of analysis. Eight units in Saskatchewan represent units of analysis for multiple local populations within an area of relatively continuous habitat. The 4 remaining units of analysis found in parts of Manitoba, Ontario, Quebec and Labrador include possible multiple local populations within a large area of relatively continuous habitat. In the absence of defined local populations and units of analysis for these areas, the extent of occurrence was considered to comprise the unit of analysis for these 4 units.



map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
Cross-Jurisdictional						
1	AB/BC Chinchaga	AB – Annual BC - 2004	AB – Precise pop. Trend estimate only (AB does not enumerate caribou) BC - Incomplete	250-300 (includes former Hotchkiss Local Population)	AB- Size estimate based on professional judgement and available field data BC – Average based on several different extrapolations from partial inventory coverage	AB - Rapidly decline (Mean $\lambda = 0.93$ during 2002-2006; Range $\lambda =$ 0.80-1.06) BC – Suspected Declining based on professional judgement
2	AB/NWT Bistcho	AB – 2005 NWT - unknown	AB – Precise pop. Trend estimate only (AB does not enumerate caribou) NWT – Incomplete	300	AB- Size estimate based on professional judgement and available field data NWT – Estimates based on minimum numbers observed from flights	Suspected declining based on professional judgement agreed to by both jurisdictions
3	AB/NWT Steen River \ Yates	AB – 2005 NWT - unknown	AB – Precise pop. Trend estimate only (AB does not enumerate caribou) NWT - unknown	300	AB- Size estimate based on professional judgement and available field data NWT – unknown	Unknown



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
Northwest Territories						
Reported data: Estimates for the units representing continuously distributed local population were derived from density estimates surrounding collared animals, and then extrapolated to larger geographic areas, or for the North Slave region, a density estimate was developed from aerial surveys. Reported trends are expert opinion from NWT based on size estimates over time.						
4	NWT Inuvialuit	2005	Incomplete	Unknown	Unknown	Unknown
5	NWT Gwich'in	2005	Incomplete	500	The population estimate is based on extrapolation of densities from minimum numbers observed from other areas in NWT with collared animals	Increasing based on professional judgement
6	NWT Sahtu	2005	Incomplete	2000	The population estimate is based on extrapolation of densities from minimum numbers observed from other areas in NWT with collared animals	Unknown
7	NWT North Slave	2005	Incomplete	700	The population estimate is based on extrapolation of densities from minimum numbers observed from other areas in NWT with collared animals	Unknown
8	NWT Dehcho (N/SW)	2005	Incomplete	2000	The population estimate is based on extrapolation of densities from minimum numbers observed from other areas in NWT with collared animals	likely decline based on professional judgement
9	NWT South Slave/SE Dehcho	2005	Incomplete	600	The population estimate is based on extrapolation of densities from minimum numbers observed from other areas in NWT with collared animals	likely declining based on recruitment and cow survival—based on 5 years of trend data



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
British Columbia						
10	BC Maxhamish	2004	Incomplete	306	Average based on several different extrapolations from partial inventory coverage	Unknown
11	BC Calendar	2004	Incomplete	291 (best estimate)	Average based on several different extrapolations from partial inventory coverage	Unknown
12	BC Snake Sahtaneh	2004	Incomplete	365 (best estimate)	Average based on several different extrapolations from partial inventory coverage	Suspected Declining Report for the Snake Sahtaneh had 94% adult female survival and calf recruitment of 5-9 calves/100 cows which is essentially a lambda of 1, but the low calf recruitment concluded that the local population was suspected declining. However, the study was too short to make any firm conclusions.
13	BC Parker Core	2007	Incomplete	24 (best estimate)	Average based on several different extrapolations from partial inventory coverage	Unknown
14	BC Prophet Core	2004	Incomplete	54 (best estimate)	Average based on several different extrapolations from partial inventory coverage	Unknown
Alberta						
15	AB Deadwood	2005	Local population trend estimate (AB does not enumerate caribou)	40	Local population size estimate based on professional judgement and available field data	Suspect declining. Local population trend not measured.
16	AB Caribou Mountains	Annual	Local population trend estimate (AB does not enumerate caribou)	400-500	Local population size estimate based on professional judgement and available field data	Rapidly declining (mean $\lambda = 0.92$ during 1995 – 2007 Range $\lambda = 0.73 - 1.14$)



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LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
17	AB Red Earth	Annual	Local population trend estimate (AB does not enumerate caribou)	250-350	Local population size estimate based on professional judgement and available field data	Rapidly declining (mean $\lambda = 0.94$ during 1995 – 2007; Range $\lambda = 0.81 – 1.30$)
18	AB West Side Athabasca River	Annual	Local population trend estimate (AB does not enumerate caribou)	300-400	Local population size estimate based on professional judgement and available field data	Declining (mean $\lambda = 0.99$ during 1993 – 2007; range $\lambda = 0.83 – 1.14$)
19	AB Richardson		Local population trend estimate (AB does not enumerate caribou)	<100	Local population size estimate based on professional judgement and available field data	Unknown. Local population trend not measured.
20	AB East Side Athabasca River	Annual	Local population trend estimate (AB does not enumerate caribou)	150-250	Local population size estimate based on professional judgement and available field data	Declining (mean $\lambda = 0.95$ during 1993 – 2007; range $\lambda = 0.80 – 1.08$)
21	AB Cold Lake Air Weapons Range	Annual	Local population trend estimate (AB does not enumerate caribou)	100-150	Local population size estimate based on professional judgement and available field data	Rapidly declining (mean $\lambda = 0.93$ during 1998 – 2007; range $\lambda = 0.75 – 1.05$)
22	AB Nipisi			60-70		Unknown
23	AB Slave Lake	Annual	Local population trend estimate (AB does not enumerate caribou)	75	Local population size estimate based on professional judgement and available field data	Unknown
24	AB Little Smoky	Annual	Local population trend estimate (AB does not enumerate caribou)	80	Local population size estimate based on professional judgement and available field data	Rapidly declining (mean $\lambda = 0.89$ during 1999 – 2007; Range $\lambda = 0.77 – 1.04$)



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
<p>Saskatchewan Data reported: The survey used by Saskatchewan Wildlife Branch in the 1980s and early 1990s was one developed by government staff based on advice from some caribou researchers at the time. Surveys were conducted in late November or early December (but were never successful for a variety of reasons). Staff then chose to fly as soon as possible after a fresh snowfall, conducting a transect survey each morning using tightly spaced lines to pick up fresh caribou signs and record. Each afternoon staff would return with a helicopter to search out the sign, locate, count and sex/age the animals. In a survey in 1992, a helicopter was used for everything and simply went off transect each time fresh caribou signs were encountered - following up the sign, recording it, and returning to transect. Sunny days with shadows to show up the tracks were preferable in contrast to a typical moose survey. Staff also stratified survey areas for the southern ones that were off the shield. In retrospect minimum counts were obtained rather than total local population estimates, and no attempts were made to define confidence limits. (pers comm. T. Trotter)</p>						
25	SK Davy-Athabasca	2006	N/A	310	Estimate based on habitat based on a density estimate of 0.031 (Al Arsenault pers. Comm.)	Unknown
26	SK Clearwater	2006	N/A	425	Estimate based on habitat based on density estimate of 0.036 (average of density estimates from two adjacent WCMUs)	Unknown
27	SK Highrock-Key	2006	Incomplete	1060	Estimate based on habitat surveys of portions of range based on density estimate of 0.041 (average of two surveys)	Unknown
28	SK Steephill-Foster	2006	Incomplete	1075	Estimate based on habitat and aerial surveys of portions of range and aerial survey in late 1980s based on density estimate of 0.033	Unknown



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
29	SK Primrose-Cold Lake	2006	Incomplete	350	Estimate based on habitat and aerial surveys in early 1990s, and data collected by Alberta based on density estimate of 0.047 (average of two surveys)	Unknown
30	SK Smoothstone-Wapawekka	2006	Incomplete	700	Estimate based on habitat and previous aerial surveys of portions of range in early 1990s, and documented range recession based on density estimate of 0.027 (average of three surveys)	Declining with habitat change based on professional judgement
31	SK Suggi-Amisk-Kississing	2006	Incomplete	430	Estimate based on habitat and previous aerial surveys of portions of range in late 1980s based on density estimate of 0.055 (average of two surveys)	Unknown
32	SK Pasquia-Bog	2006	Incomplete	30	Estimate based on recent genetic work cooperative with Manitoba. Documented range recession. based on density estimate of 0.012 (AI Arsenault pers. Comm.)	Threat of decline based on professional judgement
Manitoba						
Data reported: Year of census (except for Owl Flinstone) and extent of survey coverage were not reported. Trend data is based on local population estimates carried out in the 70's and 80's and in recent years (2007 for Owl Flinstone), that reported similar estimates.						
33	MB Kississing	N/A (not available)	N/A	50-75	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
34	MB Naosap	N/A	N/A	100-200	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
35	MB Reed	N/A	N/A	100-150	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
39	MB William Lake	N/A	N/A	25-40	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
37	MB Wapisu	N/A	N/A	100-125	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
36	MB The Bog	N/A	N/A	50-75	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
38	MB Wabowden	N/A	N/A	200-225	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
40	MB North Interlake	N/A	N/A	50-75	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
41	MB Atikaki-Berens	N/A	N/A	300-500	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
42	MB Owl Flintstone	2007	N/A	71-85	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
43	Manitoba (Remainder of boreal caribou in MB)	N/A	N/A	775-1585	based on professional judgement and periodic local population counts	Stable based on professional judgement and periodic local population counts
Ontario						
44	ON North East Superior (includes Pukaskwa, Gargantua and Pic Islands)	N/A (not available)	N/A	42	estimate is based on compilation of expert opinions and varied survey techniques across the province	Decreasing based on expert opinion
45	ON Michipicoten	N/A	N/A	200	estimate is based on compilation of expert opinions and varied survey techniques across the province	Increasing based on expert opinion
46	ON Slate Islands	N/A	N/A	250	estimate is based on compilation of expert opinions and varied survey techniques across the province	unknown
47	Ontario (remainder of boreal caribou in Ontario)	1996 (questionnaire survey)	Incomplete	5000	estimate is based on compilation of expert opinions and varied survey techniques across the province	unknown
Quebec						
48	QC Val d'Or	N/A (not available)	Complete	30	Local population size estimate based on professional judgement and available field data	Declining based on professional judgement and available field data
49	QC Charlevoix	1998	Complete	75	Local population size estimate based on professional judgement and available field data	Stable based on professional judgement and available field data



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
50	QC Pimouacan	N/A	N/A	134	Local population size estimate based on professional judgement and available field data	Stable based on professional judgement and available field data
51	QC Manouane	N/A	N/A	358	Local population size estimate based on professional judgement and available field data	Stable based on professional judgement and available field data
52	QC Manicouagan	N/A	N/A	181	Local population size estimate based on professional judgement and available field data	Increasing based on professional judgement and available field data
53	Quebec (Remainder of boreal caribou in QC)	Incomplete	Incomplete	6000-12 000	Local population size estimate based on professional judgement and available inventory data for the southern part of the range extent	Suspected stable Supported by Quebec's Comité de rétablissement based on surveyed areas and data confirming that the range extent has not changed.



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
<p>Newfoundland and Labrador</p> <p>Data reported: No Newfoundland local populations are included in the 'threatened' designation. The following local populations occur in Labrador, not Newfoundland and there fore the abbreviation "LAB" has been used at the beginning of the local population's name. Lac Joseph - surveyed in 2000. Full Range (38 00 km2), using a mark-recapture method, Lincoln-peterson/joint hypergeometric maximum likelihood estimator. We have conducted late-winter classifications (March, best indicator of recruitment as calves are 9.5 months old) every year since 2000. Percent calves has ranged between 15 and 20% over that time period, and sex ratios of males/females indicate there are approximately 50 adult males per 100 females (or about 33% males). Between 1999 and 2006 adult survival ranged between 0.788 to 0.913 with a mean value of .852 in this herd, and mean calf survival over the same period is 0.4. Collectively, these suggest that this herd is either stable or slightly declining. Calf recruitment is good, but adult female survival could be better.</p> <p>Red Wine Mountain: The survey in 2001 covered the full range of this herd, or 29 900 km2. The estimator used was also a maximum likelihood estimator (mark-recapture technique). The minimum count (number of unique animals observed) was 67, and revised to 87 in 2003 based on a partial survey of animals associated in groups with radio-collared females in 2003. Calf recruitment is similar to LJ, as is adult female survival. However, survival rates need to be adjusted to account for losses of adult animals due to illegal hunting.</p> <p>Mealy Mountain: Survey in 2005 covered an area of 62 000 km2 (full range). Type was a density-distribution survey (after Gasaway 1986). Survey repeated methods/extent of 2002 census and estimates of population size do not differ significantly (statistically speaking), which suggests that teh population is stable. This herd declined sharply from 2600 to 284 between 1958 and 1975, and has recovered to numbers in excess of 2000 since 2002. Calf recruitment in 2005 was 16%, and adult female survival averaged 89% between 2002 and 2006. The current rate of growth in this herd appears to be 0. However, observed parturition, recruitment, and survivorship schedules suggest this herd has the potential to increase. It is possible that any gains in recruitment are being offset by enhanced mortality of adult (uncollared) animals.</p>						
54	LAB Lac Joseph	2000	Complete	1101	756-1933 ($\alpha = 0.10$)	Unknown
55	LAB Red Wine Mountain	2001	Complete	97	72-189 ($\alpha = 0.10$)	Declining based on professional judgement and available field data; declined from over 800 animals in 1997 to less than 100, and a corresponding change in range size/use has been documented
56	LAB Mealy Mountain	2005	Complete (high density offshore island not included ~ 300 caribou)	2106	765-3447 ($\alpha = 0.10$)	Stable based on professional judgement and available field data



LP map #	Local Population or Unit of analysis	Year of Census	Extent of Survey Coverage	Local Population Size Estimate	Confidence Limits	Current Local Population Trend
57	Labrador (remainder of boreal caribou in Labrador)	N/A	Incomplete	Unknown	N/A	Unknown





7.0

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