Quantifying forest stand and landscape attributes that influence mountain caribou habitat fragmentation and predation rates

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Introduction

Mountain caribou are considered endangered by COSEWIC and the Species at Risk Act has highlighted the need for integrating caribou habitat needs with forest management. Due to the enormous implication of mountain caribou conservation, considerable research has documented habitats needed for winter foraging and ways to maintain this habitat using partial-cutting silvicultural systems, however, conditions needed for connectivity both within home ranges and among populations remains unknown. As well, predation appears to be a dominant source of mortality of mountain caribou. It has been suggested that the increase in predation is linked to an increase in predator numbers due to an increase in alternative prey, which, in turn is caused by a more fragmented landscape of young and old forests, plus, improved predator hunting efficiency due to the road network.

This project has 4 main objectives: 1) test if young forests block movement of mountain caribou. If these forests do block caribou movements, then we will determine if young forests can be managed through silvicultural practices such as thinning, pruning, or uneven planting, to encourage caribou movement. We will also document attributes of young stands that provide foraging opportunities. 2) determine if the apparent discrete populations act as a meta-population with animals dispersing among the populations and determine how the level of inter-population movement affects population viability. We will test factors that influence population fragmentation. 3) investigate the foraging efficiency of predators (wolves) on landscapes of various levels of fragmentation and identify factors effecting kill success, and to document changes in predator functional and numerical response to changes in moose abundance (due to increases in hunting pressure) and, if there are changes in these responses, determine if there are changes in predation rates on caribou. 4) work closely with land-use planners and operational foresters to implement what has been learned about caribou ecology.

In addition to these main objectives, this project will maintain a sample of radio-collared animals over the Columbia, Okanagan/Shuswap, and Headwaters Forest Districts that will be used to: a) refine our knowledge of movements and habitat selection b) be the marked sample in a mark-resight population estimate c) determine mortality factors of adult and juvenile caribou (a major knowledge gap).

This project collects new data on factors that may cause fragmentation of caribou habitat at a variety of spatial scales as well using data previously collected across
the distribution of mountain caribou. This fiscal year, 46 radio-collared caribou were tracked and provided 707 VHF locations and thousands of GPS locations. Seventeen caribou were captured in late March. In addition, 17 moose were monitored, 9 were captured, and 220 VHF locations obtained. Three moose had GPS collars so many locations from these are expected. Also 5 wolves were located 80 times and 6 more were captured in March. Four of these wolves were collared with GPS collars.

Based mostly on the analysis of previously collected data, the project team had 2 papers accepted to scientific journals and 2 other papers were submitted during the 2004-2005 fiscal year. We also completed 5 annual reports, drafts of 2 other reports (not attached) and completed a significant portion of the analyses on another paper planned to be submitted to a journal next year. Due to the numerous and variety of reports already prepared by the project team, the format of this Annual Technical Report is different from the standard scientific format. To meet copyright laws and to not overburden the reader with a report that is large and varied, this Annual Technical Report includes the abstracts of the journal papers and executive summaries of the annual reports. This format was agreed to by the Research Investment Manager of the BC Forest Science Program. The unpublished annual reports that were completed (excluding the drafts) are appended. Some figures are not attached as they increase the size of the reports significantly.

**Abstracts of papers submitted to scientific journals, 2004-2005.**


**Abstract:** We used census results and radiotelemetry locations of >380 collared individuals sampled over the entire distribution of the endangered mountain ecotype of woodland caribou (*Rangifer tarandus caribou* Gmelin, 1788) in British Columbia, Canada, to delineate population structure and document the size and trend of the identified populations. We also describe the spatial pattern of decline, causes and timing of adult mortality and provide estimates of vital rates necessary to develop a population viability analysis. Our results indicate that the abundance of mountain caribou in British Columbia is declining. We found adult female annual survival rates below annual survival rates commonly reported for large ungulates. The major proximate cause of population decline appears to be predation on adult caribou. Spatial patterns of population dynamics revealed a continuous range contraction and an increasing fragmentation of mountain caribou into smaller, isolated subpopulations. The population fragmentation process predominantly occurs at the outer boundaries of the current distribution. Our results indicate that recovery strategies for mountain caribou should be directed at factors contributing to the fragmentation and isolation of mountain caribou populations as well as management strategies aimed to increase adult survival.

Abstract: To select appropriate recovery strategies for endangered populations, we must understand the dynamics of small populations and distinguish between the possible causes that drive such populations to low numbers. It has been suggested that the pattern of population decline may be inversely density-dependent with the per capita rate of increase decreasing as populations become very small, however, empirical evidence of such accelerated declines at low densities is rare. Here we analyzed the pattern of decline of a threatened population of woodland caribou (Rangifer tarandus caribou) in British Columbia, Canada. Using information on the rate of increase relative to caribou density in suitable winter foraging habitat, as well as on pregnancy rates and on causes and temporal distribution of mortalities from a sample of 349 radiocollared animals from 15 subpopulations, we tested 3 hypothesised causes of decline: a) food regulation caused by loss of suitable winter foraging habitat, b) predation-sensitive foraging caused by loss of suitable winter foraging habitat and c) predation with caribou being secondary prey. Population sizes of caribou subpopulations ranged from <5 to >500 individuals. Our results showed that the rates of increase of these subpopulations varied from -0.1871 to 0.0496 with smaller subpopulations declining faster than larger subpopulations. Rates of increase were positively related to the density of caribou in suitable winter foraging habitat. Pregnancy rates averaged 92.4% ±2.24 and did not differ among subpopulations. In addition, we found predation to be the primary cause of mortality in 11 of 13 subpopulations with known causes of mortality and predation predominantly occurred during summer. These results are consistent with predictions that caribou subpopulations are declining as a consequence of increased predation. Recovery of these woodland caribou will thus require a multi-species perspective and an appreciation for the influence of inverse density dependence on population trajectories.


Abstract: Mountain caribou, an ecotype of woodland caribou, are endangered due to the loss and fragmentation of old forests on which they depend. However, a wider array of natural and human factors may limit caribou persistence and isolate populations, and understanding these may help to stop or reverse population declines by forecasting risk and targeting core habitat areas and key linkages for protection, enhancement or restoration. Across most of their historic range, we conducted a bi-level analysis to evaluate factors related to the persistence of, and landscape occupancy within, mountain caribou subpopulations. We used caribou location data from 235 radio-collared animals across 13 subpopulations to derive a landscape occupancy index, while accounting for inherent sampling biases. We analyzed this index against 33 landscape variables of forest overstory, land cover, terrain, climate, and human influence. At the metapopulation level, subpopulation persistence was explained by the extent of wet and very wet climatic conditions, the distribution of both old (>140 yr) forests, particularly of cedar and hemlock composition, and alpine areas. Other important factors were remoteness from human presence, low road density, and little motorized access. At the subpopulation level, caribou landscape occupancy within subpopulations was explained by the distribution of old cedar/hemlock and spruce/subalpine fir forests and the lack of deciduous forests. Other factors impeding population contiguity were icefields, non-forested alpine, hydro reservoirs, extensive road networks, and primary highway
routes. Model outputs at both levels were combined to predict the potential for mountain caribou population persistence, isolation, and restoration. We combined this output with the original occupancy index to gauge the potential vulnerability of caribou to extirpation within landscapes known to have recently supported animals. We discuss implications as they pertain to range-wide caribou population connectivity and conservation.


Abstract: 1) Although population growth rates in ungulates are most sensitive to changes in adult female survival, there is abundant evidence demonstrating that changes in population growth are predominately caused by variation in juvenile survival because adult female survival is relatively stable over time while juveniles are more vulnerable to adverse environmental conditions.
2) It has been hypothesized, that the decline of woodland caribou (Rangifer tarandus caribou) in North America is based on an indirect interaction between caribou, other prey species, and their shared predators (i.e. apparent competition). Apparent competition may asymmetrically impact caribou abundance because the numerical response of their predators may be dependent on alternate prey species that have increased in number because of improved foraging conditions from habitat changes primarily by logging. However, the proposed link between reduced vital rates, habitat condition and population decline consistent with the apparent competition hypothesis has not been documented.
3) Here we present data from a declining population of woodland caribou in British Columbia, Canada. Adult female survival probabilities, based on a sample of >250 radiocollared individuals, varied both spatially and temporally (range from 0.55-0.95). Trends in the number of caribou in subpopulations were correlated with mean annual survival rates of females with subpopulations with the lowest female survival rates declining at the fastest rates.
4) We also found a negative relationship between survival of adult female caribou within subpopulations and environmental variables. Consistent with the apparent competition hypothesis, differences in adult survival among subpopulations were best explained by differences in the amount of early- and mid-seral forest stands within subpopulation ranges.
5) Low adult survival rates were characteristic of rapidly declining populations. Our investigation of factors influencing the underlying causes of adult female mortality rates and population decline is consistent with the apparent competition hypothesis. These results suggest that even without the introduction of exotic species, apparent competition can cause rapid population declines and even extinction where prey availability has increased following anthropogenic habitat modifications and probably changes in climate.

Executive Summaries of Annual Reports


Executive Summary (extracted from annual report by B. McLellan): Genetic data from 196 mountain caribou samples were examined to determine the extent of
population genetic structure and genetic diversity among mountain caribou in British Columbia. Genotype data was available for 8 microsatellite loci from the 93 individuals and for 10 microsatellite loci from the 103 genotypes processed by Wildlife Genetics International. Population genetic structure among the mountain caribou was examined using two assignment tests: 1. program STRUCTURE Version 2.0, which implements a Bayesian clustering approach to identify populations of origin for individuals, and 2. a maximum likelihood-based assignment test was performed. Geographic herds and genetic clusters were compared independently using this test, and the classification method (genetic vs. geographic) which best supported the data was used for subsequent analyses. To quantify the amount of gene flow among mountain caribou, migration rates were estimated using a second Bayesian assignment test, BayesAss Version 1.2. Genetic relatedness among clusters was estimated by calculating Nei’s standard genetic distance between all pairs of populations.

STRUCTURE results indicate that the mountain caribou are separated into five genetic clusters. The five genetic clusters can be arranged geographically from north to south as Cluster 5, Cluster 1, Cluster 3, Cluster 2, and Cluster 4. Cluster 1 consists of 43 individuals from Wells Gray (18 caribou), North Cariboo Mtn (13 caribou), Columbia South (5 caribou), Central Selkirk (4 caribou), South Selkirk (2 caribou), and Columbia North (1 caribou). The distribution of Cluster 1 may suggest that the individuals are migrants or offspring of migrants that dispersed to the geographic region from which they were obtained. However, due to the fragmented nature of caribou habitat in British Columbia, a more probable explanation is that these subpopulations were derived from a common ancestral population, and that subdivision has occurred recently (within the last century).

Cluster 2 consists of 43 individuals from Columbia North (16 caribou), Columbia South (16 caribou), Frisby/Quest (4 caribou), Kinbasket (3 caribou), Wells Gray (2 caribou), Groundhog (1 caribou), and South Selkirk (1 caribou). The largest proportion of caribou forming this cluster was derived from Columbia North and Columbia South (74%). The clustering of caribou from Columbia North, Columbia South, and Frisby/Quest likely occurs because these caribou share a recent ancestor.

Cluster 3 consists of 38 individuals from Columbia North (16 caribou), Columbia South (4 caribou), Wells Gray (4 caribou), Central Selkirk (3 caribou), Groundhog (3 caribou), North Cariboo Mtn (3 caribou), Frisby/Quest (2 caribou), Hart Ranges (1 caribou), and two from unknown locations. Although Clusters 2 and 3 both consist of many caribou previously labelled as Revelstoke, Cluster 3 has a more diverse membership. Furthermore, they are sufficiently divergent to be separated into different genetic groups. However, the rate of recent migration (within the last three generations) between these herds is high (22%), as would be expected from geographically neighbouring subpopulations where movement corridors are maintained.

Cluster 4 consists of 36 individuals from Purcell (28 caribou), Columbia North (3 caribou), Columbia South (1 caribou), Central Selkirk (1 caribou), Frisby/Quest (1 caribou), Wells Gray (1 caribou), and one from an unknown location. The majority (78%) of individuals assigned to Cluster 4 come from the Purcell subpopulation (both North and South). Cluster 4 is the most genetically distinct of the groups (D_s
ranges from 0.45 to 0.66). This herd is small and isolated, due to considerable population declines during the past century. The size and isolation of this subpopulation are likely the main factors contributing to its genetic distinctness.

Cluster 5 consists of 36 individuals from North Cariboo Mtn (17 caribou), Wells Gray (5 caribou), Columbia North (4 caribou), Hart Ranges (3 caribou), Monashee (3 caribou), Allen (1 caribou), and three from unknown locations. North Cariboo Mtn composes the largest proportion of individuals assigned to this cluster (47%).

The maximum likelihood assignment test supported the Bayesian clusters. When clusters were analysed to determine the degree of genetic structure across the region, all cluster memberships were strongly supported, whereas geographic herd structure was not. Considerable population genetic structure exists across the range of mountain caribou in British Columbia. Each of the identified clusters is highly genetically differentiated. The degree of differentiation observed here is greater than that seen for other woodland or barren-ground caribou herds, possibly due to habitat fragmentation and subsequent isolation of subpopulations. The effects of genetic drift could become apparent after only a few generations of reduced gene flow. If gene flow continues to be inhibited, these groups of caribou may rapidly lose genetic diversity and become even more genetically distinct.

Migration rates between most clusters are very low (<5%) except from Cluster 2 to Cluster 3. Caribou from Clusters 2 and 3 belong primarily to Columbia North and South and their neighbouring subpopulations. Movements of caribou from south to north could be very likely, particularly in the past. Results suggest that most migrant individuals are actually offspring of migrants, and that current migration is very rare (data not shown). This is also true for other migration rates noted in Table 3; the majority of identified migrants are hybrid offspring between migrants and resident individuals (data not shown). The detected migrations likely occurred within the past three generations.


Executive Summary: The North Thompson Valley, British Columbia has a history of industrial partial cut logging in high-elevation spruce-fir forests dating back to the early 1960’s. Partial cuts include older “intermediate utilization” or “balsam residual” cuts prior to the 1980’s and more recent selection silviculture systems implemented in the 1990’s to manage impacts of logging on mountain caribou habitat. Mountain caribou use of partial cuts has been documented through VHF and GPS collar point locations over the past 8-10 years, but only recently has caribou foraging on arboreal lichens on residual trees within partial cuts been identified. As a result, caribou trailing methods were modified for the 2004/2005 field season to assess caribou forage selection in partial cuts. This report has two objectives; 1) to provide a background of habitat attributes available in partial cuts, and 2) report preliminary results of caribou forage use in partial cut forests.

Several studies have quantified lichen biomass and composition in high elevation ESSF partial cut forest stands. Older partial cuts have a lower density of large trees but often contain higher per tree lichen biomass than old uncut forests or more
recent partial cuts. High lichen loads are a result of heavy *Bryoria spp.* accumulations that have occurred over the course of several decades in response to the well-ventilated environment and available branch substrate. More recent partial cut forests maintain a higher density of large lichen-bearing trees after logging that retain lichen accumulations similar to old uncut forests.

We trailed caribou in nine (9) different partial cuts during the 2004/2005 winter field season. Caribou forage transects had more large trees (> 30 cm. Dbh) and trees with a more 10-gram lichen clumps (> 1.5 metres) than random transects. Caribou also foraged more intensively on larger diameter trees and trees with more 10-gram lichen clumps (> 1.5 metres). Caribou foraging occurred exclusively on standing trees compared to old forests where caribou utilize a greater variety of sources to obtain forage.

These results are preliminary given the low sample size but do confirm caribou forage use within partial cut forests. More research is needed to confirm target green tree retention levels and attributes for partial cuts and to assess the relative suitability of partial cuts compared to other forest types where they forage.


**Executive Summary:** Mountain caribou in the Columbia Forest District were studied using radio-telemetry from 1992 to 1999. Censuses conducted in 1994, 1995, 1996, and 1997 indicated a relatively stable population of between 290 to 373 animals. In 2002, census efforts were coordinated throughout the range of mountain caribou in British Columbia and resulted in a population estimate of 211 animals for the Revelstoke area, or a decline to 64% of the 1997 estimate. This downward trend occurred in all mountain caribou populations in southern B.C. However, because there were only 5 radio-collared animals remaining in the area in 2002, the sightability of the population (proportion of the total population actually observed) could not be well estimated resulting in a wide confidence interval of 176-269 animals. The decline was confirmed during the 2004 census, with a population estimate of 176. The 90% confidence interval for the 2004 census was 167-210 animals, based on sighting 14 of the 15 radio-collars available.


**Executive Summary:** This report summarises information collected in the first three years of monitoring mountain caribou in the Okanagan/Shuswap Forest District. Animal capture and monitoring were conducted in conjunction with efforts in the Columbia and Headwaters Forest Districts. In March 2005, four caribou were captured and fitted with collars in the following areas: Seymour/Kirbyville pass area, Queest Mountain, and Blanket Glacier. When capture was completed, there was one collared caribou at Queest Mountain, seven in the Seymour River area (includes Ratchford, Kirbyville, and Cayenne Creek) and two collars at Blanket Creek. No collared caribou mortalities were recorded in the Okanagan-Shuswap Forest District or the immediately surrounding areas. A total of 198 VHF locations were recorded
from ten caribou between March 2004 and March 2005. Caribou that were tracked over the whole year were located 19-21 times during this period. An updated shapefile of VHF locations in ArcView shapefile format (BC Albers Projection) is included with this report.

In addition to VHF locations, four GPS collars with locations in the Okanagan-Shuswap Forest District were removed and downloaded in March 2005. These collars were deployed in the Blanket Creek, Queest, Frisby Ridge and Seymour River areas in spring 2003. These locations are in separate files submitted with this report in plot-file format and in ArcView shapefile format (in BC Albers Projection). In total these four collars recorded 5844 locations in total and 2238 of these locations were recorded in the Okanagan-Shuswap Forest District.

Caribou in the Kirbyville-Seymour-Ratchford area appear to have larger group numbers, and were more often sighted with calves than caribou in the Frisby, Queest Mt. or Blanket Creek area. The group of caribou in the Anstey Range is the largest single group of caribou located during telemetry flights in the entire study area (including the Columbia and Headwaters Forest Districts). This area currently receives no heli-ski or snowmobile use. Access management should be a priority in this area and efforts should be made to limit any future recreational use.

The Monashee herd continues to exist at low numbers, with only five caribou sighted during telemetry flights this winter. These flights do not equate to a census, since it is possible another group was living elsewhere during the winter. However, it is notable that no calves were sighted in the group we did see regularly this winter. During early winter, caribou in this herd use the Joss Mountain area, which is outside the current RMZ boundary. This area should be added to the caribou RMZ. Caribou in this herd migrated south to the Vigue Creek area during summer for the second year in a row, confirming use of the RMZ in this area. Anecdotal sightings are also occasionally reported in the Monashee Park area during summer. No reports of caribou in the Pukeashun Mt. area were received, however only one over-flight of this area was conducted in early winter 2004.


Executive Summary: The amount and distribution of recreational backcountry snowmobiling has rapidly increased in southeastern British Columbia in the last 10-15 years. The growth of this activity has let to concern with the conservation of threatened populations of mountain caribou (Rangifer tarandus caribou) due to overlap of areas used by caribou and snowmobilers. The socio-economic impact of snowmobiling is considerable to many small BC communities, however, snowmobiling has the potential to greatly impact caribou herds, and thus, greater understanding of caribou-snowmobile interactions is important.

The purpose of this study is to provide a preliminary examination of the historical use of late winter ranges by caribou relative to snowmobile activity. Historical data on snowmobile use and VHF and GPS radio collar telemetry from caribou in late winter ranges are used to assess change in caribou use of snowmobile areas over time. In addition, data from caribou carrying GPS collars in the winter of 2002/2003 are
examined to determine how they can be used to provide more detailed analysis of interactions between snowmobiles, caribou and their environment.

As with other reports using retrospective analyses, results of this report are inconclusive. Caribou continue to use some areas despite continued high snowmobile use, while other areas are largely avoided. However, this inconsistency in caribou response to snowmobile traffic may be associated with topography. Snowmobiling areas such as Frisby Ridge and Allan Creek Ridge are long, relatively thin, alpine ridges and snowmobile use is largely confined to the main ridge. Caribou continue to use these areas despite the heavy snowmobile use but the caribou were found using spur ridges that provide some distance from the main snowmobile activity and possible greater security. These ridges also provide good late winter habitat in close proximity to early winter and spring habitats. In snowmobile areas such as Queest and Keystone-Standard, where the topography is more moderate and has less control over caribou and snowmobile movements, caribou largely avoid the main snowmobile possibly using alternate ridges or alpine areas as refuge from snowmobiles. Questions remain if continued presence around high snowmobile use areas (Allan and Frisby) or movement onto alternate habitats has significant impacts to animals.

Exploration of data from GPS collars from four animals in the North Thompson over the winter of 2002/2003 suggest this data may provide more detailed study of caribou–snowmobile interactions. However, to be effective this data would need to be used in a disturbance study with more detailed information collected on snowmobile use and environmental variables (i.e. snowfall, temperature).


The interior wet belt of British Columbia has highly productive conifer forests dominated by western red cedar (Thuja plicata) and western hemlock (Tsuga heterophylla) at lower elevations. Mountain caribou, an endangered ecotype of woodland caribou (Rangifer tarandus caribou), live in these late-successional coniferous forests where they depend largely on arboreal lichens as winter forage. These old forests are valuable to the forest industry and understanding winter habitat use by caribou at a variety of spatial scales is required to develop habitat management guidelines that can be implemented from operational to regional planning scales. While radio-telemetry has been used to understand caribou habitat selection patterns at broad scales among and within populations, here we use snow-trailing in Cedar-Hemlock forests between 1992-2003 to study 3 finer scales of habitat selection: 1) forest stands used for foraging from forest stands available (among-stand selection), 2) foraging paths within selected stands relative to random paths within those same stands (within-stand selection), and 3) feeding items along foraging paths. Among stands, the abundance of windthrown trees and standing snags was consistently important to mountain caribou. Within stands, caribou selected paths that had more live trees, snags with branches and bark, and trees with larger diameters. All of these habitat attributes facilitate access to arboreal lichen. Of the potential forage items encountered along foraging paths, caribou preferred to feed on windthrown trees, lichen litterfall and falsebox (Paxistima myrsinites). Our results go beyond telemetry studies by revealing that not all old
forests are of equal value to mountain caribou. Prioritization among old stands will help refine conservation measures, as will silvicultural systems that incorporate key habitat attributes to maintain winter habitat in low-elevation Cedar-Hemlock ecosystems.


Executive Summary: Efforts have been made to quantify wolf prey selection and estimate kill rates to determine the effect that wolves have on ungulate populations, guide management decisions, and direct conservation efforts. However, the majority of wolf-foraging studies occur in winter, when snow tracking permits finding kills from the air or on the ground. Investigating wolf prey selection and kill rates in other seasons may be important as recent studies have shown shifts in diet composition, greater prey diversity in the diet and higher rates of predation, compared to winter seasons. Furthermore, there is the greatest potential for overlap between moose (the wolf’s primary prey in the Columbia Mountains) and endangered mountain caribou during non-winter seasons. As a preliminary study, we used the GPS location data (May - August 2004) to determine the feasibility of determining summer predation patterns from GPS locations. Potential kill sites were identified from GPS clusters having ≥ 2 points that were < 200 m apart. Of the 25 GPS clusters sites investigated, we found evidence of a large mammal predation event at 40% of the sites. The probability of a large mammal predation event being present at a GPS cluster site increased with the number of days a wolf spent at the cluster and decreased if the cluster had been revisited over a number of non-consecutive days. Developing techniques that link GPS collar technology and animal behaviour could be important to increase our understanding of complex systems for the benefit of conservation and management efforts.

Although wolf foraging behaviour can be explained using kill rates and prey selection, landscape patterns can have important effects on the foraging efficiency of predators, and can influence anti-predator strategies of prey. We compared the locations of moose killed by wolves in winter to telemetry locations of live moose (also in winter), to identify landscape patterns that were associated with where wolves were likely to kill moose. Relative to live moose, killed moose were found >250 m further from a major water source (i.e. Goldstream River or Downie Creek). Dead moose were also at higher elevations, 30% closer to a seral edge, and found in areas that contained 30% more logged forest, relative to live moose. However, variability was high, as evidenced by the large standard error of parameter estimates, and the best multivariate model was the intercept-only (null) model. Nonetheless, the analysis identified some unique features of the study area, such as the probable use of open water as a refuge for moose during winter. In other study areas during winter, most water bodies are frozen, which often facilitates foraging efficiency for wolves and a lack of an escape feature for moose. Ultimately our work on moose predation risk will be expanded to include non-winter seasons, and contrasted with the caribou predation risk model being developed by Clayton Apps (and others) to identify zones of shared predation risk between moose and caribou.
Extension

A major objective of our project is to work closely with land-use planners and operational foresters to implement what has been learned about caribou ecology. In 2004-2005 our extension included participation on the North Kootenay Recovery Implementation Group, the Revelstoke caribou recovery group, the Revelstoke Links group, the Kamloops LRMP caribou committee, and the Species at Risk caribou “science team”. In addition, we presented information to the annual meeting of the Okanagan Innovative Forestry Society. In addition, we worked with licencees, MOF district staff, and regional WLAP employees on old-growth retention locations, access management issues, and winter commercial recreation proposals.

Appendix 1. Annual Reports.

NOTE: SOME FIGURES HAVE BEEN REMOVED TO REDUCE THE SIZE OF THE ELECTRONIC FILES. FIGURES WITH TELEMETRY LOCATIONS OF ENDANGERED MOUTAIN CARIBOU HAVE BEEN REMOVED.
Summary of Genetic Analysis
of Mountain Caribou in British Columbia
for the Development of a Habitat Supply Model

Keri McFarlane

2005
Methods

Genetic data from 196 mountain caribou samples were examined to determine the extent of population genetic structure and genetic diversity among mountain caribou in British Columbia. These data included 93 genotypes from Zittlau (2004) and 103 genotypes of individuals obtained from Bruce McLellan and processed by WGI. Genotype data was available for eight microsatellite loci from the 93 individuals described in Zittlau (2004), and for ten microsatellite loci from the 103 genotypes processed by WGI.

Population genetic structure

Population genetic structure among the mountain caribou was examined using two assignment tests. The first assignment test was performed using STRUCTURE Version 2.0, which implements a Bayesian clustering approach to identify populations of origin for individuals (Pritchard et al. 2000). STRUCTURE was used to estimate the probable number of genetic clusters (K) that exist among the 196 mountain caribou. Twenty independent tests were performed, estimating the likelihood of K=1-20 clusters.

To determine how well these genetic clusters were supported, a maximum likelihood-based assignment test (Paetkau et al. 1997) was performed. Geographic herds and genetic clusters were compared independently using this test, and the classification method (genetic vs. geographic) which best supported the data was used for subsequent analyses.

Genetic relationships and migration rates

To quantify the amount of gene flow among mountain caribou, migration rates were estimated using a second Bayesian assignment test, BayesAss Version 1.2 (Wilson and Rannala 2003). This analysis relaxes the assumption of Hardy-Weinberg Equilibrium (HWE) and may be more reasonable for examining populations that do not meet Hardy-Weinberg criteria. The relatively small herd sizes of BC mountain caribou compared to other woodland caribou may lead to deviations from HWE expectations. Conformity of the data to HWE expectations was not examined in this study, but previous studies of other caribou populations have shown that no significant deviations occur for eight of the ten microsatellites (Zittlau 2004).

Genetic relatedness among clusters was estimated by calculating Nei’s standard genetic distance ($D_S$; Nei 1972) between all pairs of populations. Although many genetic distance measures have been described, $D_S$ has been shown to be effective for discerning population-level genetic structure (Takezaki and Nei 1996, Paetkau et al. 1997).

Genetic diversity

Genetic diversity within each cluster was estimated from eight or ten microsatellite loci. The allelic diversity, $A$, indicates the number of alleles per locus, averaged across all loci. Although high allelic diversity generally suggests a high degree of genetic variation, this value can be influenced by sample size. Unbiased expected heterozygosity, $H_E$, estimates the proportion of individuals in each cluster that is expected to possess two different alleles at a given locus (Nei and Roychoudhury 1974). Genetically diverse populations typically possess high $H_E$ values. The unbiased probability of identity, $pI$, is the probability that two randomly chosen individuals are genetically distinct (Paetkau et al. 1998). The lower the $pI$, the more genetically variable is a population.
Results and Discussion

Population genetic structure

STRUCTURE results indicate that the mountain caribou are separated into five genetic clusters, identified as Clusters 1 through 5. The cluster identity of each individual is shown in Column D of the included spreadsheet, Mtn Caribou Genetic Summary.xls. Cluster membership loosely corresponds with geographic herd membership. The five genetic clusters can be arranged geographically from north to south as Cluster 5, Cluster 1, Cluster 3, Cluster 2, and Cluster 4.

Cluster 1 consists of 43 individuals from Wells Gray (18 caribou), North Cariboo Mtn (13 caribou), Columbia South (5 caribou), Central Selkirk (4 caribou), South Selkirk (2 caribou), and Columbia North (1 caribou). The largest proportion of assigned individuals (72%) was from caribou labelled as Wells Gray or North Cariboo Mtn. These are geographically adjacent subpopulations. An additional 12 caribou (28% of the cluster) from more distantly located subpopulations (Columbia South, Central Selkirk, South Selkirk, and Columbia North) were also assigned to this cluster. These additional individuals, although not geographically proximate to the other caribou assigned to the cluster, contain alleles that occur at frequencies representative of Cluster 1. This may suggest that the individuals are migrants or offspring of migrants that dispersed to the geographic region from which they were obtained. However, due to the fragmented nature of caribou habitat in British Columbia, a more probable explanation is that these subpopulations were derived from a common ancestral population, and that subdivision has occurred recently (within the last century).

Cluster 2 consists of 43 individuals from Columbia North (16 caribou), Columbia South (16 caribou), Frisby/Queest (4 caribou), Kinbasket (3 caribou), Wells Gray (2 caribou), Groundhog (1 caribou), and South Selkirk (1 caribou). The largest proportion of caribou forming this cluster was derived from Columbia North and Columbia South (74%). These subpopulations are adjacent and were previously considered to form a single subpopulation, Revelstoke. The Revelstoke subpopulation also included the current Frisby/Queest subpopulation, from which another 10% (4 individuals) of the cluster was formed. The clustering of caribou from Columbia North, Columbia South, and Frisby/Queest likely occurs because these caribou share a recent ancestor, as would be expected if subdivision of Revelstoke occurred recently due to habitat fragmentation.

Cluster 3 consists of 38 individuals from Columbia North (16 caribou), Columbia South (4 caribou), Wells Gray (4 caribou), Central Selkirk (3 caribou), Groundhog (3 caribou), North Cariboo Mtn (3 caribou), Frisby/Queest (2 caribou), Hart Ranges (1 caribou), and two from unknown locations. A large proportion of assignments (42%) to Cluster 3 were from individuals labelled as Columbia North. Several additional assignments to the cluster (34%) occurred from caribou from the neighbouring subpopulations of Columbia South, Wells Gray, Groundhog, and Frisby/Queest.
Although Clusters 2 and 3 both consist of many caribou previously labelled as Revelstoke (84% and 58%, respectively), Cluster 3 appears to have a more diverse membership. Furthermore, they are sufficiently divergent to be separated into different genetic groups using both the Bayesian and maximum likelihood-based assignment tests. It is important to note, however, that the rate of recent migration (within the last three generations) between these herds is high (22%; Table 3), as would be expected from geographically neighbouring subpopulations where movement corridors are maintained.

Cluster 4 consists of 36 individuals from Purcell (28 caribou), Columbia North (3 caribou), Columbia South (1 caribou), Central Selkirk (1 caribou), Frisby/Queest (1 caribou), Wells Gray (1 caribou), and one from an unknown location. The majority (78%) of individuals assigned to Cluster 4 come from the Purcell subpopulation (both North and South). This herd is small and isolated, due to considerable population declines during the past century. The size and isolation of this subpopulation are likely the main factors contributing to its genetic distinctness.

Cluster 5 consists of 36 individuals from North Cariboo Mtn (17 caribou), Wells Gray (5 caribou), Columbia North (4 caribou), Hart Ranges (3 caribou), Monashee (3 caribou), Allen (1 caribou), and three from unknown locations. North Cariboo Mtn composes the largest proportion of individuals assigned to this cluster (47%). The Hart Ranges and Wells Gray subpopulations are adjacent to North Cariboo Mtn and compose an additional 22% of the cluster.

Results from the maximum likelihood assignment test supported the Bayesian clusters. When clusters were analysed to determine the degree of genetic structure across the region, all cluster memberships were strongly supported (Table 1), whereas geographic herd structure was not (data not shown). In four of the five clusters, over 90% of individuals were assigned to their own cluster. Cross-assignments only occurred for Cluster 4, where 11% of individuals cross-assigned to Cluster 3. Possibly, individuals from Clusters 3 and 4 came from a common ancestral herd. A second explanation is that gene flow occurs at low rates between Clusters 3 and 4. However, the low migration rates detected using BayesAss do not support the occurrence of recent gene flow (i.e., within the past three generations; Table 3). Alternatively, these cross-assignments could reflect the high diversity of Cluster 3 compared to Cluster 4 (Table 4). Cluster 3 contains a wide range of alleles compared to Cluster 4, which could skew assignments toward Cluster 3 if Cluster 4 individuals possessed alleles that have become rare in that cluster due to population bottlenecks.

Considerable population genetic structure exists across the range of mountain caribou in British Columbia. Each of the identified clusters is highly genetically differentiated. The degree of differentiation observed here is greater than that seen for other woodland or barren-ground caribou herds (Zittlau 2004). Possibly, this is a result of habitat fragmentation and subsequent isolation of subpopulations. The effects of genetic drift could become apparent after only a few generations of reduced gene flow. If gene flow continues to be inhibited, these groups of caribou may rapidly lose genetic diversity and become even more genetically distinct.
Table 1. Proportion of assignments from each cluster, as determined by the maximum likelihood assignment test. Values along the diagonal indicate the proportion of individuals that were assigned to their source cluster. Values greater than 10% are bolded.

<table>
<thead>
<tr>
<th>Source</th>
<th>Sink Cluster 1</th>
<th>Cluster 2</th>
<th>Cluster 3</th>
<th>Cluster 4</th>
<th>Cluster 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cluster 1</td>
<td><strong>0.91</strong></td>
<td>0.05</td>
<td>0.02</td>
<td>-</td>
<td>0.02</td>
</tr>
<tr>
<td>Cluster 2</td>
<td>-</td>
<td><strong>0.98</strong></td>
<td>-</td>
<td>-</td>
<td>0.02</td>
</tr>
<tr>
<td>Cluster 3</td>
<td>-</td>
<td>0.03</td>
<td><strong>0.95</strong></td>
<td>-</td>
<td>0.03</td>
</tr>
<tr>
<td>Cluster 4</td>
<td>0.06</td>
<td>0.06</td>
<td>0.11</td>
<td><strong>0.78</strong></td>
<td>-</td>
</tr>
<tr>
<td>Cluster 5</td>
<td>-</td>
<td>-</td>
<td>0.06</td>
<td>0.03</td>
<td><strong>0.92</strong></td>
</tr>
</tbody>
</table>

Genetic relationships and migration rates

Pairwise genetic distances between the five clusters of mountain caribou ranged from $D_S = 0.24$ to 0.66 (Table 2). These distances fall within the range of $D_S$ values measured between caribou herds across northwestern North America using eight microsatellite loci (Zittlau 2004). However, the $D_S$ values reported here are generally smaller than those previously measured between Mountain and Northern ecotype caribou from British Columbia ($D_S = 0.174$ to 1.061; Zittlau 2004). Sampling of mountain caribou in Zittlau (2004) was not as extensive as in this study, and may not have accurately reflected the genetic relationships among the herds examined.

The smallest genetic distance was measured between Clusters 1 and 5 ($D_S = 0.24$), suggesting that the caribou belonging to these clusters are more closely related than to caribou belonging to other clusters. Clusters 1 and 5 consist of geographically proximate individuals, and it is likely that caribou from the two areas have mixed to some extent. Cluster 4 is the most genetically distinct of the groups ($D_S$ ranges from 0.45 to 0.66). The genetic distinctness reflects the fact that this cluster consists primarily of Purcell caribou, which have experienced significant population declines and subsequent isolation. Without increased gene flow to the subpopulations in southern British Columbia, this group of caribou may become more genetically distinct due to the effects of genetic drift.
Table 2. Pairwise genetic distance (DS) between clusters of mountain caribou. Bolded values indicate highest and lowest DS.

<table>
<thead>
<tr>
<th></th>
<th>Cluster 1</th>
<th>Cluster 2</th>
<th>Cluster 3</th>
<th>Cluster 4</th>
<th>Cluster 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cluster 1</td>
<td></td>
<td>0.41</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cluster 2</td>
<td>0.31</td>
<td>0.27</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cluster 3</td>
<td>0.66</td>
<td></td>
<td>0.52</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cluster 4</td>
<td></td>
<td>0.24</td>
<td></td>
<td>0.48</td>
<td></td>
</tr>
<tr>
<td>Cluster 5</td>
<td></td>
<td></td>
<td>0.36</td>
<td></td>
<td>0.48</td>
</tr>
</tbody>
</table>

Migration rates among clusters are shown in Table 3. Migration rates between most clusters are very low (<5%) except from Cluster 2 to Cluster 3. Caribou from Clusters 2 and 3 belong primarily to Columbia North and South and their neighbouring subpopulations. Movements of caribou from south to north could be very likely, particularly in the past. Results suggest that most migrant individuals are actually offspring of migrants, and that current migration is very rare (data not shown). This is also true for other migration rates noted in Table 3; the majority of identified migrants are hybrid offspring between migrants and resident individuals (data not shown). The detected migrations occurred within the past three generations.

Table 3. Mean migration rates (m) into each cluster of mountain caribou. Values along the diagonal indicate the proportion of individuals that remained in their source cluster. Values greater than 10% are bolded. No standard deviations were greater than 0.05.

<table>
<thead>
<tr>
<th>Immigrants From</th>
<th>To Cluster 1</th>
<th>Cluster 2</th>
<th>Cluster 3</th>
<th>Cluster 4</th>
<th>Cluster 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cluster 1</td>
<td>0.9766</td>
<td>0.0034</td>
<td>0.0113</td>
<td>0.0070</td>
<td>0.0073</td>
</tr>
<tr>
<td>Cluster 2</td>
<td>0.0054</td>
<td>0.9864</td>
<td>0.2200</td>
<td>0.0319</td>
<td>0.0065</td>
</tr>
<tr>
<td>Cluster 3</td>
<td>0.0039</td>
<td>0.0024</td>
<td>0.7502</td>
<td>0.0048</td>
<td>0.0294</td>
</tr>
<tr>
<td>Cluster 4</td>
<td>0.0072</td>
<td>0.0047</td>
<td>0.0079</td>
<td>0.9516</td>
<td>0.0095</td>
</tr>
<tr>
<td>Cluster 5</td>
<td>0.0069</td>
<td>0.0031</td>
<td>0.0106</td>
<td>0.0048</td>
<td>0.9472</td>
</tr>
</tbody>
</table>

**Genetic diversity**

Significant genetic diversity exists across the range of mountain caribou in British Columbia (Table 4). The mountain caribou clusters have higher diversity levels than those reported for most other ungulate species (Broders et al. 1999, Holm et al. 1999, Wilson and Strobeck 1999, Polziehn et al. 2000, Anderson et al. 2002). The values reported in this study are comparable to genetic diversities measured in other woodland caribou herds, but lower than those reported for barren-ground herds (Zittlau 2004). Expected heterozygosity and allelic diversity can reflect the genetic health of a herd. Overall heterozygosity can be indicative of the short-term potential for adaptation. Individuals with high heterozygosity have more alleles available that could increase fitness and confer an improved adaptation to changing environments (Lande and Shannon 1995). In comparison, the allelic diversity of a population can reflect the long-term potential of a population to adapt (Lande and Shannon 1995).

Cluster 4, which consists predominantly of endangered Purcell caribou, has the lowest levels of diversity. Low genetic diversity can be a consequence of population
fragmentation, isolation, and/or bottlenecks. Furthermore, when populations are maintained at small sizes, genetic diversity can be lost rapidly and increase the risk of inbreeding depression (Keller and Waller 2002). Inbreeding depression could lead to reduced fitness or reproduction (Ralls et al. 1979, Saccheri et al. 1998). In addition, reduced genetic diversity may limit the adaptive potential of a population. Although the diversity levels in Cluster 4 are not so low as to suggest that inbreeding is an immediate threat, the lower diversity levels may reflect a warning sign. These caribou may require additional management attention to reduce the rate at which diversity is further lost. When considered together, mountain caribou possess very high levels of diversity, comparable to that observed in large barren-ground herds (Zittlau 2004). This is likely due to the fragmented nature of mountain caribou in British Columbia. Genetic diversity can be better maintained across subdivided populations than in a single randomly breeding population (Lacy 1987). This could occur because a different subset of alleles may become established in each subpopulation due to genetic drift. The differences in allele frequencies can be further intensified if each subdivided population is adapted to a different environment. The drawback, however, is that each subpopulation may lose genetic diversity rapidly and be at risk for stochastic effects.

Table 4. Measures of genetic diversity in mountain caribou herds from British Columbia

<table>
<thead>
<tr>
<th>Genetic Cluster</th>
<th>Sample Size</th>
<th>Allelic Diversity</th>
<th>Expected Heterozygosity</th>
<th>Probability of Identity (1 in)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>43</td>
<td>9.0</td>
<td>76.6%</td>
<td>1.5 x 10^{-11}</td>
</tr>
<tr>
<td>2</td>
<td>43</td>
<td>7.4</td>
<td>75.1%</td>
<td>4.6 x 10^{-10}</td>
</tr>
<tr>
<td>3</td>
<td>38</td>
<td>8.2</td>
<td>77.6%</td>
<td>2.9 x 10^{-11}</td>
</tr>
<tr>
<td>4</td>
<td>36</td>
<td>6.3</td>
<td>63.3%</td>
<td>2.1 x 10^{-9}</td>
</tr>
<tr>
<td>5</td>
<td>36</td>
<td>8.3</td>
<td>77.8%</td>
<td>4.3 x 10^{-11}</td>
</tr>
<tr>
<td>All mountain caribou</td>
<td>196</td>
<td>11.7</td>
<td>80.5%</td>
<td>2.1 x 10^{-12}</td>
</tr>
</tbody>
</table>
References


Mountain Caribou Use of Partial-Cut Forests in the North Thompson Valley, British Columbia

A review of habitat attributes in partial cut stands and results from forage use transects from the 2004/2005 winter field season

March 2005

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Acknowledgements

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Summary

The North Thompson Valley, British Columbia, has a 40 year history of industrial partial cut logging in high elevation spruce-fir forests. Partial cuts include older “intermediate utilization” or “balsam residual” cuts prior to the 1980’s and more recent selection silviculture systems implemented in the 1990’s to manage impacts of logging on mountain caribou habitat. Mountain caribou use of partial cuts has been documented through VHF and GPS radio collar point locations over the past 8-10 years, but only recently has caribou foraging on arboreal lichens on residual trees within partial cuts been identified. As a result, caribou trailing methods were modified for the 2004/2005 field season to assess caribou forage selection in partial cuts. This report has two objectives; 1) to provide a background of habitat attributes available in partial cuts, and 2) report preliminary results of caribou forage use in partial cut forests.

Several studies have quantified lichen biomass and composition in high elevation ESSF partial cut forest stands. Older partial cuts have a lower density of large trees but often contain more lichen per tree than old uncut forests or more recent partial cuts. High lichen loads and in particular an abundance of Bryoria spp. have accumulated over several decades in response to the well-ventilated environment and available branch substrate. More recent partial cut forests maintain a higher density of large lichen-bearing trees after logging that retain lichen accumulations similar to old uncut forests.

We trailed caribou in nine (9) different partial cuts during the 2004/2005 winter field season. Caribou forage transects had more large trees (>30 cm. Dbh) and trees with a more 10-gram lichen clumps (<1.5 m high in the canopy) than random transects. Caribou also foraged more intensively on larger diameter trees and trees with more lichen (> 1.5 m above the ground). Caribou foraging occurred exclusively on standing trees compared to old forests where caribou utilize a greater variety of sources to obtain forage.

These results are preliminary given the low sample size but do confirm caribou use of partial cut forests for foraging. More research is needed to confirm target green tree retention levels and attributes of partial cuts used by caribou for foraging and to assess the relative suitability of partial cuts compared to other forest types used by caribou for foraging.
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Introduction

**Background**

Partial-cut logging has occurred in high elevation Engelmann spruce – sub-alpine fir (ESSF) forests in the North Thompson Valley of British Columbia since the 1960’s. An early form of partial cutting known as Intermediate Utilization or “balsam residual” logging, was commonly practiced from the 1960’s to the early 1980’s. The term “Intermediate Utilization” (IU) refers to timber utilization standards of the time, where only trees greater than 17.5 cm DBH or 5 metres in height were harvested. Trees under these specifications were left standing after logging often leaving cut-blocks with a wide range of residual forest structure. Very little partial cutting occurred in the 1980’s and residual trees in many of the existing IU blocks throughout the valley were later felled and the sites either chemically or mechanically site prepped to improve regeneration success. In the early 1990’s selection silviculture systems “single-tree selection” and “group selection” were introduced specifically to address concerns over loss of mountain caribou habitat (Stevenson et al. 1994). Each selection system reduced the existing stand volume by about 30% as part of a 3-pass approach with 80-120 years between subsequent passes. Using this system, at least one third of the stand would be older than 160 yrs at all times. The goal of maintaining this proportion of old trees was to ensure that some large, lichen-bearing trees remained through the rotation to provide suitable lichen biomass conditions for continued caribou forage (Armleder and Stevenson, 1996).

Mountain caribou use of partial cut forests in the North Thompson Valley has been recorded over the past 8-10 years through monitoring of VHF and GPS telemetry collars. These methods documented caribou use within partial cut forests, but contains no information on activity within the stand. The first documented use of partial cuts by caribou for foraging in the North Thompson Valley occurred in 2003/2004 while trailing caribou on the snow. Caribou trails were observed in partial cuts with evidence of foraging on lower canopy lichen in standing residual trees (Lewis 2004, unpublished data). In some instances the caribou were foraging intensively within the stand and tracks were often observed in the same stand over several days suggesting significant use.

While a substantial amount of work has been completed in the North Thompson and other areas quantifying stand structural and arboreal lichen attributes within partial cuts, no information existed on caribou forage selection within these stands. As a result, we modified the trailing
methodology for the 2004/2005 season to assess caribou foraging activity in partial cut stands. This methodology is adapted from similar studies used primarily to quantify microhabitat and forage selection in mature and old stands (Terry et al. 2000, Furk 2003, McLellan et al. In Prep).

The objective of this report is to summarize existing knowledge on lichen availability and use by caribou in partial cut forests. The report is separated into two components:

1. A review of lichen forage availability in different partial cut stands relative to stand history and residual forest structure.

2. To summarize preliminary results of the 2004/2005 field season detailing attributes selected along caribou forage transects
Part 1. A Review of Lichen Forage Availability within Various Partial Cutting Conditions

**Lichen Biomass and Composition in Partial Cut Forests**

Several studies have compared lichen biomass in partial cut forests to uncut old forest stands. Rominger et al. (1994) compared lichen abundance on lower branches of live subalpine fir in 10-year old intermediate utilization (IU) stands with low residual basal areas (9m² and 23m²/ha) to adjacent uncut forests in southeastern BC. They found lichen biomass did not significantly differ between partial cuts and the adjacent uncut old forests. Lewis (2001) used a retrospective analysis to compare lower canopy (<4.5 metres) lichen biomass between single-tree selection (ST) stands (33% basal area removal), older high removal intermediate utilization stands (IU) and adjacent uncut old-growth stands in the North Thompson Valley. Overall he found no significant difference in average lower canopy lichen biomass between the different the IU and uncut stands, while the ST stand had lower average lichen biomass.

While partial cuts typically have a lower density of large trees, they also have a higher proportion of trees with high lichen biomass (Miller et. al. 1999, Lewis 2001). Miller et al. (1999) found partial cuts to have more trees with high lichen loads and fewer trees with low lichen loads compared to uncut forests where caribou foraged. Lewis (2001) found partial-cut IU stands had a slightly higher proportion of trees in Lichen Class¹ 4 and 5 (average 250 and 800 grams) lichen compared to ST and UC stands (Figure 1-1). Single Tree selection stands had a lower proportion of Lichen Abundance Class 4 and 5 and a higher proportion of Class 1(5 grams). Otherwise, the distribution of trees in each Lichen Class was similar between the UC and ST stands.

---

¹ Lichen Abundance Classes follow Armleder et al. (1992)
Figure 1-1. The proportion of measured trees in each lichen abundance class category in uncut old growth (UC), intermediate utilization partial cuts (IU) and single-tree selection (ST) stands from Lewis (2001). Lichen abundance classes follow Armleder et al. (1992). Trees with Lichen Class = 0 indicate trees with no lichen loading.

The high lichen biomass accumulations found on larger residual trees in old IU partial cuts are a result of *Bryoria spp.* accumulation following logging (Lewis 2004). Rominger et al. (1994) noted that of all arboreal lichen, lower branches in partial cuts 10 years after logging contained a higher proportion *Bryoria spp.* than uncut stands. Miller et al. (1999) also noted that residual trees in partial cuts had higher proportions of *Bryoria spp.* than at caribou forage locations in old stands. Lewis (2001) found that total lower canopy lichen biomass on residual trees where lichen was present consisted of greater than 80% *Bryoria spp.* in IU partial cut blocks in the North Thompson (Figure 1-2). In addition, lower canopy lichen composition on trees in ST stands also had a slightly higher proportion of trees with 30-50% *Bryoria spp.* compared to uncut stands.
Lewis (2004) found that *Bryoria spp.* composition in the lower canopy of trees in partial cuts in the North Thompson increased with time since logging and in partial cuts with lower residual basal area. This increase in the proportion of *Bryoria spp.* was due to a dramatic increase in lower canopy *Bryoria spp.* biomass after logging, with as much as a ten-fold increase in biomass in forests with similar residual structure in as little as 15-20 years (Lewis 2004). Lewis (2004) concluded these heavy accumulations of *Bryoria spp.* lichen on residual standing trees in old IU stands are a result three conditions that are unique to these old partial cuts. First, many of the small residual sub-alpine fir trees were very old trees (HOW OLD) and thus, provide very suitable substrate (old branches with a inner defoliated zone) for lichen accumulation. Second, the high levels of tree removal resulted in a very different microclimate than the pre-logging old-growth forests. The residual trees have been subject to greater sun exposure and very well ventilated conditions that result in more rapid drying of branches following moisture events (See Goward 1998). Third, *Bryoria* has had many years to accumulate. *Bryoria* spp. does not appear to be significantly limited by dispersal (Goward 2004) and colonize and accumulate relatively quickly on branches (Arsenault et al. 1997). Lewis (2004) concluded that a minimum of 15-20 years is
required for *Bryoria* spp. to accumulate to appreciable biomass loading within these stands. In contrast to *Bryoria*, *Alectoria* was generally absent in IU partial cuts except on larger trees. Unlike *Bryoria*, Lewis (2004) found a strong linear relationship between *Alectoria* abundance and remnant forest structure (basal area) rather than time since logging, which was more important for *Bryoria*. Greater *Alectoria* biomass occurred in the lower canopy in stands with higher residual basal area and stands with a higher proportion of large trees. Lewis (2004) concluded that somewhat more sheltered microenvironments afforded by larger trees provide more optimal conditions for *Alectoria* to persist in high removal partial cuts after logging. He suggested that theretention of large trees and particularly patches of large trees will be important to ensure *Alectoria* thalli remain available to grow and colonize new trees in the regenerating partial cut forest.

**Lichen relationships to Residual Forest Structure**

Post-logging lichen biomass in partial cut stands is largely a function of retention of large trees. Stevenson (1979) found no relationship between tree size and lower canopy lichen mass in uncut forests due to the high variability in lower canopy lichen biomass on trees of the same size. Lewis (2001) showed that while lichen biomass in the lower canopy of individual trees can be quite variable, higher lichen abundances typically occur on larger diameter trees (Figure 1-3). Much of the variability in lower canopy lichen abundance is due to differences in tree canopy architecture between tree species and trees at different positions in the canopy (Lewis 2003). Many large trees, primarily large Engelmann spruce, had negligible lower canopy lichen and were rated into class 1 and 2 mainly because these trees had few low branches to support lichen. Lichen class 2 and 3 showed the greatest range of tree diameters, ranging from 7.5 cm Dbh to 60cm Dbh. High lichen loads (Lichen Class 4 and 5) occur mainly on mid to large diameter co-dominant and intermediate canopy trees (20-40cm Dbh) that have more defoliated or partially defoliated branches near the ground (Lewis 2003).
Figure 1-3. A comparison of tree diameters associated with lichen abundance classes in uncut old growth (UC), intermediate utilization partial cuts (IU) and single-tree selection (ST) stands from Lewis (2001). Box plots show the median and 25% and 75% inter-quartile range, while whiskers present the 90% confidence interval. Sample sizes are reported below each box plot on the x-axis.
Caribou point locations, monitored through VHF and GPS radio collars over the past 8-10 years, have documented caribou using partial cut forests. Caribou trailing from the 2003-2004 season identified caribou foraging on arboreal lichens from residual trees within older high elevation ESSF partial cuts stands. The objectives of this portion of the study are:

- To document caribou use of partial-cut forest stands for foraging at two scales: 1) Comparing the path chosen by caribou relative to random paths within the partially cut stand 2) Along the caribou path, compare food items that were foraged on at different intensities relative to items that were ignored.

- To assess caribou forage selection in partial-cut forests and to identify factors that influence caribou movements while foraging within a partial-cut stand.

- To provide information to help determine target green-tree retention levels and tree attributes that may provide stand-level lichen biomass conditions suitable for caribou use after partial-cut logging.

**Methods**

When caribou use of a partial cut stand was identified we assessed caribou forage use within the partial cut stand based on methods similar to those of Terry et al. (2000) and McLellan et al. (In prep). The tracks of the target group of caribou were followed until evidence of foraging (e.g. trampling around base of tree) was encountered. From this starting point we investigated microhabitat selection and the foraging decision process by collecting data along two linear transect types. First, we centred foraging path transects (FPT) along the caribou track. We used FPTs to identify snow and vegetative characteristics along the path chosen by the animal and also the characteristics of trees that caribou did or did not forage on. Each FPT is 150 m in length and 2-m wide to reflect the "search path" of a foraging caribou that may take a small step to reach shrubs or lichen-bearing branches (Terry et al. 2000). Second, to sample the immediate area of forest stand that the animal was using, we used foraging area transects (FAT). Each FAT was also 2 x 150 m, but was
subdivided into three 50-m-straight segments starting every 50 m along the FPT, but following a random bearing.

Along each transect type, we estimated feeding intensity at each tree encountered by the amount of trampling in the snow around the tree or shrub and categorized as (i) *walk past* where there is no evidence that caribou took any additional steps towards the tree, (ii) *stepped towards*, where caribou spent some time at the tree, but individual hoof prints could be seen (<25% of area around tree tracked), (iv) *moderate trample*, where 25-50% of area around tree is tracked, and (v) *heavy trample*, where >50% of the area around the tree is trampled and individual foot prints can not be distinguished.

At each tree encountered on the transects we recorded the following tree characteristics: (1) tree species; (2) cumulative distance from start of transect; (3) diameter at breast height (DBH) measured in centimetres; (4) Tree class²; (5) lichen abundance visually estimated using a standard ~10 g clump as a single unit and divided into two strata: (i) number of lichen clumps within approximate reach of caribou (0m to ~1.6 m above where we stood in the snow); (ii) number of lichen clumps not within reach of caribou under present snow-pack depths (1.6 m- 4.5 m above where we stood in the snow); (6) lichen genera composition estimated visually (*Alectoria sarmentosa/ Bryoria spp.*) using 10 percent intervals.

To describe the tree characteristics of the stand where caribou foraged, we completed six fixed area plots (0.01 ha each) along the forage transects. We completed three plots on the FPT at 50 m, 100 m and 150 m. and one plot at the end of each 50 m FAT. In each 0.01 ha plot we recorded the Dbh, wildlife tree class and species of all trees with a DBH greater than 5 cm to determine stand tree density, average DBH and species composition. We recorded conifers less than 5 cm DBH and shrubs separately as percent area coverage of the plot.

**Data Analysis**

We summarized tree and lichen characteristics along caribou forage transects by pooling data from each transect with all partial cut stands sampled. Transect summaries include the average and range (95% ² 1= live tree, 2= declining live tree, 3= new snag, 4 = loose bark snag, 5= clean snag, 6 – 8= clean, broken snag in various stages of decay, 9 = downed trees. Adapted from 'Describing Ecosystems in the Field', 2nd Edition.
confidence interval) of the number of live trees, live trees > 30cm Dbh and number of lichen 10-gram lichen clumps found above the caribou reach (>1.5 metres above the height of tracks in the snow as an estimate of pre-forage lichen availability) for each 50 metre length. We further summarized tree diameters and the number of 10gram lichen clumps (>1.5 metres) based on forage intensity on forage transects.
Results

We recorded caribou forage selection in 9 different partial cut stands during the 2004/2005 field season. These partial cut stands consisted of 3 older “IU” partial cuts and 6 single tree selection stands logged in the mid-1990’s (Table 2-1; maps showing the location of blocks and forage transects in Appendix 1).

Table 2-1. List of Partial Cuts where Forage Selection Assessed

<table>
<thead>
<tr>
<th>Block</th>
<th>Cut Type</th>
<th>Years Since Logging</th>
<th>Forest Cover Label</th>
<th>Date Assessed</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Opening 05</td>
<td>IU</td>
<td>26</td>
<td>BS 7221 – 7/15</td>
<td>11/09/2004</td>
<td>Heavy use – many days</td>
</tr>
<tr>
<td>CP-136-F</td>
<td>STS</td>
<td>9</td>
<td>BS 1100-11</td>
<td>11/09/2004</td>
<td>Moved through quickly</td>
</tr>
<tr>
<td>CP-118-A</td>
<td>STS</td>
<td>9</td>
<td>B(S) 9221-06/15</td>
<td>12/01/2004</td>
<td>Heavy use – at least 1 week – beds in block</td>
</tr>
<tr>
<td>CP-136-C</td>
<td>STS</td>
<td>9</td>
<td>NSR</td>
<td>02/19/2005</td>
<td>Several days in block</td>
</tr>
<tr>
<td>Opening 18</td>
<td>IU</td>
<td>27</td>
<td>S 1101-16</td>
<td>02/19/2005</td>
<td>Moved through quickly</td>
</tr>
<tr>
<td>CP-136-A</td>
<td>STS</td>
<td>9</td>
<td>BS 5300-18</td>
<td>02/19/2005</td>
<td>Moved through quickly</td>
</tr>
<tr>
<td>CP-154-C</td>
<td>STS</td>
<td>9</td>
<td>n/a</td>
<td>03/16/2005</td>
<td>Heavy use</td>
</tr>
<tr>
<td>Opening 07</td>
<td>IU</td>
<td>25</td>
<td>BS 4101-9</td>
<td>03/16/2005</td>
<td>Block has open areas – use confined to areas with large trees</td>
</tr>
<tr>
<td>Cp 118-D</td>
<td>STS</td>
<td>9</td>
<td>B 1100-15</td>
<td>03/16/2005</td>
<td>Very heavy use – almost every tree foraged on</td>
</tr>
</tbody>
</table>

Tree and lichen characteristics along forage and random transects

Caribou foraging path transects (FPTs) had a similar amount of live trees per 50 metres of transect as random, foraging area transects (FATs) mean (7.4 +/- 1.16 SE and 6.8 +/- 0.8 SE). However, FPTs had a greater average number of large trees (>30 cm Dbh) than FATs (2.03 +/- 0.34 SE on FPT to 1.07 +/- 0.19 SE FATS (Figure 2-1). Caribou trails also had a larger number of 10-gram lichen clumps (8.09 +/- 2.2 SE) compared to FATs (4.04 +/- 0.7 SE). The proportion of Bryoria spp. did not differ between FPTs and FATs (63.1 +/- 7.2 for FPTs and 66.4 +/- 8.3 for FATs).
Figure 2-1. The average number of total live trees and live trees greater than 30cm dbh encountered along 50metre- long caribou forage trails and random trails. Error bars indicate the 95% confidence intervals might want to replace these with ones I redid below, cuz the ones here I made more for the powerpoint slide.

**Tree and lichen characteristics at forage locations**

Caribou foraged more intensively on trees with greater average diameter (Figure 2-2). Trees where caribou did not forage (WP) and trees along random transects (X) also had lower average diameter compared to trees that were foraging did occur.
Figure 2-2. The distribution of tree diameters by caribou forage intensity. Caribou forage intensity increases from left to right; stepped towards (SI), light trampling (LT), moderate trampling (MT) and heavy trampling (HT). (WP) refers to trees caribou did not forage at on forage transects, while (X) refers to all trees on random transects.
Caribou forage more intensively on trees that had a higher number of 10-gram clumps of lichen above 1.6 metres. Trees where caribou did not forage (WP) and trees along random transects (X) also had a lower average number of 10 gram lichen clumps compared to trees where foraging did occur.

**Figure 2-3.** The number of 10-gram lichen clumps (>1.6 m from the ground) by caribou forage intensity. Caribou forage intensity increases from left to right; stepped towards (SI), light trampling (LT), moderate trampling (MT) and heavy trampling (HT). (WP) refers to trees caribou did not forage at on forage transects, while (X) refers to all trees on random transects.
Discussion

The efficacy of partial cuts to continue to provide suitable habitat for caribou after logging has been questioned since their inception (Stevenson et al. 1994, Coxson et al. 2003). While these results are preliminary, they do suggest partial cut stands do provide habitat value for caribou. In addition, the results suggest caribou make use of larger diameter trees within these stands that have greater lichen abundance. Partial cuts may also provide good visibility for detecting predators due to the more open stand conditions. Many of the partial cuts also provide both lichen and stand structural conditions similar to those of ESSF Parkland forests where caribou spend much of the late winter. Under these conditions *Bryoria spp.* lichen forage is more available lower in the canopy than many of the adjacent uncut old-growth forests. Several questions still remain regarding the ability of partial cuts to provide suitable habitat:

- What is the relative suitability of partial cuts compared to uncut old forest stands?
- How does suitability change over time?
- How does the amount of tree retention and effect suitability?
- How important are partial cuts within the landscape?
- How does the condition of surrounding forest stands impact the suitability of partial cuts as habitat?
- Does a landscape with a matrix of partial cuts surrounded by old uncut forests have more value than a landscape of old forest interspersed with small patch cuts?

References


Executive Summary

Mountain caribou in the Columbia Forest District were studied using radio-telemetry from 1992 to 1999. Censuses conducted in 1994, 1995, 1996, and 1997 indicated a relatively stable population of between 290 to 373 animals. In 2002, census efforts were coordinated throughout the range of mountain caribou in British Columbia and resulted in a population estimate of 211 animals for the Revelstoke area, or a decline to 64% of the 1997 estimate. This downward trend occurred in all mountain caribou populations in southern B.C. However, because there were only 5 radio-collared animals remaining in the area in 2002, the sightability of the population (proportion of the total population actually observed) could not be well estimated resulting in a wide confidence interval of 176-269 animals. The decline was confirmed during the 2004 census, with a population estimate of 176. The 90% confidence interval for the 2004 census was 167-210 animals, based on sighting 14 of the 15 radio-collars available.

Introduction

Woodland caribou (Rangifer tarandus caribou) in southeastern British Columbia are commonly referred to as mountain caribou. They are a unique ecotype of caribou distinguished from others by their use of the deep wet-belt snowpack (>2m) as a platform to access arboreal lichens in the canopy of subalpine forests during winter. Because of their low numbers, decreasing population trend, and shrinking geographic distribution, these caribou are considered threatened by COSEWIC (Committee on the Status of Endangered Wildlife in Canada) and, in the fall of 2002, were added to the red-list (species at risk of extinction or extirpation) by the British Columbia Conservation Data Centre.

Mountain caribou habitat is contiguous in British Columbia from north of Prince George to the Columbia Forest District (CFD). Four small, isolated populations are found south of the CFD with one extending below the international border. Population characteristics of caribou on the Revelstoke Reservoir side of the CFD were examined from 1981 to 1985 by Simpson and Woods (1987) and from 1992 to 1999 (McLellan and Flaa 1993, McLellan et al. 1994a, b, Flaa and McLellan 2001). Although more intensive research on this population ended in 1999, limited work continues to investigate the relationship of mountain caribou to snowmobiling (Hooge and McLellan 2001).

In the past, research efforts were subject to jurisdictional limitations such as forest district or regional boundaries. Delineation of populations was often based on
inadequate knowledge of caribou distribution and movements. These arbitrary geographic boundaries have made population comparisons among years difficult because animals could easily move in and out of census areas. Recently, Wittmer (2004) determined that there are four separate subpopulations (Columbia North, Columbia South, Kinbasket, and Frisby-Boulder-Queest) (Figure 1.) inhabiting the region north of the TransCanada Highway and bounded by the Adams River in the west and the Rockies in the east. Although animals were known to move between the Frisby-Boulder-Queest and the Columbia South subpopulations in the early 1980s (K. Simpson unpubl. data), there has been no known occurrences of movement between these areas since. Censuses are now conducted over the complete ranges of the populations of interest (Columbia North, Columbia South, Kinbasket, Frisby-Boulder-Queest and South Monashee). The population most affected by these changes in census coverage is the Columbia North population. It would have been underestimated in previous censuses since areas on the periphery of its range were outside the CFD, and therefore outside the scope of the original project.

Census Area

The CFD was created in 1996 by the amalgamation of the Golden and Revelstoke forest districts. Historically, caribou censuses were conducted in these two districts separately and sometimes not during the same year. Although we now know that caribou move between the two districts at their northern extremities, enumeration by the previous district boundaries will be included in this report to enable comparisons to previous reports.

Before 1993, caribou censuses on the Revelstoke side of the CFD did not cover the entire area and effort was inconsistent among years, reflecting inconsistent budgets. To ensure more complete coverage and systematic effort over time, the area was divided into 3 census blocks in 1993: Revelstoke North, Center, and South. The size of each block enabled coverage in one day, with boundaries selected to virtually eliminate the chance of inter-block movements during the census. Radio-tracking caribou in the area (summarised in Wittmer, 2004) has indicated that there are four subpopulations in the Revelstoke portion of the census area with little or no movement among them. These subpopulations vary greatly in both numbers and areal extent. The ranges of the subpopulations do not coincide with the census blocks. Although members of these subpopulations are usually within the Revelstoke portion of the CFD and would have been counted during censuses conducted in the 1990s, some animals in the Columbia North subpopulation move into the Rocky Mountains and some cross the height of land in the Monashees into the Okanagan-Shuswap Forest District. An additional subpopulation is found adjacent to the southern (from Stitt-Windy creeks south) portion of the Kinbasket Reservoir on the Golden side of the CFD. The Kinbasket subpopulation was censused in 1995, 1997, and 2002 and the Salmon Arm District was first censused in 2002.

Methods

Between 1992 and 2004, 96 caribou were captured by net-gunning from a helicopter in subalpine habitat in the CFD and adjacent portions of the Okanagan/Shuswap Forest District. All caribou were fitted with mortality/motion sensitive radiocollars. Caribou were censused during late winter when they were in open subalpine parkland
(Simpson and Woods 1987, McLellan et al. 1995, Seip 1992). Censuses in 1993, 1994, 1995, 1996, 1997, 2002 and 2004 were conducted shortly after new snowfalls using a Bell 206 helicopter with 3 experienced observers plus the pilot. A contour near treeline (1800–2130 m) was flown including all suitable habitat within the study area, and tracks were followed until animals were sighted unless the tracks descended into mature timber and were lost from view. An estimate of the number of caribou in these unsighted groups was based on separate tracks and/or beds. The location of each group was marked and numbered on a 1:250,000 map and recorded with a GPS. Although all male groups were easily identifiable, discrimination of young males from adult females in large groups was difficult and would have required additional harassment. We therefore limited our classes to calves and adults in these larger mixed groups.

During the census, collars were used as marks for mark-resight calculations only and not to locate animals. Population estimates were calculated using the program NOREMARK, using the maximum likelihood joint hypergeometric estimator for closed populations and 90% confidence intervals (White 1996).
Figure 1. Mountain caribou subpopulations in British Columbia as described by Wittmer (2004). Subpopulations in the CFD include CN – Columbia North, KS – Kinbasket South, FB = Frisby/Boulder/Queest, CS = Columbia South, MS = Monashee South.
Results
During the 1994, 1996, 1997, 2002 and 2004 censuses, 347, 268, 308, 176 and 165 caribou were observed and the populations in the Revelstoke side of the CFD estimated to be 373, 290, 331, 211 and 176 with upper 90% CI of 407, 316, 357, 269 and 211 respectively (Table 1, Figure 2). During all censuses, 91 of 99 radio-collared animals were seen for an average sightability of 92%.

Table 1. Revelstoke Reservoir portion of the Columbia Forest District caribou census results: 1994-2002.

<table>
<thead>
<tr>
<th>Year</th>
<th>Number Observed</th>
<th>Estimated from Tracks</th>
<th>Tracks plus Observed</th>
<th>Number Collared</th>
<th>Number Collared Observed</th>
<th>Calculated Estimate</th>
<th>90% CI</th>
</tr>
</thead>
<tbody>
<tr>
<td>1994</td>
<td>347</td>
<td>15</td>
<td>362</td>
<td>26</td>
<td>24</td>
<td>373</td>
<td>347-407</td>
</tr>
<tr>
<td>1996</td>
<td>268</td>
<td>39</td>
<td>307</td>
<td>25</td>
<td>23</td>
<td>290</td>
<td>268-316</td>
</tr>
<tr>
<td>1997</td>
<td>308</td>
<td>1</td>
<td>309</td>
<td>28</td>
<td>26</td>
<td>331</td>
<td>308-357</td>
</tr>
<tr>
<td>2002</td>
<td>176</td>
<td>12</td>
<td>188</td>
<td>5</td>
<td>4</td>
<td>211</td>
<td>176-269</td>
</tr>
<tr>
<td>2004</td>
<td>165</td>
<td>16</td>
<td>181</td>
<td>15</td>
<td>14</td>
<td>176</td>
<td>167-210</td>
</tr>
</tbody>
</table>

Because the intensive research on caribou in this area ended in 1999, only 5 radio-collars were active during the 2002 census resulting in a wide confidence interval in the NOREMARK estimate. For the 2004 census, there were 19 radiocollared caribou in the entire area and 18 were observed for a sightability of 95% (Table 1). In the south Monashees, (Blanket Creek area) 10 animals were observed in 1994, 4 in 2002, a maximum of 6 were seen during telemetry flights in 2003, and 7 during the census in 2004.
Figure 2. Trend of the caribou population inhabiting the Revelstoke portion of the Columbia Forest District, 1994 to 2004.
Table 2. Total number of caribou observed, estimates from tracks, and percent calves of the subpopulations in the Columbia Forest District. Includes NOREMARK estimates for populations with sufficient numbers of collared animals, 1994 to 2004.

<table>
<thead>
<tr>
<th></th>
<th>Observed (%calves)</th>
<th>Estimated from Tracks</th>
<th>Tracks plus Observe</th>
<th>Number Collared</th>
<th>Collars Observed</th>
<th>Calculated Estimate</th>
<th>90% CL</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Columbia South</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994</td>
<td>101 (12)</td>
<td>12</td>
<td>113</td>
<td>12</td>
<td>10</td>
<td>121</td>
<td>106-161</td>
</tr>
<tr>
<td>1996</td>
<td>75 (12)</td>
<td>19</td>
<td>94</td>
<td>10</td>
<td>9</td>
<td>83</td>
<td>76-108</td>
</tr>
<tr>
<td>1997</td>
<td>93 (14)</td>
<td>0</td>
<td>93</td>
<td>9</td>
<td>9</td>
<td>93</td>
<td>93-107</td>
</tr>
<tr>
<td>2002</td>
<td>20 (10)</td>
<td>5</td>
<td>25</td>
<td>3</td>
<td>2</td>
<td>30</td>
<td>21-86</td>
</tr>
<tr>
<td>2004</td>
<td>38 (16)</td>
<td>2</td>
<td>40</td>
<td>3</td>
<td>3</td>
<td>38</td>
<td>38-59</td>
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<tr>
<td><strong>Columbia North</strong></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1994</td>
<td>206 (19)</td>
<td>3</td>
<td>209</td>
<td>13</td>
<td>13</td>
<td>206</td>
<td>206-227</td>
</tr>
<tr>
<td>1996</td>
<td>167 (19)</td>
<td>26</td>
<td>193</td>
<td>11</td>
<td>11</td>
<td>167</td>
<td>167-188</td>
</tr>
<tr>
<td>1997</td>
<td>195 (13)</td>
<td>13</td>
<td>208</td>
<td>17</td>
<td>15</td>
<td>237</td>
<td>217-290</td>
</tr>
<tr>
<td>2002</td>
<td>145 (12)</td>
<td>7</td>
<td>152</td>
<td>1</td>
<td>1</td>
<td>145</td>
<td>145-171</td>
</tr>
<tr>
<td>2004</td>
<td>129 (14)</td>
<td>7</td>
<td>136</td>
<td>12</td>
<td>12</td>
<td>129</td>
<td>129-143</td>
</tr>
<tr>
<td><strong>Frisby</strong></td>
<td>Observed (%calves)</td>
<td>Estimated from Tracks</td>
<td>Tracks plus</td>
<td>Number Collared</td>
<td>Collars Observed</td>
<td>Calculated Estimate</td>
<td>90% CL</td>
</tr>
<tr>
<td>1994</td>
<td>36 (8)</td>
<td>0</td>
<td>36</td>
<td>2</td>
<td>1</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>1996</td>
<td>20 (25)</td>
<td>0</td>
<td>20</td>
<td>4</td>
<td>3</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>1997</td>
<td>30 (10)</td>
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<tr>
<td>2002</td>
<td>10 (10)</td>
<td>0</td>
<td>10</td>
<td>2</td>
<td>2</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2004</td>
<td>5 (20)</td>
<td>0</td>
<td>5</td>
<td>1</td>
<td>0</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td><strong>Kinbasket</strong></td>
<td>Observed (%calves)</td>
<td>Estimated from Tracks</td>
<td>Tracks plus</td>
<td>Number Collared</td>
<td>Collars Observed</td>
<td>Calculated Estimate</td>
<td>90% CL</td>
</tr>
<tr>
<td>1995</td>
<td>8 (13)</td>
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<td>18</td>
<td>3</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>1997</td>
<td>17 (0.06)</td>
<td>4</td>
<td>21</td>
<td>5</td>
<td>5</td>
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<td>NA</td>
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<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2004</td>
<td>0*</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

* one small group was observed in this area two weeks prior, but was missed during the census in 2004
Table 2. Cont.

<table>
<thead>
<tr>
<th>Boulder-Queest</th>
<th>Observed (%calves)</th>
<th>Estimated from Tracks</th>
<th>Tracks plus Observed</th>
<th>Number Collared</th>
<th>Collars Observed</th>
<th>Calculated Estimate</th>
<th>90% CL</th>
</tr>
</thead>
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<tr>
<td>2002</td>
<td>10 (40)</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>3</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2004</td>
<td>11 (36)</td>
<td>0</td>
<td>11</td>
<td>1</td>
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<td>NA</td>
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<table>
<thead>
<tr>
<th>Monashee</th>
<th>Observed (%calves)</th>
<th>Estimated from Tracks</th>
<th>Tracks plus Observed</th>
<th>Number Collared</th>
<th>Collars Observed</th>
<th>Calculated Estimate</th>
<th>90% CL</th>
</tr>
</thead>
<tbody>
<tr>
<td>1994</td>
<td>10 (30)</td>
<td>0</td>
<td>10</td>
<td>0</td>
<td>0</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2002</td>
<td>4 (0)</td>
<td>0</td>
<td>4</td>
<td>0</td>
<td>0</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>2004</td>
<td>7 (16)</td>
<td>0</td>
<td>7</td>
<td>2</td>
<td>2</td>
<td>NA</td>
<td>NA</td>
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Figure 3: NOREMARK estimates for two subpopulations in the Columbia Forest District.
Discussion

The census results indicate that the overall caribou population in the CFD remained relatively stable between 1994 and 1997 but by 2002 had decreased by 64% of what it was in 1997. If the population declined consistently over the 5 year interval, this would have been an annual decline of 8.6%, a rate that apparently continues to the estimate of 176 animals in 2004. Based on the 2004 census estimate, the number of caribou in this area had been reduced to 53% of the 1994 estimate. A similar decrease in population size was noted in the Central Selkirk where the number of caribou observed during the census in 2004 was only a third of the number observed in 1997 (Hamilton 2004).

Estimates from the 2002 census increased concern for the population in the CFD, however, without a larger sample of marked animals, the level of certainty in the estimate was low. The 2004 census confirmed the decline and suggests that the decline has continued. The subpopulations did not change at the same rate and the declines appear to have started earlier in some subpopulations than in others. The Columbia South population may have began to decline earlier in the 1990s than the Columbia North. In 2002, the number of animals or their tracks seen in the area used by the subpopulation centred on Frisby Ridge decreased to 28% of the number recorded in 1994 and have continued a gradual decline since. The Columbia North population has decrease by 36% since 1994. The proportion of the entire population that consisted of calves decreased from an average of 16.1% (1994) and 17.5% (1996) to 12.4 (1997) and 11.6 (2002) (Table 2). The proportion of calves was slightly higher in 2004.

Since 1992, the leading causes of death of caribou in the CFD have included predation and accidents including avalanches (Flaa and McLellan 2000) and these likely remain the significant factors that have influenced the population size. Dynamics in the large mammal predator-prey system, particularly changes in moose and deer numbers and related changes and time lags in changes in wolf and cougars may have adversely affected caribou. It is probable that cougar numbers increased with deer numbers in the mid-1990s until the deer population rapidly declined due to the deep-snow winter of 1996/1997. This dynamic of the predator/prey system, if true, may have influenced the earlier decline in the Columbia South population. This population has the largest potential for overlap with deer and cougars and would thus have been more likely to be affected by cougar predation.

Since the 2002 census, 7 radio-collared caribou have died – 6 were killed by predators: grizzly (2), cougar (1), wolf (1) and unknown (2). However, without monitoring mortality rates and causes of a sufficient sample of radio-collared caribou, we cannot be confident in a true change in predation rate. An increase in avalanche mortality due to unstable snow conditions and displacement from more subdued terrain by winter recreation activities may also have caused an increase in caribou mortality.

Caribou censuses conducted in the Revelstoke portion of the CFD between 1983 and 1985 indicated a population size of 200 to 261 (Simpson and Woods 1987). During this period > 20% of the population was calves, suggesting an increasing population. Although the censuses in the 1980s may not have been as thorough as recent censuses, the presence of 9 marked animals enabled correction for reduced
sightability. It is possible that the caribou population increased from between 200 and 261 animals in the early 1980s to the estimated 373 animals in 1994 and has now decreased to below 200 animals. The apparent decline in population size without a clear indication of factors responsible indicates that the monitoring that ended in 1999 should be resumed.

**Literature Cited**


Mountain Caribou Habitat Use in the Okanagan Shuswap Forest District:

April 2004- March 2005

Preliminary Report #3

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Introduction

The Okanagan Shuswap Land and Resource Management Plan (OSLRMP)\(^5\) established a Mountain Caribou Resource Management Zone (RMZ) (Figure 1) that encompasses Mountain Caribou Habitat in the Plan area. The LRMP plan area is made up of the old Salmon Arm, Penticton, and Vernon Forest Districts that were amalgamated into the new Okanagan Shuswap Forest District in 2003. Specific objectives and strategies within the LRMP define how Crown resources in the RMZ are to be managed. The goal of management in the RMZ is to provide adequate habitat to maintain viable Mountain Caribou populations as well as to maintain habitat connectivity. The Okanagan Innovative Forestry Society (OIFS) contracted a caribou study in response to the OSLRMP call for further information on how to integrate caribou habitat needs with intensive forest management. This report documents progress of the study in the fourth year of a seven-year project. Progress in the first two years was summarized in a progress report produced in 2003 (Furk et. al 2003) and third year progress was summarized in a 2004 report (Furk et. al 2004).

The OIFS Forestry Plan (2001) identifies five main objectives for this project:

1. Identifying the caribou habitat requirements in the Interior Cedar Hemlock (ICH) and Engelmann Spruce Subalpine Fir (ESSF) zones;
2. Examining the relationship between forest management activities and relative caribou habitat use in the ICH and ESSF zones;
3. Identifying opportunities to supply suitable caribou habitat attributes through forest management activities;

\(^{5}\) website: [http://srmwww.gov.bc.ca/sir/lrmp/okan/](http://srmwww.gov.bc.ca/sir/lrmp/okan/)
4. Investigating caribou population dynamics, including predation and mortality;
5. Link monitoring from this project to ongoing caribou research projects in the Columbia and Clearwater Forest Districts;
6. Suggest areas for deployment of caribou old-growth retention zones in locations that are most likely to benefit caribou.

Caribou research projects in the Columbia and Headwaters Forest districts have also identified caribou movements in the former Salmon Arm Forest District area (now part of Okanagan Shuswap Forest District). Efforts are currently ongoing to track moose and wolf habitat use in the Headwaters Forest District and Columbia Forest Districts. This information is often relevant to habitat management in the Okanagan Shuswap Forest District and is summarised in the background section. Data from all three studies will be combined to complete the objectives of this study upon completion of the seven-year project.
Figure 1. Okanagan Shuswap Land and Resource Management Plan designated Mountain caribou resource management zone.
Background

Status and ecology

Mountain caribou are an ecotype\(^6\) of woodland caribou (*Rangifer tarandus caribou*) that, except for a small population in Idaho, are found exclusively in south-eastern and east-central British Columbia. Mountain caribou are characterized by their use of high elevation late winter habitat where they forage almost exclusively on arboreal lichens (Mountain Caribou Technical Advisory Committee [MCTAC] 2002). During early winter mountain caribou use lower elevation, older closed canopy forests that provide snow interception and access to forage that includes shrubs, arboreal lichen, and lichen litter fall. During late winter, mountain caribou move to higher elevation ESSF and sub-alpine forests where they feed primarily on arboreal lichens (Simpson et al. 1997, Stevenson et al. 2001).

During spring, some caribou remain at high elevations but many move to low elevations where they access snow-free foraging areas. Female caribou move to the subalpine or alpine to avoid predators while giving birth. In addition to requirements for forage and cover, caribou may distribute themselves over large areas to reduce efficiency of predators. Habitat alteration that results in increased predator numbers and access to caribou may contribute to population decline.

Caribou were historically more widely distributed than they are today and appear to have declined in all portions of their range (MCTAC 2002). In 2000, mountain caribou were placed on the “red” or endangered list of the B.C. Conservation Data Centre (CDC) indicating that populations within the province are at risk of becoming endangered. In May 2002 the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) confirmed that caribou in the southern mountain ecoprovince were threatened\(^7\).

Mountain Caribou in the Caribou RMZ

The Okanagan Shuswap Land and Resource Management (OSLRMP) area contains portions of three local caribou populations (Simpson et. al 1997). These are the Wells Gray South, Revelstoke, and Monashee populations (Figure 2).

There is movement between the Revelstoke and Wells Gray South populations. An analysis of province wide caribou census results (MCTAC 2002) indicated the Wells Gray South population long term\(^8\), short term and current

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\(^6\) the ecotype division is based on differences in habitat use, behaviour and migration patterns (MCTAC 2002). All caribou in BC belong to the same subspecies of caribou known as woodland caribou.

\(^7\) For more background on the status of Mountain Caribou see: A Strategy for the Recovery of Mountain Caribou in British Columbia VERSION 1.0 SEPTEMBER, 2002 Prepared by The Mountain Caribou Technical Advisory Committee Recovery Team for the Arboreal Lichen–Winter Feeding Ecotype of Woodland Caribou (*Rangifer tarandus caribou*) within the Southern Mountains National Ecological Area

\(^8\) long-term trend (>20% change in 20 years); short-term trend (>20% change in 7 years); current trend (>10% change in past 2 years),
trend stable. However, the reliability of data was low. The current and short-term trend for the Revelstoke population indicates a decline of about 8.6% per year since 1994; and this estimate is based on reliable data (Hooge et al. 2004, MCTAC 2002). The Monashee population trend is estimated to be declining in the short, current, and long term. However, reliability of data is considered low (MCTAC 2002). Numbers appear relatively stable since 1994 at 5-10 animals (Hooge et al. 2004).

Figure 2. Distribution of 13 local populations of Mountain Caribou from Simpson et.al 1997.

A recent analysis of extensive province-wide telemetry data (including data from this project) by Wittmer et al. (2004) indicates that there are three sub-populations that range in the Okanagan Shuswap Forest District area (Figure 3). They are the Columbia North, the Frisby-Boulder-Queest, and the Monashee subpopulations. Annual or lifetime home ranges of animals radio-collared within each subpopulation overlap extensively with each other but there is no overlap of animals among the subpopulations. Subpopulation boundaries are delineated using a 90% adaptive kernel estimator of all radiolocations or all animals in the sub-population.
Summary of progress of Caribou research in the Columbia and Headwaters Forest Districts 2004-2005

Research projects in the adjacent Headwaters and Columbia Forest Districts are conducted in conjunction with research in the Okanagan Shuswap Forest District. A research team based primarily in Revelstoke that includes Bruce McLellan (MOF), Robert Serrouya (Serrouya and Associates), Kelsey Furk, Doug Lewis, and Gary Pavan (independent contractors), Shannon Stotyn (M. Sc. Candidate- University of Alberta), and John Flaa (Parks Canada) manages the project on the ground in the Columbia and Okanagan Shuswap Forest Districts. This team works in conjunction with MWLAP Kamloops personnel on the North Thompson (Headwaters Forest District) project.

Caribou have been tracked using radio-collars in the Columbia Forest District since 1992. However, the research effort (and the number of collared caribou being monitored) was reduced in the late 1990’s. After a census in 2002 found a major decline in caribou numbers throughout most of southeastern B.C., the caribou research in the Columbia Forest District was expanded and now includes research on wolves and moose in addition to caribou. These projects are considered as part of the Revelstoke Predator-Prey Project.

Telemetry flights in all districts for these studies are coordinated in order to minimize flight times and maximize efficiencies associated with data administration. These projects have been funded largely through the Forest Science Program and the Forest Investment Account.

The objectives of the Revelstoke Predator-Prey Project are to:

1) collect information that describes the number and distribution of wolves within the study area;
2) assess and compare seasonal variation of wolf overlap with caribou and moose;
3) examine foraging behaviour (kill rate, diet) of wolves within the study area.

During the course of the Project (commenced in 2003), twelve wolves in five packs have had collars placed on them in the Columbia Forest District. It has been possible to estimate pack sizes and home ranges of the collared wolf packs through bi-weekly telemetry flights and GPS collar locations. The study also monitored sixty-four radio collared caribou and 35 moose with bi-weekly telemetry flights. The GPS collar data were downloaded and have helped to address the second question of seasonal spatial overlap. To address the third objective, one season of summer kill site investigations using GPS cluster analysis and a second season of winter wolf tracking have been completed. A total of forty-one kill sites and six scavenge sites were located and it was possible to identify species, sex, and body condition of most prey. Prey included 38 moose, 1 beaver, 1 deer, and 7 unknown ungulates (unidentifiable from the evidence remaining at the kill site).
Caribou were tracked in early winter 2004 through mid-seral and partial cut stands as well as fragmented landscapes in the Headwaters and Columbia Forest Districts. Data collected during trailing are being used in conjunction with a substantial data set of old-growth caribou trailing data already collected in the Columbia and Clearwater Forest Districts in order analyse habitat selection and foraging habits at a variety of scales in fragmented landscapes.

There were thirteen caribou tracked using fixed-wing flights during 2004 in the Columbia Forest District in addition to eight caribou in the Kirbyville Lakes area that use the Okanagan-Shuswap Forest District. The thirteen caribou were distributed throughout the Columbia Forest District from Foster Creek in the north (Headwaters Forest District) to Glacier National Park and Frisby Ridge in the south and the Cummins River in the east. There were three radio-collared caribou mortalities in the Columbia Forest District between March 2004 and March 2005. Two collared caribou from the Columbia North subpopulation died in summer 2004; one was killed by a cougar at Cummins River, and the other died of unknown causes (possible grizzly bear) at Goosegrass Creek. A grizzly bear killed a caribou in Standard Basin (Columbia South subpopulation) in summer 2004. Also of interest was the movement of a female adult caribou from the Columbia North subpopulation to the Groundhog subpopulation area in summer 2004. This is the first recorded movement of a radio-collared animal between these two areas.

Caribou have been studied in the North Thompson area since 1996. Caribou continue to be monitored in the North Thompson area (Headwaters Forest District) in order to gain further information on causes of mortality and to facilitate effective caribou censuses. Collaring caribou in this area also facilitates research investigating habitat use in fragmented landscapes. Six radio-collared caribou died in the Headwaters Forest District between March 2004 and March 2005. Three were killed in the calving and summer season of 2004. The mortality causes were wolverine (Wells Gray Park), poor condition (possibly age related in the upper Adams River) and one was unknown (Wells Gray Park). Three caribou died during winter 2004-2005. The mortality causes were wolverine (Raft River), avalanche (Wells Gray Park) and an accident (slip and fall – age and condition related) (upper Adams River).

In order to determine the distribution, movements and habitat use of wolves in relation to caribou habitat use, an effort was made to capture wolves in the Headwaters Forest District in 2004. Work was conducted by personnel at the Kamloops Ministry of Water, Land and Air Protection (John Surgenor and Kurt Kier). One wolf was captured in the Adams River in April 2004. The collar worked inconsistently. The wolf was in a pack of approximately four animals (based on sightings during flights) and was located above Vavenby around Otter Creek, in the Adams River, in the Cayenne Creek valley and above Humamilt Lake. This wolf was legally trapped in February 2005.

**Study Area**

The study area is located in the Monashee Mountains surrounding the Shuswap Lake area. Major watersheds in the area include the Seymour River and the Eagle River watersheds. The study area falls in the Northern Columbia Mountains Ecoregion. Biogeoclimatic zones in the study area include the Interior
Cedar Hemlock Zone (ICH), the Engelmann Spruce – Subalpine Fir zone (ESSF) and the Alpine Tundra Zone (AT).

Methods

Capture and radio-telemetry

This year’s work continued on the progress made in the first three years of the project. Capture activities were conducted in conjunction with efforts in the Columbia and Headwaters Forest Districts as part of ongoing caribou projects. Known late winter caribou range was searched in March 2005, in areas where there were few operating radio-collars. During this period, caribou use sub-alpine forests and their tracks can be easily seen. When caribou were located, one or more (depending on group size and history of data from the area) individuals were captured using a net gun fired from a helicopter (Bighorn Helicopters) and handled by a crew that specializes in animal capture and MOF personnel. Live Animal Capture and Handling Guidelines for Wild Mammals, Birds, Amphibians & Reptiles (Resources Inventory Committee 1998) were followed during these operations. Caribou calves were outfitted with conventional (Telonics, Mesa Arizona) Very High Frequency (VHF) expanding collars. Adults were fitted with conventional VHF collars (Lotek Wireless Inc.) or GPS collars (Lotek Wireless Inc. and Advanced Telemetry Systems Inc.).

A monitoring program using fixed wing aircraft with radio tracking equipment recorded the location of all collared animals on a regular schedule using an on-board GPS system. Standards outlined in Wildlife Radio Telemetry (Resources Inventory Committee 1998) were followed during these operations. Flights were weather dependent, but were attempted every ten days. Telemetry flights were conducted in conjunction with caribou telemetry monitoring in the Clearwater and Columbia Forest Districts.

Each VHF location was checked in a Geographic Information System (GIS) to ensure locations were correct prior to being entered into the database. During flights a visual confirmation of animal location was attempted and if seen, the number of caribou including calves (when possible to distinguish) in the group were recorded. For all re-locations, the habitat type of the caribou location was recorded as well as the general location. The location habitat types are as follows:

<table>
<thead>
<tr>
<th>1. Alpine</th>
<th>8. slide-path</th>
</tr>
</thead>
<tbody>
<tr>
<td>2. sub-alpine parkland</td>
<td>9. immature forest (class 2-3)</td>
</tr>
<tr>
<td>3. meadow</td>
<td>10. riparian wetland</td>
</tr>
<tr>
<td>4. ESSF</td>
<td>11. cedar-spruce</td>
</tr>
<tr>
<td>5. clearcuts (&lt;20 years old)</td>
<td>12. rock, ice, lake</td>
</tr>
<tr>
<td>6. burns (&lt;30 years old)</td>
<td>13. semi-mature forest (class 4-5)</td>
</tr>
<tr>
<td>7. cedar-hemlock</td>
<td>14. riparian parkland</td>
</tr>
</tbody>
</table>

All collars and ear tags carry mortality/motion sensors that trigger a change of VHF signal beat rate when the tag or collar is motionless over a period...
of time (usually 6 hours). This change in signal can be detected during telemetry flights. When a change in signal was detected, the site was investigated as soon as possible. Often a helicopter was used to access remote and otherwise inaccessible locations. The location, condition, and position of the carcass, as well as any predator sign, and evidence of bleeding, were used to determine the cause of death. If present, long bones and tissue samples are collected in order to determine condition of the animal at the time of death.

During telemetry flights, snowmobile and ski activity was recorded when it was observed in close proximity to radio-collared caribou. Snowmobile tracking activity was classed as low (1-2 snowmobiles), moderate (3-10 snowmobiles) and high (10 or more snowmobiles or the snow in the area was extensively packed by tracks). Where present the type of skiing was recorded (heli-skiing or ski-touring).

Results

Capture and radio-telemetry

In March 2005, four caribou were captured and fitted with collars in the following general areas: two in Seymour/Kirbyville pass, one at Queest Mountain, and one at Blanket Glacier. Two female adult caribou were fitted with Lotek Wireless Inc. VHF collars, one female adult caribou was fitted with a Lotek Wireless Inc. GPS collar at Queest Mountain and one female caribou was fitted with a Telonics Inc. VHF collar.

By the time capture efforts were completed at the end of March 2005, there was one collared caribou at Queest Mountain, one collar at Frisby Ridge, seven in the Seymour River area (includes Ratchford, Kirbyville, and Cayenne Creek) and two collars at Blanket Creek. One of these collars is part of the Headwaters Forest District caribou project (at Cayenne Creek).

No collared caribou mortalities were recorded in the Okanagan-Shuswap Forest District or the adjacent areas. Mortalities recorded in 2004-2005 in the Columbia and Headwaters Forest District are summarized in the background section.

A total of 198 VHF locations were recorded from 10 caribou between March 2004 and March 2005. Caribou that were tracked over the whole year were located 19-21 times. An updated shapefile of VHF locations in ArcView shapefile format (BC Albers Projection) is included with this report.

In addition to VHF locations, four GPS collars with locations in the Okanagan-Shuswap Forest District were removed and downloaded in March 2005. These collars were deployed in the Blanket Creek, Queest, Frisby Ridge and Seymour River areas in spring 2003. These locations are in separate files submitted with this report in plot-file format and in ArcView shapefile format (in BC Albers Projection). In total, these four collars recorded 5844 locations in total and 2238 of these locations were recorded in the Okanagan-Shuswap Forest District. Success rates varied between collars and seasons (Figure 4, Table 1).
Figure 4 Successful fix rate by season of four Televilt GPS collars (identified by their Caribou ID in the right hand corner) deployed between March 2003 and March 2005.

Table 1 Successful fix rate (the number of attempted GPS fixes that successfully recorded a location) of Televilt collars deployed between March 2003 and March 2005 with locations in the Okanagan Shuswap Forest District.

<table>
<thead>
<tr>
<th>Caribou ID</th>
<th>Location</th>
<th>% of total fixes that were successful</th>
</tr>
</thead>
<tbody>
<tr>
<td>38</td>
<td>Seymour</td>
<td>47.7</td>
</tr>
<tr>
<td>64</td>
<td>Queest</td>
<td>71.3</td>
</tr>
<tr>
<td>71</td>
<td>Frisby</td>
<td>46.6</td>
</tr>
<tr>
<td>76</td>
<td>Blanket</td>
<td>35.0</td>
</tr>
</tbody>
</table>

Between March 2004 and March 2005 there were two collared caribou in the Monashee subpopulation (Wittmer 2004), two collared caribou in the Frisby-Boulder subpopulation, and seven collared caribou in the Ratchford/Seymour area (these are part of the Columbia North subpopulation).

Collared caribou in the Monashee subpopulation migrated south to Vigue Creek for part of the summer, spent the early winter around Joss Mountain and calving season, summer, and late winter at the headwaters of Blanket Creek (Figure 5). The largest number of caribou seen in this group during telemetry flights over winter 2004-05 was five, and no calves were seen in this group over winter. The census in 2004 found a total of seven caribou in two groups and included one calf.

The two collared caribou in the Frisby-Boulder subpopulation each have distinct seasonal movement patterns. One of the caribou wintered at Queest Mountain and spent the summer in the headwaters of Kirkup Creek and Crazy Creek. The other spent the winter on Frisby Ridge and summer in the Copeland...
Creek/Bews Creek area. Both of these caribou wore GPS collars that were recently downloaded (Figure 6).

Caribou in the Seymour River area spent the summer, spring and calving seasons in the Upper Seymour River, Kirbyville Lakes, and Bischoff Lakes. Early winter locations were recorded around Mt. Grace, Cayenne Creek, and Seymour River. Late winter locations were recorded at the north end of the Anstey Range, above Kirbyville Lake and at Bischoff Lakes. One caribou in this group carried a GPS collar that was recently downloaded and is presented along with 2004 VHF locations in Figure 7. The group that spent winter in the Anstey Range (south of Ratchford Creek) regularly included more than 20 individuals including several calves.

Snowmobile activity (based on tracking observed during flights) was regularly moderate to high intensity at Queest and North Queest Mountain. There was also moderate ski-touring use recorded in the North Queest area during some flights. Most late winter caribou locations and tracks observed during flights were recorded along ridges and bowls that generally received little or no snowmobile use. Caribou that spent winter at Queest Mountain migrated to Boulder Mountain in early spring, 2005. Snowmobile intensity in this area is generally high. No snowmobile activity was recorded in the North Anstey Range. Occasional light to moderate (tracks from an estimated 2-10 snowmobiles) snowmobile use was recorded in the Mt. Grace, Kirbyville Lakes and Bischoff Lakes area. There was occasional tracking recorded in the closure area within the Columbia Forest District above Kirbyville lakes.

Figure 5 VHF locations of 2 collared caribou in the Monashee suppopulation (Wittmer 2004). VHF locations recorded between March 2004 and March 2005. One of these caribou had a GPS collar. The GPS locations were recorded between March 2003 and March 2005.

Figure 6 VHF and GPS locations for two collared caribou in the Frisby-Boulder subpopulation (Wittmer 2004). VHF locations recorded between March 2004 and March 2005. The GPS locations were recorded between March 2003 and March 2005.

Figure 7 VHF and GPS locations for seven collared caribou in the Columbia North subpopulation (Wittmer 2004). VHF locations recorded between March 2004 and March 2005. The GPS locations were recorded between March 2003 and March 2005.
Discussion

The increasing number of VHF locations of caribou plus the GPS data downloaded this year provides an opportunity for a more detailed analysis of habitat use as well as investigating the implications of young forests on caribou movements. These data will also help facilitate research on caribou habitat use in areas where forest management and backcountry recreation occupy much of the landscape.

Based on sightings during flights, caribou in the Kirbyville-Seymour-Ratchford area appear to be in larger groups and with more calves than caribou in the Frisby, Queest Mountain or Blanket Creek areas. These observations are consistent with previous census results (Hooge et. al 2004). The wintering group of caribou in the Anstey Range is the largest single group of caribou located during telemetry flights in the entire study area spanning the three forest districts. This area currently receives no heli-skiing or snowmobile use. Due to the relatively large group and numerous calves in this area, access management should be a priority and efforts should be made to minimize any type of recreational use during winter.

The Monashee subpopulation continues to exist at low numbers, with five caribou sighted during telemetry flights this winter. These flights, however, are not a complete census, since it is possible another group or groups were elsewhere during winter. However, it is notable that no calves were sighted in the group seen regularly this winter. During early winter, caribou in this group use the Joss Mountain area, which is outside the current RMZ boundary. This area should be added to the caribou RMZ. Caribou in this group migrated south to the Vigue Creek area during summer for the second year in a row, confirming use of the RMZ in this area. Anecdotal sightings are also occasionally reported in the Monashee Park area during summer.

No reports of caribou in the Pukeashun Mountain area were received, however only one over-flight of this area was conducted in early winter 2004. The small wolf pack collared in the Adams River in March 2004, was rarely located, but based on a few sightings, may have migrated between the Humamilt Lake-Momich Lake Corridor and the Adams River via Cayenne Creek. There is ample late winter caribou habitat in the Pukeashun area (Serrouya and Furk 2004), but large-scale factors related to fragmentation and predator abundance around the Pukeashun area likely limit the abundance of caribou in the Pukeashun area. Efforts to retain habitat, minimize fragmentation and resource use should continue, in order to provide the opportunity for caribou to re-colonize the area.

Snowmobiling continues to be a concern in the range of the Frisby-Boulder subpopulation and may be displacing some caribou to areas inaccessible to snowmobilers in the Queest Mt. area. A large portion of the winter range of this subpopulation is subject to moderate or high intensity snowmobile activity. There is currently no closure area set aside for caribou in the Queest Mountain area. A closure area in the North Queest area specifically may be beneficial to caribou in this area.
Snowmobile activity in the Mt. Grace – Kirbyville area appeared to be higher (more tracks were sighted more frequently during flights) this winter than last. There is now a cabin located above Seymour River that may facilitate snowmobile access to this area. This area contains a large portion of the late winter range of the Columbia North subpopulation and caribou in this area have not been previously exposed to snowmobile activity. Because these caribou have not had an opportunity to gradually habituate to snowmobiles and snowmobile activity may increase rapidly as it has in other areas such as the Keystone/Standard Basin area in the Columbia Forest District, and much of Mount Grace is wide, plateau habitat, there is potential that future snowmobile activity could be detrimental to the large numbers of caribou that use this area. The snowmobile terrain in this area appears to be of high quality and there are currently no snowmobile closure areas on the Seymour River side of Monashee height of land. The closure on the Lake Revelstoke side of the height of land is occasionally being violated and is extremely difficult to enforce due to the remoteness of this area.

Preliminary efforts were made this year to communicate the recent locations of collared caribou to two local heli-ski companies. Recent location information helps to inform decisions and will help to minimize interaction between helicopters, skiers and caribou.

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The predator-prey dynamics of wolves and moose in the northern Columbia Mountains: spatial and functional patterns in relation to mountain caribou decline.

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Summary

Considerable effort has been made to quantify prey selection by wolves and to estimate kill rates to determine the effect that wolves have on ungulate populations, and therefore guide management decisions and direct conservation efforts. However, the majority of wolf-foraging studies occur in winter, when snow tracking enables researchers to more easily find kills from the air or on the ground. Investigating wolf prey selection and kill rates in other seasons may be important as studies have shown shifts in diet composition, greater prey diversity, and higher predation rates when compared to winter. Furthermore, there is the greatest potential for overlap between moose (the wolf’s primary prey in the Columbia Mountains) and endangered mountain caribou during non-winter seasons. As a preliminary study, we used the GPS location data (May - August 2004) to determine the feasibility of quantifying summer predation patterns from GPS locations. Potential kill sites were identified from GPS clusters having $\geq 2$ points that were <200 m apart. Of the 25 GPS clusters sites investigated, we found evidence of a large mammal predation event at 40% of the sites. The probability of a large mammal predation event being present at a GPS cluster site increased with the number of days a radio-collared wolf spent at the cluster and decreased if the cluster had been revisited over a number of non-consecutive days. Developing techniques that link GPS collar technology and animal behaviour could be important to increase our understanding of complex systems for the benefit of conservation and management efforts.

Although wolf foraging behaviour can be explained using kill rates and prey selection, landscape patterns can have important effects on the foraging efficiency of predators and can influence anti-predator strategies of prey. We compared locations of moose killed by wolves in winter to telemetry locations of live moose during winter, to identify landscape patterns that were associated with locations where wolves were likely to kill moose. Relative to live moose, killed moose were found >250 m further from a major water source (i.e. Goldstream River or Downie Creek). Dead moose were also at higher elevations, 30% closer to a seral edge, and in areas that contained 30% more logged forest, relative to live moose. However, variability was high, as evidenced by the large standard error of parameter estimates, and the winning multivariate model was the intercept-only (null) model. Nonetheless, the analysis identified some unique features of the study area, such as the probable use of open water as a refuge for moose. In other study areas during winter, most water bodies are frozen, which often facilitates foraging efficiency for wolves but does not provide an escape feature for moose. Ultimately our work on moose predation risk will be expanded to include non-winter seasons and contrasted with the caribou predation risk model being developed by other members of our research team (C. Apps, H. Wittmer, B. McLellan), to identify zones of shared predation risk between moose and caribou.
Introduction

Woodland caribou within the Southern Mountains National Ecological Area, that includes mountain caribou, are listed as Threatened (Committee on the Status of Endangered Wildlife in Canada), and Threatened or Endangered (Conservation Data Centre Red List) in British Columbia (B.C.; Mountain Caribou Technical Advisory Committee 2002). In a recent meta-analysis of mountain caribou populations in B.C., predation was identified as the proximate cause of this decline (Wittmer et al. 2004;). This increased level of predation is linked to increasing amounts of young forests and fragmented habitat (Wittmer and McLellan, submitted).

Although wolf predation is not the main mortality factor implicated in the recent decline of the mountain caribou population near Revelstoke, B.C., recent increases in moose and wolf populations has led to concerns that encounters with mountain caribou will increase, leading to higher predation rates. A 2003 moose census documented a 100% increase (0.7 moose/km$^2$ to 1.54 moose/ km$^2$) in moose densities over the past decade (Poole and Serrouya 2003). This increase has been attributed to a higher proportion of young seral stage forest on the landscape (Serrouya and D’Eon 2002), and possibly milder winters that facilitate overwinter survival. Since there has likely been a numerical response of wolves to the increased moose density, there may be a negative effect on the long-term persistence of mountain caribou populations in the region (Bergerud and Elliott 1998; Seip 1992). If management strategies for the recovery and maintenance of caribou herds are to be effective, a greater understanding of this complex predator-prey system in conjunction with the changing landscape is necessary.

Considerable effort has been made to quantify prey selection by wolves and estimate kill rates to determine the effect that wolves have on ungulate populations, guide management decisions, and direct conservation efforts (Huggard 1993; Hayes et al. 2000; Hebblewhite and Pletscher 2002; Kunkel et al. 2004; Smith et al. 2004). However, most of this research has been focused on the winter season, due to the logistical problems of wolf tracking in snow-free seasons. Investigating wolf prey selection and kill rates in other seasons may be important as past studies have shown shifts in diet composition, (Darimont and Reimchen 2002) greater prey diversity in the diet (Jedrzejewski et al. 2002) and higher predation rates (Sand et al. 2003) when compared to winter seasons. Furthermore, the potential for wolves to overlap with caribou range is greatest during the non-winter seasons, and more than 60% of caribou predation mortality occurs in the non-winter seasons (Wittmer et al., in press).

Summer kill rates and prey selection are traditionally investigated using intensive radio-tracking and scat analysis (Jedrzejewski et al. 2002). These techniques are costly, time intensive, and difficult in remote areas and potentially dangerous in areas with grizzly bear (Ursus arctos) populations. Recent advances in GPS radio-tracking technology have expanded the techniques used to determine prey selection and kill rates of carnivores. Using a combination of
intensive radio tracking and GPS location data, Sand et al. (2003) identified the summer predation patterns of Scandinavian wolves. Anderson and Lindzey (2003) estimated cougar predation rates from GPS clusters and developed a model to estimate predation events using GPS data.

Once the effects of predator foraging behavior on ungulate populations have been clarified, wildlife managers can attempt to manipulate the system using harvest management (and potentially other tools) to try and favour the threatened species. However, ecologists have argued that the spatial patterns of predator foraging can help explain the mechanisms behind prey selection and kill rates (Lima and Dill 1990). Several studies have linked landscape attributes to the location of prey kills sites (Kunkel and Pletcher 2000, Thogmartin 2000, Johnson et al. 2002) or predator/prey encounter and kill sites (Hebblewhite et al. in review). The results of these studies can help guide land-management decisions because some landscape features, such as the spatial configuration, distribution, and intensity of forest harvesting, road building, and other resource uses, can be manipulated by humans.

Our ultimate goal is to clarify wolf foraging patterns in the northern Columbia Mountains. More specifically, we had 3 objectives:

1) Determine the amount spatial overlap between caribou, wolves, and moose;
2) Determine the prey selection and foraging rates of wolves, and determine the feasibility of using GPS collars to quantify these parameters;
3) Describe stand and landscape-level factors that affect the foraging patterns of wolves on moose.

In this report we present the preliminary results of these objectives.

Methods

Study Area

The study area is in the Northern Columbia Mountain ecoregion in southeastern British Columbia (Demarchi 1996). The study area is bounded by Encampment Creek in the north and the town of Revelstoke in the south. The eastern and western boundaries extend to the heights of land surrounding the Lake Revelstoke reservoir. Elevations range from 550 m to 3,050 m. The study area receives 946 mm of precipitation annually (425 cm of snow falls annually).

The study area is composed of a mosaic of forests, regenerating clearcuts, riparian forests, shrublands, upper elevation basins, and avalanche chutes. Biogeoclimatic subzones within the study area range from Interior Cedar-Hemlock (variants ICHwk, ICHvk1, ICHmw3) in the valley bottoms and mid-elevations, to Engelmann Spruce-Subalpine Fir (variant ESSFvc) at approximately 1,280 m to 1,400 m (Braumandl and Curran 1992). The Alpine Tundra subzone occurs at elevations above the ESSF.
The ICH subzones are dominated by forests of western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*) and less commonly, Douglas-fir (*Pseudotsuga menziesii*). The ESSF zone is comprised of coniferous forests of primarily Engleman Spruce (*Picea Engelmannii*) and subalpine fir (*Abies lasiocarpa*). Alpine meadows, shrublands, snowfields, glaciers and rock dominate the Alpine Tundra zone.

Moose are the most abundant ungulate species, with mule deer (*Odocoileus hemionus*) and white-tailed deer (*O. virginianus*) occurring at lower densities in the southern portion of the study area (Poole and Serrouya 2003). Mountain goats (*Oreamnos americanus*) and mountain caribou are also found in the study area. Grizzly bears, black bears (*Ursus americanus*), wolverine (*Gulo gulo*), wolves and cougars (*Felis concolor*) also occur.

**Capture and Handling**

Wolves were captured from 2003 to 2005 using modified foothold traps (Livestock Protection Co., Alpine Tex.) in the summer and helicopter net-gunning (Bighorn Helicopters Inc., Cranbrook, B.C.) in the winter. In 2004, two wolves were captured using modified foothold traps and five wolves were captured using a netgun fired from a helicopter. These wolves were affixed with 5 VHF LMRT-3 and 2 GPS 3300s collars (University of Alberta-Protocol 2004-09D; Lotek Inc., Newmarket, Ontario). GPS collars were programmed to attempt position acquisitions every 3 hours. By February 2005, no wolves remained radio-collared, because of collar failure, dropped collars, possible dispersal, and death from trapping and hunting. In March 2005, 6 more wolves were captured in the study area and affixed with 2 VHF LRMT-3 collars, 1 Lotek GPS 3300s collar, and 3 remotely downloadable GPS collars (HABIT, Victoria, B.C.).

**Spatial Overlap of Moose, Wolves, and Caribou**

Ultimately our goal is to use resource selection functions (RSF) to quantify the spatial overlap among these 3 species. We plan to follow methods similar to Johnson et al. (2000), where they used RSF functions to quantify the degree of spatial separation of mule deer and elk. They compared the direction of parameter estimates for the same habitat variables for each species, and used the predictive equation for 1 species and tested it with data from the other species. In their case, the RSF for elk accurately predicted mule deer locations, but the converse did not hold, suggesting that mule deer avoided elk, but elk did not avoid mule deer (Johnson et al. 2000). However, the current amount of data for moose and particularly wolves did not warrant such an approach at this time. Until more data become available, we used a simpler approach by plotting the mean weekly elevation use of moose and caribou to examine the seasons where these species are most likely to come into contact with one another. Because sample sizes were small for wolves, we plotted the mean weekly elevational use of each wolf against the pooled data of moose and caribou.
**GPS Radio-collar Cluster Analysis**

**Predation Site Investigation**

To identify GPS clusters, we calculated the distance between fixes using Arcview 3.3. These results were grouped into categories of 100 m and graphed using a histogram to determine a breakpoint that would represent the difference between resting/feeding distances and moving distances. A breakpoint of 200 m was visually estimated from the histogram. A cluster was defined as ≥2 locations less than 200 m apart. The average cluster location was delineated using the mean UTMs of all points making up the cluster. Twenty-nine clusters were randomly selected to be investigated in the field. Cluster sites were methodically searched for a minimum of 1 hour to find evidence of a large-mammal predation event. When kills were located, the species, age, and sex were recorded if possible. The incisor teeth, femur, and hair samples were taken to confirm age, nutritional condition and sex of the prey species (Sergeant and Pimlott 1959; Neiland 1970).

**GPS Cluster Predation model**

Using binary logistic regression, we modeled the probability that a large-mammal predation event occurred at a GPS cluster site using 6 predictor variables pre-screened for multicollinearity. The probability of a large-mammal predation event being present (1) or not present (0) at the GPS cluster site was modeled as:

\[
P_{PE} = \frac{\exp(\beta_0 + \beta_1 x_1 + \beta_2 x_2 + \ldots + \beta_n x_n)}{1 + \exp(\beta_0 + \beta_1 x_1 + \beta_2 x_2 + \ldots + \beta_n x_n)}
\]

where \(P_{PE}\) is the probability of a large-mammal predation event, \(\beta_0\) is the regression constant and \(\beta_1 \ldots \beta_n\) are coefficients estimated for variables \(x_1 \ldots x_n\) (Hosmer and Lemeshow 2000). The predictor variables included GPS precision (% 3D locations), number of days the collared wolf spent at a cluster (DAY), if the wolf made multiple visits to the site (REVISIT; yes=1 or no=0), habitat type (HAB; forested, wetland, cutblock), time spent searching the area, and ground visibility (poor=1, good=0). Using Akaike’s information criterion (AIC_c) for small sample sizes with no overdispersion (c), we assessed 8 apriori candidate models to identify the best model that predicted the probability of finding a large-mammal predation event at the cluster site.

Using \(\Delta_i\) values (\(\Delta_i = AIC_i - \text{minAIC}\)), the candidate models were ranked and the strength of evidence for each model was determined (Burnham and Anderson 2001). We also calculated Akaike weights (\(w\)), which provides a measure of the relative likelihood of an individual model given the data and the chosen set of candidate models. The assessment of the relative importance of the predictor variables was estimated from the set of models, rather than the best model, using the sum of the Akaike weights for each
Due to small sample sizes, all data were used for model building. All statistical analyses were performed using SPSS 13.0 software (SSPS 2004).

**Kill Rate Estimation**

Radiocollared wolf packs were monitored using a combination of fixed-wing aircraft and ground-based winter tracking between November and March of 2003/04 and 2004/05. Two wolf packs, named the Downie and Goldstream packs after the major drainage in which they spend most of their time, were monitored for 1 and 2 winters respectively. Wolf locations and kill sites were sighted from the air using a fixed-wing aircraft. To augment aerial searching, wolves were tracked on the ground from the last known telemetry point to locate kills, document habitat use, determine wolf pack size, snow depth and snow characteristics (Smith et al. 2004; Hayes et al. 2000). Mean pack size for the analysis was determined from visual counts during aerial telemetry (Hebblewhite et al. 2003).

When kills or scavenges were located, the species, age, and sex of the prey was recorded if possible. The incisor teeth, femur, and hair samples were also taken to confirm age, nutritional condition and to provide DNA of the prey (Neiland 1970; Sergent and Pimlott 1959). Criteria used to determine the difference between a kill or scavenge was based on research completed by Huggard (1993). Moose were categorized as calf (<1 yrs), subadult (1-2yrs) and adult (>3 yrs).

Kill rates and scavenge rates were estimated using the ratio estimation method (Hebblewhite et al. 2003). When attempting to estimate winter kill rates, the winter is often partitioned into sampled and unsampled time periods. These periods are often randomly distributed and of variable lengths due to weather and logistical constraints of maintaining a continuous tracking session. By accounting for these constraints, this method estimates kill rates using a linear model where $\beta$=kill rate, $y_i$=the number of kills in sampling interval $i$, and $x_i$= number of days in sampling interval $i$ (Hebblewhite et al. 2003). Therefore kill rate is estimated by,

$$\beta = \frac{\sum y_i}{\sum x_i}$$

where $i$=the sampling period, 1 to $n$, and $n$=total number of sampling periods.

The ratio estimator approach was found to provide the least bias and most precise kill rate estimates when compared to other methods (Hebblewhite et al. 2003; Ballard et al. 1997; Dale et al. 1994). Total wolf kill rates are reported in kills/day/pack (k/d/p). Scavenge rates are reported as scavenges/day/pack (s/d/p).
To allow for greater comparisons among studies, we converted the number of prey to kilograms of prey killed per day per wolf (kg/d/w) using moose mass estimates from Hayes et al. (2000) and corrected for the percent of carcass consumed (65%, Hayes et al. 2000).

**Landscape Attributes that Affect the Predation Risk of Moose by Wolves**

We used locations of dead moose from the kill rate estimation section, plus other mortalities that did not form part of continuous tracking sessions, as our sample of where moose were killed by wolves. Telemetry data from radio-collared moose was our sample of where moose live. Each moose kill represented an independent data point, whereas individual live moose represented independent data points, with telemetry locations nested as subsamples within each live moose, to avoid pseudoreplication. The analysis was restricted to the winter season (Dec. 5 to May 10) because there were too few kill sites identified during the non-winter season. In a GIS we created a 200-m radius buffer around each kill site and telemetry location. This distance represents the approximate mean difference between where wolves encountered moose to where wolves successfully preyed on moose, as determined from snow-tracking sessions (Stotyn, unpublished data). Within each buffer, we queried digital forest cover and TRIM files to extract the percent cover of each tree species, percent of logged (<30 yrs old), riparian, and deciduous habitat, and mean values for stand age, canopy cover, elevation, and slope. We also used the point locations of kill sites and telemetry locations to extract the nearest distance to major water sources (e.g., Goldstream River, Downie Creek, Lake Revelstoke), all water sources (any small creek or major water source), all roads (including forestry mainlines, spur roads, and highways), and edge of early seral (< 30 yrs)/late seral forest types. We conducted univariate comparisons between kill sites and telemetry locations of live moose. To contrast the strength of different variables and account for potential interactions among variables, we conducted logistic regression analyses to predict the location of a kill site (y=1) relative to live moose sites (y=0), based on the aforementioned predictor variables. Similar to the GPS cluster analysis, we used AIC corrected for small sample sizes to gauge the strength of different models and particularly different variables, which may explain the location of moose kills.

**Results**

**Spatial Overlap of Wolves, Moose and Caribou**

The degree of elevational separation between moose and caribou was greatest during the “late winter” caribou season (January 11 to April 22; Apps et
al. 2001; Figure 1), when caribou migrate to high elevations to access arboreal lichen, their primary winter food source. During summer, moose and caribou are more dispersed throughout the landscape, but on average, the elevational separation between the 2 species is at its minimum relative to the other seasons.

**GPS Cluster Analysis**

**Predation Site Investigations**

Potential kill sites were identified from GPS data downloaded from a LOTEK GPS_3300 collar retrieved in August 2004. The LOTEK GPS_3300 collar recorded wolf movements from March 10 to August 18th, 2004 and had a fix rate of 48.1%. Of those successful fixes, 62% were 3D and 38% were 2D fixes. There was evidence of a large mammal predation event at 40% of the sites investigated (Figure 9). Cluster sites were located in forested (52%), wetland (12%) and cutblock (40%) habitat types (Figure 2). Of the 29 GPS cluster sites identified, 4 were not investigated because of recent snow cover and poor accessibility. Sixty percent of the sites investigated were classified as having poor ground visibility due to leaf cover, dense forbs or snow cover. The identification of prey species, age, and body condition from hair and bones collected at the site are not complete at this time. Sites that had bones with evidence of human handling (saw marks) were assumed to be scavenged by wolves, and not included in the analysis.

**GPS Cluster Predation Model**

AIC values were used to rank the models using the smallest AIC value as an indicator of the best-supported model given the data (Table 2). Models with $\Delta_i \leq 2$ are considered to have substantial support and those models with $4 \leq \Delta_i \leq 7$ have reduced support.

The model $(\phi) = \text{DAY} + \text{REVISIT}$ was the best-fitting model based on AIC model selection (Table 2). The second ranked model, $(\phi) = \text{HAB} + \text{DAY} + \text{REVISIT}$, had 11.6 times less support than the top ranked model ($w_i/w_j = 11.6$). The third model, $(\phi) = \text{DAY}$, is two times less supported than the second ranked model. The last 5 models have considerably less support than the top 3 models. The best model $(\phi) = \text{DAY} + \text{REVISIT}$ correctly predicted clusters with a large mammal predation event 80% of the time. In this model, the probability of a large mammal predation event being present at a GPS cluster site increased with the number of days a wolf spent at the cluster ($p=0.018$) and decreased if the cluster had been revisited over a number of non-consecutive days ($p=0.024$; Table 3).

Based on AIC weights for individual variables, number of days (DAY) and revisit occurrence (REVISIT), were 92% more important than habitat and 99.7% more important than the variables precision, ground cover, and search time in their ability to predict the occurrence of a large mammal predation event (Table 3).
**Kill Rate Estimation**

The territory size of the Goldstream and Downie wolf packs, determined from bi-weekly aerial monitoring, were 1001 km² (n=70) and 431 km² (n=26; MCP 95%; Figure 10), respectively. The pack size of the Goldstream and Downie wolf packs were 7 and 12 wolves respectively during winter tracking sessions. Wolves were tracked for a total of 71 days in 20 sampling intervals, averaging 4.1 days each interval (Table 5). During these sampling intervals we located 15 kills and 14 scavenges. The average wolf-pack kill rate for the Goldstream pack was 0.207 k/d/p and 0.077 k/d/p for the Downie pack (Table 5). The average wolf-pack kill rate for both packs for both years was 0.16 k/d/p, all of which were moose, with the exception of one unidentified ungulate. The total number of kills extrapolated to the entire winter (November 8 to April 30th) is 36.2 kills for the Goldstream and 13.5 kills for the Downie Pack.

The average per capita consumption rates in kg prey killed/day/wolf (adjusted for percent edible and consumed) was 8.375 kg/d/w for the Goldstream and 1.55 kg/d/w for the Downie wolf pack. Scavenge rates for the Downie wolf pack was twice as high as the Goldstream pack at 0.25 and 0.13 s/d/p (Table 6).

There were no confirmed mortalities of mountain caribou during tracking sessions. Further analysis of wolf scat and the use of stable isotope analysis may elucidate prey missed during tracking sequences. Using guard hair and bone collagen samples, stable isotope analysis has been used to determine the diet composition of wolves (Darimont and Reimchen 2002, Szepanski et al. 1999). When a wolf consumes a prey item, the distinct carbon and nitrogen signature of the prey item is deposited chronologically in inert tissues such as hair. This hair can be analyzed using stable isotope models to get an indication of relative prey consumption of each species consumed by wolves during that period of hair growth.

**Landscape Attributes that Affect the Predation Risk of Moose by Wolves**

During winter, we recorded 35 moose killed by wolves, and a total of 129 telemetry locations for 17 moose when they were alive. Unfortunately, several of the predation events and telemetry locations were found on private land (Beaumont Timber), where digital forest cover information was not readily available, so these locations were discarded from the analysis. We were left with 29 predation events and 96 telemetry locations for analysis.

Moose were killed an average of 895 m (SE=278.4) from a major water source, compared to live moose, which were found 633 m (SE= 134) from major water sources (Table 6). Dead moose were located 30% closer to seral edges (54.3 m [SE=22.6] vs. 77.9 m [SE=19.6]) and had 30% more logged forest within the 200-m buffer, relative to live moose. Moose were predated upon at an average of 839.6 m above sea level (asl; SE=52.2), compared to live moose, which were located at 780.1 m (SE=29.0) asl. The variance for most of the
parameters we estimated was large, suggesting that higher sample sizes will be required to clarify these comparisons.

The multivariate modeling revealed that the intercept-only model (i.e., null model) was the most parsimonious, confirming that the variance of parameter estimates was too large to derive strong conclusions (Table 7). However, of the models that included covariates, the model with distance-to-major-water was the best, followed by elevation, and distance-to-early-seral edge. Greater distance-to-major-water and elevation were positively associated with finding a moose kill, whereas moose were more likely to be killed with decreasing distance to early seral/late seral habitat edges.

Discussion

Spatial Overlap of Moose, Wolves, and Caribou

Our preliminary analysis of elevational overlap among the 3 species suggests that caribou are at the lowest probability of encountering moose and wolves during late winter (Allison 1998). Unfortunately, the wolf data is still too sparse to reliably quantify seasonal predation risk to caribou. However, if the moose elevation data is used as a surrogate for wolf locations, it appears that caribou are least threatened by wolves during late winter, and the greatest degree of threat occurs during late summer, with some risk during spring and early winter. Wittmer et al.’s (in press) provincial analysis of caribou mortality indicates that predation is greatest during spring and summer, and least during the 2 winter seasons, particularly during late winter.

GPS Cluster Analysis

Predation Site Investigations

We were able to successfully locate prey remains at 40% of the sites that were identified using GPS cluster analysis. This technique gave us the first look at summer wolf predation for packs in the northern Columbia Mountains. Using this technique in the future could elucidate summer predation patterns, increase sample sizes for prey selection studies, reduce field costs, and allow for the extrapolation of kill sites to estimating kill rates during all seasons.

As with any study that uses GPS data, one must be aware of the potential bias and problems inherent with this kind of data. In our study, the GPS collar had a low fix success rate (48.1%) relative to what has been reported elsewhere (84% and 88%; Mark Hebblewhite, Pers comm., Kim Lisgo, Pers. comm.). Poor GPS collar performance may be attributed to the interaction of steep terrain, dense canopy cover, seasonal ranges or animal behaviour (D'eon et al. 2002; Frair et al. 2004). Biases in GPS locations can introduce error and result in misleading conclusions. For example, uncorrected GPS bias could lead to the
inference that steep slopes, or closed conifer forests have reduced predation risk, or that wolves spend less time in these habitats. An attempt to correct for these biases have been developed by Frair et al. (2004) and D'Eon (2003).

Additional sources of bias could be introduced due to the differences in sightability in different habitats. In some habitats, dense shrub and herb layers may hamper the ability to search for prey remains, therefore reducing the probability of detection. We attempted to reduce this source of bias by conducting detailed grid pattern searches during site visits. Difference in detectability was included as a covariate in our models, but it was not a significant factor.

While other sources of bias may lead to underestimates of wolf predation rates, the inability to differentiate between a predation or scavenging event may lead to an over-estimate in kill rates. Baseline predation and scavenging rates for the study area have been determined using winter tracking in 2003 and 2004. These rates could be used as a correction factor when GPS cluster analyses are used to extrapolate kill rates in the future.

**GPS Cluster Predation Model**

With data gathered at GPS cluster sites we were able to develop a model that isolates important factors that can be used to predict large mammal predation events with future GPS data. The number of days spent the collared wolf spent at a cluster and whether the site was revisited over a number of occasions has been shown to be statistically significant in the model. What is the biological significance of these factors? The number of days a wolf pack spends near a moose kill is 2-4 days and less than 24 hours for a deer or caribou (Thurber and Peterson 1993; Hayes and Harestad 2000; Kunkel et al. 2004). Due to the prominence of moose on the landscape (3.54 moose/km²; Poole and Serrouya 2003) and in the wolves’ winter diet (Stotyn 2003), we can assume that moose make up a large proportion of the summer diet as well. Therefore, as the number of days spent at a location increases, so does the probability that a moose kill has occurred. Anderson and Lindzey (2003) also found the probability that a cougar killed a large mammal increased with the number of nights of cougar presence within a 200-m radius of the center point of the cluster.

The negative relationship between the number of times a site is revisited by the GPS collared wolf may be due repeated dispersion of the prey remains, or these clusters were favoured bedding, denning or rendezvous sites.

GPS kill clusters may also help reveal important mechanistic differences in caribou mortality patterns across seasons that would be missed using solely patterns of spatial overlap. Although there is a high degree of spatial separation between caribou and wolves during late winter, with correspondingly less mortality (Wittmer et al. in press), this separation is not as pronounced during other seasons. Despite the lack of clear spatial separation between wolves and caribou during spring, summer, and early winter, there are pronounced differences in caribou mortality rates among these seasons. At a provincial scale, caribou have more than twice the chance of being killed in summer relative
to early winter (Wittmer et al., in press). We hypothesize that the lack of predation in early winter is a function of predator foraging behaviour related to search times for primary prey. Holling’s disk equation (1959) provides a mechanism to test this hypothesis, because the equation separates wolf behaviour into search times and handling times. The type II functional response asymptotic curve predicts that as prey density increases, search times are reduced, but handling times remain the same. The asymptote is reached because search times can become negligible at high prey densities, but handling time cannot be reduced because wolves cannot consume primary prey any faster. We predict that the discrepancy in mortality patterns between summer and early winter is because moose are more dispersed during summer, resulting in greater search times by wolves, thereby increasing incidental encounters with caribou. Whereas in early winter, moose are more concentrated, so search times are expected to be lower, with less incidental encounters (and consequent predation) with caribou. Quantifying kill rates across different seasons using GPS clusters will help clarify this hypothesis because if the distance between kill clusters is greater during summer, we could infer greater probability of encounter with caribou.

We look forward to continuing this research and refining our predation model with the deployment of the three remotely downloadable GPS collars (HABIT, Victoria, B.C) in March 2005. If successful, these data may validate model predictions, increase sample size and reduce the time interval to investigating the site, possibly improving the identification of prey remains and probability of finding a kill at the site. Part of the model validation will include comparing kill rates determined from GPS clusters during winter with more traditional snow trailing and aerial-detection methods of estimating kill rates.

**Kill Rate Estimation**

When attempting to compare kill rates among studies, the per capita amount of biomass eaten by wolves per day is the desired parameter, because kill rates can vary with the prey size, wolf pack size and snow conditions of the study area (Jedrzejewski et al. 2002). The mass of prey killed/day/wolf in the Goldstream valley was within the range reported by Dale et al. (1995) and Ballard et al. (1987) of 4.1-12.0 kg/w/d and 4.5-14.9 kg/w/d respectively. Kill rates were lower for the Downie wolf pack possibly due to the limited number of sampling intervals in 2004/05. Hebblewhite et al. (2003) suggest that 25% of the winter, sampled in ≥6-8 individual sampling intervals, is required to minimize sampling variation in kill rate estimates. Lower kill rates could also be due to the high scavenging rate obtained during those sampling intervals. Huggard (1993) found that wolves are opportunistic scavengers, independent of kill rates. A high scavenge rate, especially in late winter when animals may die of other causes, may reduce kill rates because a substantial portion of the diet may be composed of scavenged food (Huggard 1993).
Winter wolf kill rates cannot be extrapolated to snow-free periods because of the influence of snow depth, seasonal pack cohesiveness, spatial arrangement of predators and prey, and changing prey vulnerabilities (Hebblewhite et al. 2003; Jedrzejewski et al. 2002; Thurber and Peterson 1993; Huggard 1993; Nelson and Mech 1986). We have attempted to address these limitations by using GPS cluster analysis to investigate summer kill rates.

**Landscape Attributes that Affect the Predation Risk of Moose by Wolves**

Ungulates use a number of behavioural adaptations to minimize predation risk including group living (Bertram 1978), dispersion (Bergerud and Page 1987), and the selection of specific terrain conditions or habitat types (Kie 1999). Some habitat conditions afford protection to ungulates by providing escape terrain, such as steep slopes for mule deer, or rugged terrain used by mountain goats. Moose have been found to use islands or water bodies to minimize predation risk (Stephens and Peterson 1984, Leptich and Gilbert 1986, Jackson et al. 1991, Addison et al. 1993). In our study, we found that in winter, moose that ventured further from a major water source were more likely to be killed by wolves. It is likely that water bodies such as the Goldstream River or Downie Creek provide a refuge to moose because wolves have difficulty killing prey in water. This result contrasts with other moose-wolf study areas, where most water bodies freeze in winter, which provides increased travel and foraging efficiency for wolves.

Moose were also more likely to be killed at higher elevations, although elevation was correlated with distance-to-major-water ($r=0.63$), but the latter is likely the more mechanistic variable. Moose were more likely to be killed near seral edges, implying that moose that were either deep within clearcuts or old forests were less likely to be eaten by wolves. While trailing wolves in snow we often noted that they traveled in old forests adjacent to logged areas and other openings. As moose forage further into logged areas, it may become more difficult for wolves to locate them because of deeper snow and reduced visibility.

Proximity to logging roads did not seem to affect predation risk, although we did not have the data to differentiate between major, cleared logging roads and minor logging roads with deep snow. In Alberta, James and Stuart-Smith (2000) found that linear corridors facilitated wolf predation by providing faster travel routes for wolves.

**Management Implications**

As GPS-animal collar technology becomes more prevalent in wildlife studies it is important to develop methods that can take full advantage of this technology. Making the link between GPS locations and animal behaviour could be important to increase our understanding of complex systems for the benefit of conservation and management efforts.
Information on winter kill rates and scavenge rates can act as a baseline from which we can measure the effects of changing prey densities as a result of management scenarios, successional processes and changing climates.

Of special interest for conservation efforts is the spatial location of kill sites on the landscape in relation to terrain features, roads, the intensity and pattern of forest harvesting, habitat and human use. Information on the interaction of predation risk and landscape attributes may be used to define high and low risk habitats for moose and caribou with regards to wolf predation. Ultimately our work on moose predation risk will be expanded to include non-winter seasons and contrasted with the caribou predation risk model being developed by other members of our research team (C. Apps, H. Wittmer, B. McLellan) to identify zones of shared predation risk between the 2 species.

Acknowledgments

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Literature Cited


SPSS. 2004. SPSS 13.0 for Windows. Illinois, USA.
Figure 8. Mean weekly elevation use for mountain caribou (purple line; modified from Apps et al. 2001), moose (red line), and individual wolves (circles – each colour is an individual), in the northern Columbia Mountains, BC. Seasons identified are based on Apps et al.’s (2001) definition of mountain caribou seasons in the Columbia Mountains.

Figure 9. The number and success of cluster investigations in each habitat type in the northern Columbia Mountains, British Columbia.
Figure 10 Home ranges of three wolf packs (Red Rock, Goldstream and Downie) in the northern Columbia Mountains, British Columbia from telemetry locations gathered during 2003 to 2005 (MCP 95%).

Table 2. Logistic regression models for GPS cluster site investigations in the northern Columbia Mountains, British Columbia, Canada. The top set of candidate models showing the model structure, maximized log likelihood (LL), the number of parameters (K), Akaike’s information criteria for small sample sizes (AICc), change in AIC from the “best” fit model (Δi) and the Akaike weights (Wi). *Interaction terms for the specified variables.

<table>
<thead>
<tr>
<th>Rank</th>
<th>Model Structure</th>
<th>LL</th>
<th>K</th>
<th>AICc</th>
<th>Δi</th>
<th>Wi</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>DAY¹+REVISIT²</td>
<td>20.33</td>
<td>4</td>
<td>30.33</td>
<td>0.00</td>
<td>0.892</td>
</tr>
<tr>
<td>2</td>
<td>HAB²+DAY+REVISIT</td>
<td>18.55</td>
<td>6</td>
<td>35.22</td>
<td>4.89</td>
<td>0.077</td>
</tr>
<tr>
<td>3</td>
<td>DAY</td>
<td>32.01</td>
<td>2</td>
<td>36.55</td>
<td>6.22</td>
<td>0.040</td>
</tr>
<tr>
<td>4</td>
<td>DAY*REVISIT</td>
<td>30.74</td>
<td>3</td>
<td>37.88</td>
<td>7.55</td>
<td>0.020</td>
</tr>
<tr>
<td>5</td>
<td>PRE⁴</td>
<td>33.40</td>
<td>3</td>
<td>40.54</td>
<td>10.21</td>
<td>0.005</td>
</tr>
<tr>
<td>6</td>
<td>SEARCH⁵+GROUND⁶</td>
<td>31.46</td>
<td>4</td>
<td>41.46</td>
<td>11.13</td>
<td>0.003</td>
</tr>
<tr>
<td>7</td>
<td>HAB+DAY</td>
<td>31.56</td>
<td>5</td>
<td>44.72</td>
<td>14.39</td>
<td>0.001</td>
</tr>
<tr>
<td>8</td>
<td>DAY+HAB+GROUND+PRE+REVISIT+SEARCH</td>
<td>17.84</td>
<td>9</td>
<td>47.84</td>
<td>17.51</td>
<td>0.000</td>
</tr>
</tbody>
</table>

¹ Number of days spent at the cluster site; ² if wolves revisited the site (Y=1, N=0); ³ habitat class (forested, wetland, cutblock); ⁴ precision of fix (% 3D fix accuracy); ⁵ time spent searching the area; ⁶ ground visibility (good=0, poor=1).

Table 3. Multi-model inference of the importance of predictor variables in the identification of large mammal predation events using wolf GPS data in the northern Columbia Mountains, British Columbia, Canada.

<table>
<thead>
<tr>
<th>Predictor variable</th>
<th>Predictor weight (w_ij)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Days</td>
<td>0.991174</td>
</tr>
<tr>
<td>Revisit</td>
<td>0.990504</td>
</tr>
<tr>
<td>Habitat</td>
<td>0.078297</td>
</tr>
<tr>
<td>Precision</td>
<td>0.003555</td>
</tr>
<tr>
<td>Search time</td>
<td>0.003555</td>
</tr>
<tr>
<td>Ground cover</td>
<td>0.003555</td>
</tr>
</tbody>
</table>

Table 4. Highest-ranked logistic regression model for estimating the probability of finding a large mammal predation event at a GPS cluster site in the northern Columbia Mountains, British Columbia, Canada (Nobs = 25).

<table>
<thead>
<tr>
<th>Variable</th>
<th>B</th>
<th>SE</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Days</td>
<td>2.335</td>
<td>0.991</td>
<td>0.018</td>
</tr>
<tr>
<td>Revisit Site</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yes</td>
<td>-6.013</td>
<td>2.656</td>
<td>0.024</td>
</tr>
<tr>
<td>Reference=No</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>-4.502</td>
<td>1.616</td>
<td>0.030</td>
</tr>
</tbody>
</table>
Table 5. Data collected during winter snow-tracking used to estimate kill rates and per capita consumption rates for 2 wolf packs in the northern Columbia Mountains, British Columbia during 2003-2005. Kill rates were calculated using a ratio estimator as outlined by Hebblewhite et al. (2003). Total wolf kill rates in kills/day/pack (k/d/p), and kg prey killed/day/wolf (kg/d/w).

<table>
<thead>
<tr>
<th>Wolf pack-year</th>
<th># of sampling intervals (N)</th>
<th>Mean length in days ($x_i$)</th>
<th># Days tracked (n)</th>
<th>% winter tracked $^a$</th>
<th># of kills ($y_i$)</th>
<th>Wolf pack size</th>
<th>Ratio-estimator</th>
<th>Estimated total number of kills for the entire winter (Y) $^a$</th>
</tr>
</thead>
<tbody>
<tr>
<td>GOLD 03/04</td>
<td>9</td>
<td>4.89</td>
<td>44</td>
<td>25.14</td>
<td>9</td>
<td>6</td>
<td>0.205</td>
<td>7.17</td>
</tr>
<tr>
<td>GOLD 04/05</td>
<td>8</td>
<td>3.00</td>
<td>14</td>
<td>8.00</td>
<td>5</td>
<td>8</td>
<td>0.208</td>
<td>9.58</td>
</tr>
<tr>
<td>DOWNIE 04/05</td>
<td>3</td>
<td>4.33</td>
<td>13</td>
<td>7.43</td>
<td>1</td>
<td>12</td>
<td>0.077</td>
<td>1.55</td>
</tr>
</tbody>
</table>

$^a$ the entire winter is considered to be from Nov. 8 - April 30 to correspond to mountain caribou early and late winter delineation (175 days).

$^b$ number of prey to kilograms of prey killed per day per wolf using moose mass estimates from Hayes et al. (2000) and corrected for the percent of carcass consumed (65%, Hayes et al. 2000).

Table 6. Data collected during winter snow-tracking used to estimate scavenging rates for 2 wolf packs in the northern Columbia Mountains, British Columbia during 2003-2005. Scavenge rates were calculated using a ratio estimator as outlined by Hebblewhite et al. (2003). Total wolf scavenge rates in scavenges/day/pack (s/d/p), and scavenges/day/wolf (s/d/w).

<table>
<thead>
<tr>
<th>Wolf pack-year</th>
<th># sampling intervals (N)</th>
<th>Mean length in days ($x_i$)</th>
<th># days tracked (n)</th>
<th>% winter tracked $^a$</th>
<th># of scavenges ($y_i$)</th>
<th>Wolf pack size</th>
<th>Ratio-estimator</th>
<th>Estimated total number of scavenges for the entire winter (Y) $^a$</th>
</tr>
</thead>
<tbody>
<tr>
<td>GOLD 03/04</td>
<td>9</td>
<td>4.89</td>
<td>44</td>
<td>25.14</td>
<td>8</td>
<td>6</td>
<td>0.182</td>
<td>0.030</td>
</tr>
<tr>
<td>GOLD 04/05</td>
<td>8</td>
<td>3.00</td>
<td>14</td>
<td>8.00</td>
<td>3</td>
<td>8</td>
<td>0.083</td>
<td>0.010</td>
</tr>
<tr>
<td>DOWNIE 04/05</td>
<td>3</td>
<td>4.33</td>
<td>13</td>
<td>7.43</td>
<td>3</td>
<td>12</td>
<td>0.250</td>
<td>0.021</td>
</tr>
</tbody>
</table>

$^a$ the entire winter is considered to be from Nov. 8 - April 30 to correspond to mountain caribou early and late winter delineation (175 days).
Table 7. Comparison of landscape factors around telemetry locations of live moose, compared to locations where wolves killed moose, in the northern Columbia Mountains, BC. Variables refer to the composition within a 200-m radius GIS buffer around each location, except distance variables, which were based on the point location.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Dead moose (N=29)</th>
<th>Live moose (N=16)</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Dead mean</td>
<td>Dead SE</td>
<td>Live mean</td>
</tr>
<tr>
<td>Hemlock (%)</td>
<td>8.8</td>
<td>2.4</td>
<td>18.3</td>
</tr>
<tr>
<td>Western red cedar (%)</td>
<td>35.9</td>
<td>3.7</td>
<td>30.0</td>
</tr>
<tr>
<td>Spruce (unclass) (%)</td>
<td>6.2</td>
<td>1.6</td>
<td>9.4</td>
</tr>
<tr>
<td>Engelmann spruce (%)</td>
<td>13.5</td>
<td>3.0</td>
<td>8.4</td>
</tr>
<tr>
<td>Douglas-fir (%)</td>
<td>5.5</td>
<td>3.0</td>
<td>6.7</td>
</tr>
<tr>
<td>Deciduous (%)</td>
<td>3.8</td>
<td>1.4</td>
<td>4.3</td>
</tr>
<tr>
<td>Slope (%)</td>
<td>28.0</td>
<td>3.4</td>
<td>31.5</td>
</tr>
<tr>
<td>Age (yr)</td>
<td>99.2</td>
<td>13.0</td>
<td>108.8</td>
</tr>
<tr>
<td>Crown closure (%)</td>
<td>32.0</td>
<td>3.0</td>
<td>33.0</td>
</tr>
<tr>
<td>Non-Productive Brush (%)</td>
<td>4.4</td>
<td>1.2</td>
<td>6.4</td>
</tr>
<tr>
<td>Riparian (%)</td>
<td>6.9</td>
<td>2.7</td>
<td>5.4</td>
</tr>
<tr>
<td>Logged (%)</td>
<td>35.7</td>
<td>5.7</td>
<td>27.5</td>
</tr>
<tr>
<td>Elevation (m)</td>
<td>839.6</td>
<td>52.2</td>
<td>780.1</td>
</tr>
<tr>
<td>Distance to major water source (m)</td>
<td>894.6</td>
<td>278.4</td>
<td>632.8</td>
</tr>
<tr>
<td>Distance to any water source (m)</td>
<td>552.9</td>
<td>163.2</td>
<td>473.2</td>
</tr>
<tr>
<td>Distance to seral edge (m)</td>
<td>54.3</td>
<td>22.6</td>
<td>77.9</td>
</tr>
<tr>
<td>Distance to road (m)</td>
<td>472.5</td>
<td>193.0</td>
<td>454.2</td>
</tr>
</tbody>
</table>

$^1$ Percent difference calculated for mean values as (dead – live)/live* 100% = “percent difference of the dead value from the live value.”
Table 8. Logistic regression models to predict the probability of finding a moose predation event (1), relative to live moose (0). The top set (0-4 $\Delta$AICc units of candidate models showing the model structure, maximized log likelihood (LL), the number of parameters (K), Akaike’s information criteria for small sample sizes (AICc), change in AIC from the “best” fit model ($\Delta_i$) and the Akaike weights ($w$). Direction of parameter estimate (+ or –ve) is given in brackets beside each variable.

<table>
<thead>
<tr>
<th>Rank</th>
<th>Model Structure</th>
<th>-2LL</th>
<th>K</th>
<th>AICc</th>
<th>$\Delta_i$</th>
<th>$w_i$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Null model</td>
<td>135.42</td>
<td>1</td>
<td>137.57</td>
<td>0.00</td>
<td>0.18</td>
</tr>
<tr>
<td>2</td>
<td>Distance to major water (+)</td>
<td>133.84</td>
<td>2</td>
<td>138.30</td>
<td>0.73</td>
<td>0.13</td>
</tr>
<tr>
<td>3</td>
<td>Elevation (+)</td>
<td>134.49</td>
<td>2</td>
<td>138.96</td>
<td>1.39</td>
<td>0.09</td>
</tr>
<tr>
<td>4</td>
<td>Distance to early seral (-)</td>
<td>134.71</td>
<td>2</td>
<td>139.17</td>
<td>1.60</td>
<td>0.08</td>
</tr>
<tr>
<td>5</td>
<td>Distance to major water (+)</td>
<td>132.33</td>
<td>3</td>
<td>139.29</td>
<td>1.72</td>
<td>0.08</td>
</tr>
<tr>
<td>6</td>
<td>Distance to early seral (-)</td>
<td>132.33</td>
<td>3</td>
<td>139.29</td>
<td>1.72</td>
<td>0.08</td>
</tr>
<tr>
<td>7</td>
<td>Age (-)</td>
<td>135.16</td>
<td>2</td>
<td>139.62</td>
<td>2.05</td>
<td>0.06</td>
</tr>
<tr>
<td>8</td>
<td>Logged(%) (+)</td>
<td>135.31</td>
<td>2</td>
<td>139.77</td>
<td>2.20</td>
<td>0.06</td>
</tr>
<tr>
<td>9</td>
<td>Distance to road (+)</td>
<td>135.38</td>
<td>2</td>
<td>139.84</td>
<td>2.27</td>
<td>0.06</td>
</tr>
<tr>
<td>10</td>
<td>Distance to major water (+)</td>
<td>133.62</td>
<td>3</td>
<td>140.58</td>
<td>3.01</td>
<td>0.04</td>
</tr>
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</table>