University of Alberta

Woodland Caribou Demography and Persistence Relative to Landscape Change in West-central Alberta.

By

Kirby Gordon Smith ©

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Dedication

This thesis is dedicated to the memory of my father, Henry Christian Smith, who instilled in me an appreciation of wildlife conservation and the outdoors.

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Monitoring woodland caribou over a 20-year interval is a task that involves a number of dedicated individuals. I would like to thank the many staff of the Alberta Fish and Wildlife Division for their tremendous contributions. Michael Bloomfield initiated the west-central Alberta woodland caribou program in 1978. Jan Ficht was instrumental in delivering the program throughout much of its duration and continues to be the main source of "corporate memory" for woodland caribou information in west-central Alberta. I thank Bernie Goski for his exceptional net-gunning expertise and safe handling of animals throughout much of the 20-year study and I am tremendously grateful for his contributions. Dave Hobson and John Taggart flew countless hours of telemetry flights and Dave ushered us into the era of GPS technology and covered for me while I attended classes at university. I am very grateful to Troy Sorensen for helping with analysis and for his contribution to the population projection models. Dave Hervieux took the lead in implementing landuse guidelines designed to formalize woodland caribou habitat protection and most recently the Provincial Woodland Caribou Recovery plan and I thank him.

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CHAPTER 1 – INTRODUCTION

Woodland caribou (*Rangifer tarandus caribou*) are classified as threatened in Alberta (Dzus 2001). Population parameters of three herds have been monitored in west-central Alberta following the termination of licensed hunting in 1981. Since that time, industrial activity (i.e. logging, oil and gas exploration and extraction, mining) continues to have the potential to cause woodland caribou to decline as it both reduces and fragments the large contiguous landscapes of older forests that characterize the caribou's winter range (Edmonds 1988, Smith et al. 2000). This development may affect caribou directly through loss of habitat and indirectly through increased wolf (*Canis lupus*) predation, probably as a function of increased alternative prey [i.e. moose (*Alces alces*), elk (*Cervus elaphus*) and deer (*Odocoileus sp.*)] (Edmonds 1988, Bergerud and Ballard 1988, Cumming 1992, Thomas 1995, Rettie and Messier 2000).

Thesis Overview

In Chapter 2, I examine the demographics of 3 woodland caribou herds in west-central Alberta for the period 1997 – 2003. The population trajectory of each herd is modeled based on annual mortality of radiocollared adult females and calf recruitment obtained from composite aerial surveys.

In Chapter 3, I examine the consequences of changes in landscape condition to adult female survival based on 141 radiocollared animals in the 3 herds between February 28, 1981 and October 30, 2001. Cumulative timber harvest and linear disturbance (seismic lines, roads, powerlines and pipelines) are the landscape changes that are tracked.

Finally, in Chapter 4, I apply the results of the analysis in Chapter 3 to the population model generated in Chapter 2 and examine the implications to population persistence of 3 woodland caribou herds in west-central Alberta. I discuss the relevance of my findings to woodland caribou conservation and make recommendations regarding future landscape management.

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CHAPTER 2 - POPULATION DEMOGRAPHICS OF WOODLAND CARIBOU IN WEST-CENTRAL ALBERTA

INTRODUCTION

Understanding the population dynamics of a population at risk can be crucial to conservation efforts. Information on this aspect of woodland caribou ecology is limited. In the Birch Mountains of Alberta, Fuller and Keith (1981) estimated calf survival at 0.25 and adult survival (includes males) at 0.85. These authors concluded that because the percent of calves in composite counts during fall and winter (12%) was less than annual adult mortality (15%) the population was in decline. More recent work in northeastern Alberta estimated the survival of adult woodland caribou (0.88) and recruitment of calves to 10 months to calculate an average population rate of increase $\bar{r} = -0.08$ (Stuart-Smith et al. 1997).

Dzus (2001) discussed population trajectories of woodland caribou in Alberta. Based on the variability of juvenile survival observed in northern Alberta (0.10-0.40) and several assumptions of age-specific survival and fecundity, the analysis suggested that adult female survival must exceed 0.85 for population maintenance. Recent indications are that many populations in the boreal forests of Alberta have a negative rate of increase (McLoughlin et al. 2003). Population demographics of woodland caribou in north-central Saskatchewan also indicated a decline ($\bar{r} = -0.05$) (Rettie and Messier 1998).

Caribou calf mortality has been studied in herds that inhabit more open landscapes (alpine, arctic tundra) during the calving period. Predators killed 43% of collared calves before 30 days of age in Denali National Park, Alaska (Adams et al. 1995). Calf survival in the Porcupine barren-ground caribou herd of 0.57 - 0.90 was documented for the first month after birth, with 51% of calves surviving to yearling age (Fancy et al. 1994). This same study determined that yearlings and 2-year-olds survived within the range of estimated survival of adults (yearling females = 0.93; yearling males = 0.85; 2-year-old females = 0.96; 2-year-old males = 1.0).

Fancy et al. (1994) studied the population dynamics of the Porcupine barren-ground caribou herd during a period of increase. These authors evaluated the sensitivity of a population model by calculating the change in vital rates that would result in a 10% decrease in the modeled population estimate. They concluded that growth of the herd was most sensitive to the survival of ≥ 3-year-old females, followed by calf production and survival. The projected population would decrease by 10% if: 1) the survival of ≥ 3-year-old females decreased by 3% or 2) calf survival, or 3) calf production declined by 8-10%. Low variability in adult female survival and high variability in production and survival of young has been reported in other ungulates (Gaillard et al. 1998). In general, population growth is more sensitive to adult survival than juvenile survival and fecundity for long-lived species (Meyer and Boyce 1994 and others).). Adult female caribou survival influenced $\bar{r}$ slightly more than productivity during a period of decline in the George River herd (Crête et al. 1996). However, because of the high variability associated with juvenile survival, this parameter still plays a major role in the population dynamics of large herbivores (Gaillard et al. 2000).

Predation effects

Predation can be one of the major influences on woodland caribou population demographics. Predation on caribou by grizzly bears (*Ursus arctos*) and other carnivores has been reported (Edmonds 1988; Young and McCabe 1998, Adams et al. 1995); however, the bulk of the literature on predation has examined the effects of wolf (*Canis lupus*) predation (Bergerud 1974, Bergerud and Elliott 1986, Seip 1992, Adams et al. 1995, Hayes et al. 2003). The ability of caribou to minimize predation by wolves has been linked to spacing away from wolves and/or their alternate prey throughout the year (Bergerud et al. 1983, Bergerud and Page 1987, Allison 1998, Rettie and Messier 2000, Stuart-Smith et al. 1997).

Weather

Weather is often examined as a variable that can influence the population demographics of ungulates. Woodland caribou are well adapted to deep snow conditions. In west-central Alberta this is probably one means of allowing them to space away from alternate prey that prefer areas with less snow (i.e. deer and elk). Caribou activity may be altered by snow in excess of 1 m (Brown and Theberge 1990), which occasionally occurs in west-central Alberta (Edmonds 1988). At both the home range and local scale, caribou in west-central Alberta prefer older and denser stands of coniferous forest (Szkorupa 2002), which intercept snow better than younger, less dense stands. In springs with high snow packs, adult females do not appear to reach their remote calving areas in the mountains and calf survival suffers (Edmonds and Smith 1991). These authors suggest that heavy snowpack results in females calving too close to alternate prey and therefore calves experience a higher likelihood of wolf predation.

I present the results from a 22-year study of 3 woodland caribou herds in west-central Alberta. The first objective was to examine the population trajectory of 3 herds of woodland caribou based on a stage-structured model combining adult female survival and estimates of calf recruitment.

METHODS

Study Area

This study was conducted in west-central Alberta, Canada (54° N, 119° W) (Fig. 2.1). The area includes the upper foothills, sub-alpine and alpine ecoregions (Beckingham et al. 1996). The upper foothills ecoregion is characterized by an overstory of lodgepole pine (*Pinus contorta*) and white spruce (*Picea glauca*) with small patches of trembling aspen (*Populus tremuloidies*). The subalpine ecoregion is characterized by an overstory of Englemann Spruce (*P. Englemannia*) and subalpine fir (*Abies lasiocarpa*), while the alpine ecoregion has little overstory and is characterized by graminoids, sedges (*Carex spp.*) and bare ground. Three herds of woodland caribou were examined over a 22-year period (1981 – 2003). The A La Peche (ALP) and Redrock-Prairie Creek (RPC) herds had previously been categorized as *mountain ecotypes* (summers in the mountains, winters in the subalpine forest), while the Little Smoky (LSM) herd had been categorized as a boreal ecotype (spends the entire year in the subalpine and upper foothills

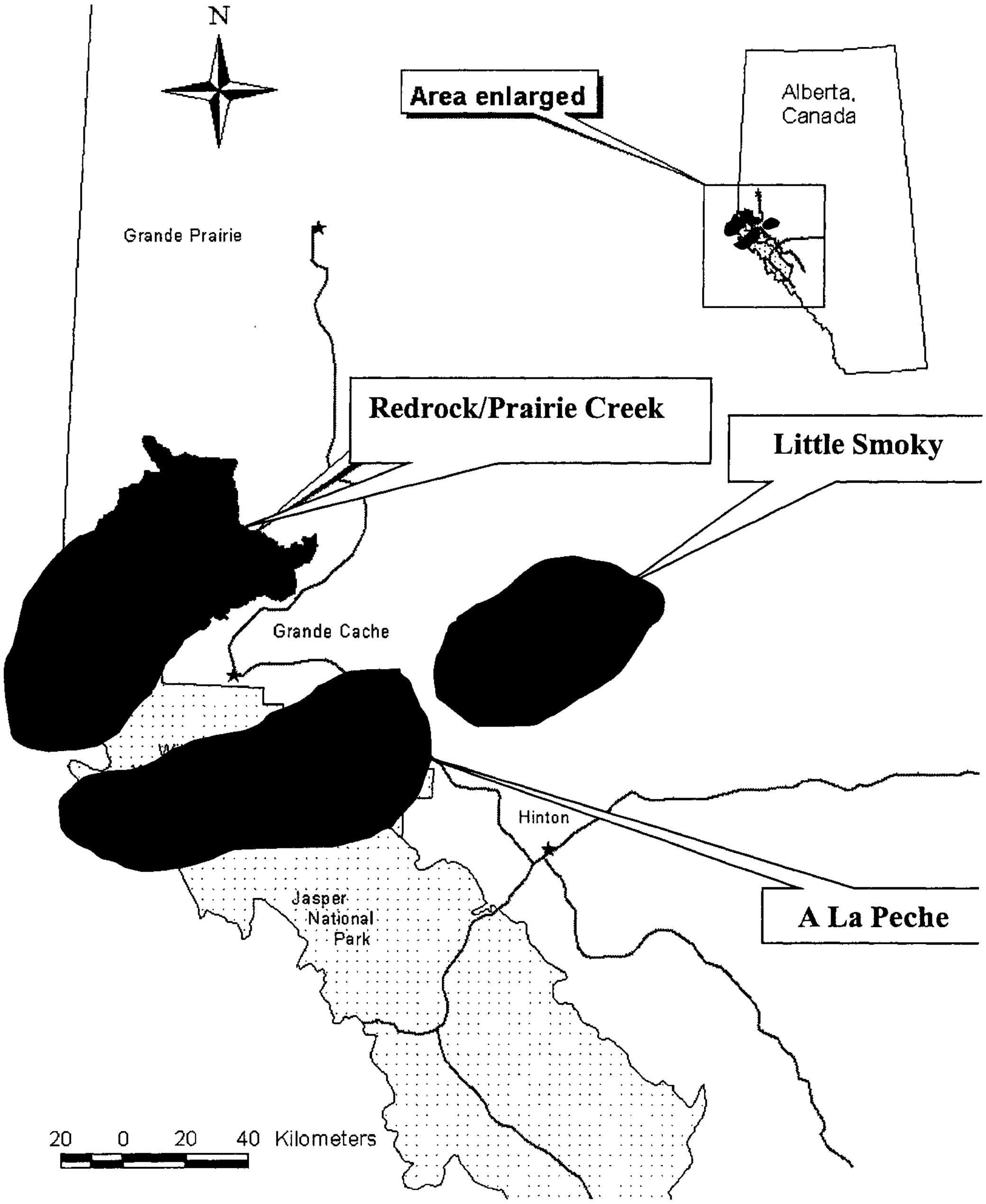


Figure 2.1. Map of the study area, which encompasses the A La Peche, Little Smoky and Redrock/Prairie Creek woodland caribou ranges, in west-central Alberta.

ecoregions) (Edmonds 1988). Other ungulates present include moose, elk, white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), bighorn sheep (*Ovis canadensis*) and mountain goats (*Oreamnos americanus*). Predators found in the area include wolves, grizzly bears (*Ursus arctos*), black bears (*U. americanus*), cougars (*Felis concolor*), lynx (*Lynx canadensis*), wolverine (*Gulo gulo*) and coyotes (*C. latrans*). The climate is characterized by short, cool, wet summers and long, cold, dry winters. Descriptions of the study area have been provided elsewhere (Edmonds 1988, Smith et al. 2000, Oberg 2001, Szkorupa 2002, Kuzyk 2002).

Radiocollared Sample

A total of 174 adult female woodland caribou were captured by helicopter-borne netgun (Barrett et al. 1982) from 3 ranges in west-central Alberta between 1993 and 2002. Animals were either fitted with very high frequency (VHF) or Global Positioning System (GPS) radio-collars (Lotek Engineering Inc., Newmarket, Ontario, Canada). Survival of caribou was monitored monthly by aerial tracking during the winter and monthly - bimonthly during the summer. The objective was to maintain a sample of at least 20 adult females in each woodland caribou herd range (Mean = 24.1, SD = 7.5, n = 15 herd years) by redeploying radiocollars where mortalities or collar failures were detected. Where possible, cause of death was determined by recovering collars as soon as mercury switches indicated prolonged (>8 hrs) lack of animal activity. To increase sample size, calf recruitment from recent work (1993 – 2003) was pooled with calf recruitment collected during previous studies on the same herds (Edmonds 1988, Edmonds and Smith 1991).

Adult Female Survival Rates

Range-specific adult female survival was estimated using Pollock et al.'s (1989) staggered-entry modification of Kaplan and Meier's (1958) survivorship model (Kaplan-Meier PL Estimator of Survival Rate, Version 2.1, Exeter Software, 47 Route 25A, Setauket, New York, New York, USA, coded by C. J. Krebs). Mean adult female survival was compared between herds (Nonparametric Program Ksample from E.T. Lee 1980, Version 3.1, coded by C. J. Krebs). The projected variability of a simulation model will be much larger than that of the actual population if sampling variance is incorporated in the calculations (Boyce 1992). Consequently, the sampling variance for each year was calculated to properly estimate the environmental variation of the series of years using the following formula adapted from White (2000).

[Eqn. 1] $$\hat{\text{var}}(\hat{S}_i / S_i) = \frac{\hat{S}_i(1-\hat{S}_i)}{n_i}$$

where $\hat{S}_i$ is the survival value for each year and n_i is the number of animals monitored that year. The mean of the yearly sampling variances was removed from the overall variance to determine the environmental variance.

Calf Recruitment

The proportion of calves in a composite sample was obtained through aerial surveys conducted between mid-September and late March. When the survey location provided opportunity for ground classifications to accurately sex adults (based on the presence/absence of a penis sheath), the calf/cow ratio was simply the number of calves observed divided by the number of cows observed. However, when the survey was conducted in forested environments, adult classification was not always possible. In those instances, the proportion of calves in each survey was transformed into a calf/cow ratio (*CCR*) as follows:

[Eqn. 2]

$$CCR = \frac{N \cdot C_p}{N - (N \cdot C_p) - M_p \cdot \left[N - (N \cdot C_p)\right]}$$

where N is the total number of animals in the aerial survey sample, C_p is the proportion of calves observed in the aerial survey sample and M_p is the proportion of males in the adult cohort based on previous ground classification surveys (0.35; Edmonds 1988, Edmonds and Smith 1991). This proportion of males is consistent with the mean sex ratio for other North American populations (0.36, Bergerud 1980). In years where fall composite counts were the only gauge of recruitment, calf-cow ratios were reduced by an additional 15% to account for over-winter mortality of calves (an approximation based on the comparison of calf-cow ratios of radiocollared females in the fall vs late winter and results of fall composite surveys vs late winter). An index of calf mortality in the LSM herd was assessed by tracking the number of calves-at-heel from early June through late March during 2 years. In order to determine if fall aerial survey counts obtained during the rut were useful indicators of recruitment, calf-cow ratios obtained during composite surveys were compared to ratios obtained from the presence of calves at the heel of radiocollared females during the same time period (χ^2 test). Finally, a review of the percent of males in samples of calves collected or observed in 5 different barren ground or woodland caribou herds was obtained from published literature to evaluate the assumption of a 50:50 sex ratio of calves that was used in the population modelling.

Population Estimates

The population of each herd was estimated using mark-resight methods from aerial surveys with radiocollars providing the "marks". Each aerial survey area was defined by locating all the radiocollared animals from helicopter and then "buffering" the outside of all radiocollared animal locations by 10 km. A second, independent helicopter crew flew the survey area the following day and recorded all marked and unmarked animals. The population estimate was calculated using the standard Lincoln-Petersen equation (Overton and Davis 1969):

[Eqn. 3] $$\hat{N} = \frac{nM}{x}$$

where $\hat{N}$ is the population estimate, n is the total number of animals observed during the survey, M is the number of animals that are marked and x is the number of marked animals observed during the survey.

Population Projections

The finite rate of increase ($\lambda_{i,t}$) for the female cohort was estimated for each caribou herd, i, and for each year, t, from annual recruitment of females, $R_{i,t}$ (or ½ of the calf-cow ratio assuming a 50:50 sex ratio of calves) and annual adult female mortality, $M_{i,t}$ equals $(1\text{-}S_{i,t})$, based on the following formula from Hatter and Bergerud (1991):

[Eqn. 4]

$$\lambda_{i,t} = \frac{(1 - M_{i,t})}{(1 - R_{i,t})}$$

These crude growth rates were plotted as a percent change.

In addition, a stage-structured stochastic population projection was calculated as follows:

[Eqn. 5]

$$N_F(t+1) = N_F(t) \cdot S_{random} + N_F(t) \cdot \frac{CCR_{random}}{2}$$

where $N_F(t)$ is the number of females in spring of year t, S_{random} is adult female survival generated from a normal random distribution (see below) and CCR_{random}

is the calf/cow ratio also generated from a normal random distribution. Dividing the product of the calf cow ratio and the number of adult females by 2 assumes a 50:50 sex ratio of calves.

A log-odds (or logistic) transformation of stochastic projections of adult female survival was applied to avoid unrealistic survival values (> 1.0) using the following formula adapted from Burnham et al. (1987):

[Eqn. 6] $$\text{logit} = \ln\left[\frac{\widehat{S}}{1-\widehat{S}}\right]$$

where $\widehat{S}$ = adult female survival. These survival values were then back-transformed to the original values with the following formula:

[Eqn. 7] $$\widehat{S} = \left[\frac{\exp^{\log it}}{1+\exp^{\log it}}\right]$$

This process resulted in log-normal survival values lower than the original. Consequently, the transformed values were increased until the mean survival rate was identical to the original values (Manly 1991).

Each 20-year population projection was replicated 200 times and the 200 individual projections summarized graphically in a single figure. Additionally the projections were replicated 1000 times and the mean population ± 1 standard deviation per year was plotted.

Results

Adult Female Survival

For the period 1997 – 2003, adult female survival was significantly different between the ALP and LSM herds ($F_{18,20} = 2.836$, $p < 0.025$) (Table 2.1). Adult female survival in the RPC herd was higher during the period 1993 – 1996 than 1998 – 2003, but the difference was not significant ($F_{8,34} = 1.306$, $p > 0.25$) (Table 2.1). The variance in adult female survival was similar between herds, both before and after sampling variance was removed (Table 2.2).

Table 2.1. Mean adult female survival, 95% confidence limits, the number of animals, the number of years and dates survival was monitored for 3 herds of woodland caribou in west-central Alberta.

Herd	Mean	Lower C. L.*	Upper C. L.*	N Animals	N Years	Dates
ALP	0.919[a]	0.870	0.958	48	5.6	10/97 – 04/03
RPC	0.947	0.884	1.000	24	3.4	10/93 – 03/97
RPC	0.873	0.821	0.915	65	4.7	10/98 – 04/03
LSM	0.848[a]	0.779	0.902	37	4.2	03/99 – 04/03

*These values include sampling variance

[a] Significant Difference ($F = 2.836$, df = 18, 20, $p<0.025$)

Table 2.2. Decomposition of the estimated variance in adult female survival into sampling variance and environmental (process) variance for 3 herds of woodland caribou in west-central Alberta.

Herd	Mean	Total Variance	Sampling Variance	Environmental Variance	Standard Deviation
ALP	0.919	0.0224	0.0032	0.0193	0.1388
RPC	0.873	0.0240	0.0030	0.0210	0.1448
LSM	0.848	0.0314	0.0077	0.0237	0.1539

Adult females died throughout the year with peaks in mortality occurring in late winter and early fall (Fig. 2.2).

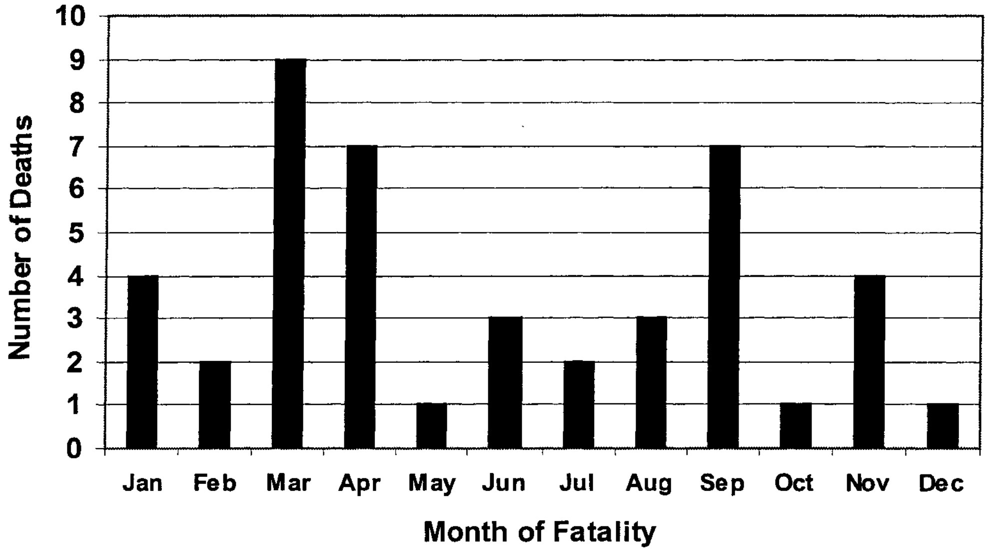


Fig.2.2. Frequency distribution of fatalities by month of year for adult female woodland caribou in west-central Alberta (1981-2003, n = 44).

Recruitment

The proportion of calves observed in composite aerial surveys (Table 2.3) and the resulting calf-cow ratios (Tables 2.4 – 2.6) were similar between the ALP and RPC herds but much lower in the LSM herd. The variance associated with calf-cow ratios was similar to the variance associated with adult female survival of the 3 herds (Table 2.7). Calf-cow ratios obtained from composite aerial surveys were not statistically different (with 1 exception) than calf-cow ratios calculated from the presence of calves at the heel of radiocollared females (Table 2.8).

Table 2.3. Mean proportion of calves in composite aerial surveys, the standard deviation, the number of surveys and years that surveys were conducted for 3 herds of woodland caribou in west-central Alberta.

Herd	Mean	Standard Deviation	N Surveys	Years
RPC	0.156[a]	0.046	20	1981 – 2002
ALP	0.150[b]	0.050	14	1982 – 2003
LSM	0.109[b]	0.068	7	1982 - 2003

[a] Data from fall surveys only

[b] Data from fall and late winter surveys

Table 2.4. Mean calf-cow ratio based on aerial surveys of the Redrock-Prairie Creek woodland caribou herd, 1981 – 2002. N is the total number of animals observed during the survey.

Survey Date	**N**	**Number Calves**	**Number Cows**
29-Sep-81	78	15	41
29-Sep-82	51	6	29
20-Oct.83	94	17	50
25-Oct-84	98	11	57
3-Oct-85	116	22	61
16-Oct-86	75	9	43
14-Oct-87	65	11	35
9-Nov-88	182	24	103
24-Nov-89	60	7	34
23-Oct-90	305	60	159
28-Oct-91	193	37	101
3-Nov-92	160	20	91
8-Nov-93	226	26	130
2-Nov-95	82	19	41
13-Nov-96	61	11	33
23-Nov-98	62	13	32
17-Sep-99	39	8	20
22-Sep-00	129	6	80
6-Nov-01	122	19	67
21-Oct-02	121	18	67
Total	**2319**	**359**	**1274**
	Mean Calf-Cow Ratio = 0.28		

Table 2.5. Mean calf-cow ratio based on aerial surveys of the A La Peche woodland caribou herd, 1982 – 2003. N is the total number of animals observed during the survey.

Survey Date	N	Number of Calves	Number of Cows
30-Sep-82	32	6	17
21-Oct-83	30	5	16
14-Oct-87	35	6	19
09-Nov-88	90	17	47
30-Oct-91	74	11	41
05-Nov-92	50	3	31
02-Nov-94	126	18	70
01-Nov-95	138	21	76
12-Nov-96	110	12	64
15-Sep-98	99	23	49
13-Sep-99	33	7	17
11-Sep-00	51	7	29
10-Dec-01	114	18	62
18-Mar-03	95	12	54
Total	**1077**	**166**	**592**
		Mean Calf-Cow Ratio = 0.28	

Table 2.6. Mean calf-cow ratio based on aerial surveys of the Little Smoky woodland caribou herd, 1982 – 2003. N is the total number of animals observed during the survey.

Date	N	Number of Calves	Number of Cows
30-Sep-82	18	3	10
19-Oct-83	21	4	11
12-Apr-99	39	2	24
14-Sep-99	28	1	17
29-Mar-01	42	6	36
19-Mar-02	49	6	36
26-Mar-03	52	2	33
Total	**249**	**24**	**167**
		Mean Calf-Cow Ratio = 0.14	

Table 2.7. Mean calf-cow ratios, standard deviation, the number of surveys and years that ratios were obtained for 3 herds of woodland caribou in west-central Alberta.

Herd	**Mean**	**Standard Deviation**	**N Surveys**	**Years**
RPC	0.28	0.09754	20	1981 – 2002
ALP	0.28	0.09222	14	1982 – 2003
LSM	0.14	0.12017	7	1982 - 2003

Table 2.8. Comparative fall recruitment ratios between calves at heel of radiocollared females and composite aerial survey samples obtained from 3 herds of woodland caribou in west-central Alberta, 1995 - 2002. (*n* = sample size).

		Collared Sample		**Survey Sample**				
Year	Herd	*n*	Calf-Cow Ratio	*N*	Calf-Cow Ratio	χ^2	df	P
1998	ALP	20	0.600	49	0.466	0.969	1	0.325
1999	ALP	13	0.538	17	0.409	0.475	1	0.491
2000	ALP	15	0.267	29	0.245	0.034	1	0.854
1995	RPC	15	0.470	41	0.464	0.000	1	0.983
1996	RPC	13	0.380	33	0.338	0.108	1	0.742
1999	RPC	17	0.350	20	0.409	0.087	1	0.769
2000	**RPC**	**18**	**0.280**	**80**	**0.075**	**6.063**	**1**	**0.014***
1999	LSM	13	0.000	17	0.064	0.791	1	0.374
2000	LSM	12	0.170	25	0.079	0.632	1	0.427
2001	LSM	13	0.310	36	0.166	1.169	1	0.280
2002	LSM	12	0.170	36	0.166	0.000	1	1.000

Calf-cow ratios obtained during fall surveys of the LSM herd were higher by a range of 33% - 50% based on calves-at-heel of radiocollared females, with 1 exception (Table 2.9). Similarly, the calf-cow ratio obtained from a fall composite survey of the RPC herd was 64% higher than a late-winter survey conducted the following year (Table 2.10). The mean percent males in samples of calves collected or observed in 5 barren ground or woodland caribou herds averaged 50% (Table 2.11). Calf mortality was most significant in the LSM herd between June and September of 2002 and 2003, but additional mortality occurred during the winters of 2002-2003 and 2003-2004 (Table 2.12).

Table 2.9. A comparison of calf-cow ratios between September and March surveys of radiocollared adult females in the Little Smoky woodland caribou herd, 2000 – 2003.

Date	No. Cows	No. Calves	Calf-Cow Ratio	No. Calves lost
September 13, 2000	12	2	0.167	
March 29, 2001	12	1	0.083	1
September 21 ,2001	12	4	0.333	
March 19, 2002	12	4	0.333	0
September 20, 2002	13	3	0.231	
March 26, 2003	13	2	0.154	1
September 25, 2003	10	3	0.333	
March 24, 2004	10	1	0.100	2

Table 2.10. A comparison of calf-cow ratios between October and April surveys of the Redrock-Prairie Creek woodland caribou herd, 2003 – 2004.

Date	No. Cows[a]	No. Calves	Calf-Cow Ratio	Difference in Calf-Cow Ratio
October 31, 2003	81	28	0.347	
April 7, 2004	56	7	0.125	0.222

[a] Estimated based on number of adults*0.65

Table 2.11. Percent males in samples of calves collected or observed in 5 different barren ground or woodland caribou herds in Canada and Alaska.

% Males	N	Herd	Reference
45%	29	George River, Lab./Que.	Parker 1981
49%	421	Beverly, N.W.T.	Thomas et al. 1989
40%	124	Kaminuriak, N.W.T.	Dauphine 1976
50%	110	Beverly, N.W.T.	Kelsall 1968
53%	62	Beverly, N.W.T.	Kelsall 1968
53%	32	Beverly, N.W.T.	Kelsall 1968
52%	402	Interior Herd, Nfld.	Bergerud 1971
51%	231	Denali, Alaska	Adams et al. 1995
50%	1411	Weighted Mean	All references combined

Table 2.12. A comparison of calves at heel of radiocollared female caribou in the Little Smoky herd, June 2002 – March 2003 and June 2003 – March 2004.

Date	No. Cows	No. Calves	% with Calves
June 1, 2002	14	10	71.4%
June 24, 2002	14	8	57.0%
September 23, 2002	13	3	23.1%
March 26, 2003	13	2	15.4%
June 3, 2003	16	7	43.7%
September 25, 2003	15	3	20.0%
March 24, 2004	10	1	10.0%

Rate of Increase

The caribou herds varied in population growth rate during 1998 – 2003 (Fig. 2.3). The ALP herd appeared to be increasing steadily ($\bar{r}$ = 0.059), the RPC herd was variable, but increasing slightly ($\bar{r}$ = 0.021), and the LSM herd was in steady decline ($\bar{r}$ = - 0.124).

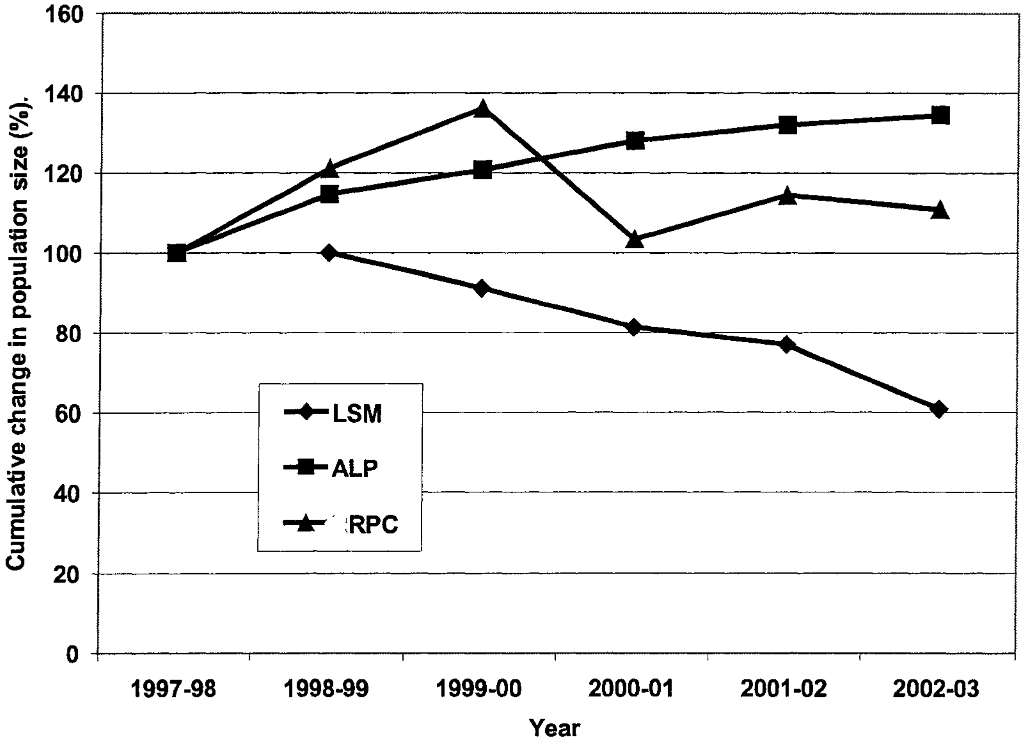


Fig. 2.3. Cumulative change in adult female population size (%) for woodland caribou in the Little Smoky, A La Peche and Redrock/Prairie Creek ranges of west-central Alberta (1997-2003).

Population Projections

The estimated number of adult females (> 1 year) in each herd was derived from the overall population estimate (Table 2.13) minus the estimated number of calves and the estimated number of adult males (> 1 year) (Table 2.14). Stochastic population projections suggested that the A La Peche herd would increase based on adult female survival observed during 1998-2003 and the calf-cow ratios observed 1982-2003 (Fig. 2.4). The Redrock/Prairie Creek herd is projected to decline slowly (Fig. 2.5) and the Little Smoky herd is projected to decline significantly throughout the 20-year projection (Fig. 2.6).

Table 2.13. Estimated number of woodland caribou in 3 herds in west-central Alberta based on the Lincoln-Petersen Index or best total count.

Herd	Date	No. Observed in Survey	Marks Observed / Marks Present	Pop. Est.	95% Lower and Upper Confidence Limits	Liberal Population Estimate[a]
RPC	Nov. 8, 1993	187	12/20	312	157 - 549	350
ALP	Nov. 16, 1998	147	19/20	155	92 - 242	200
LSM	Sept. 25, 2003	60	NA	NA	NA	80

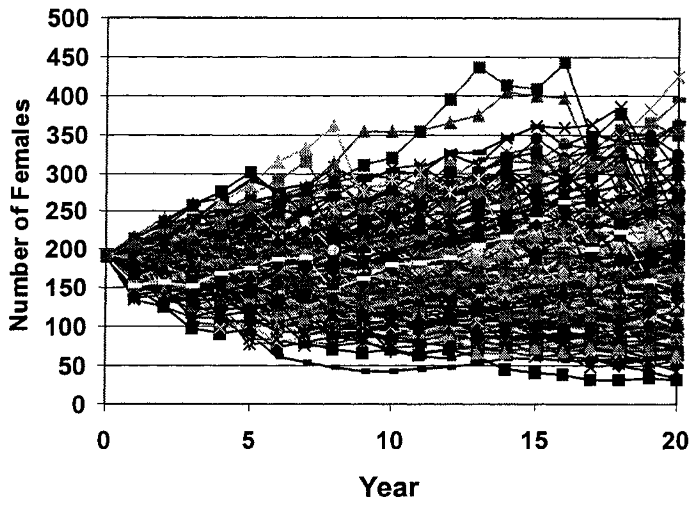
[a] this estimate accounts for the radiocollared sample not including the entire herd.

Table 2.14. Estimated number of adult females in 3 herds of woodland caribou in west-central Alberta, 2003.

Herd	Population Estimate[a]	Mean % Calves[b]	Estimated Adult Cohort (total minus calves).	Proportion Females in Adult Cohort	Adult Female Estimate (total adults × proportion females)
RPC	350	0.156	295	0.65	192
ALP	200	0.150	170	0.65	111
LSM	80	0.109	71	0.65	46

[a] from Table 2.13
[b] from Table 2.3

a



b

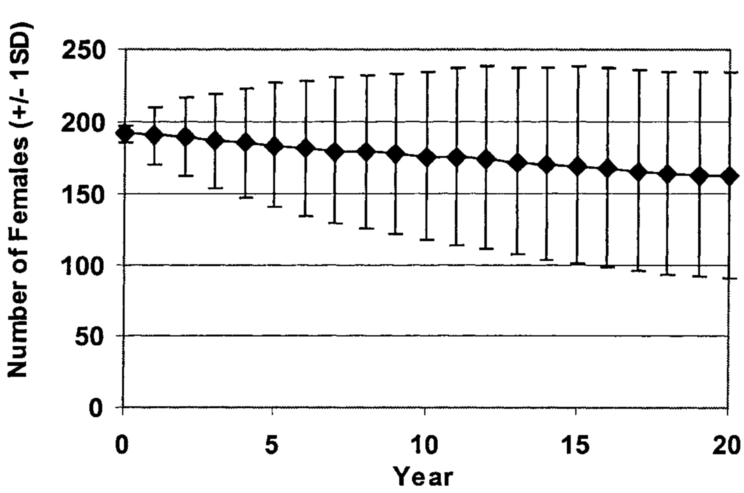
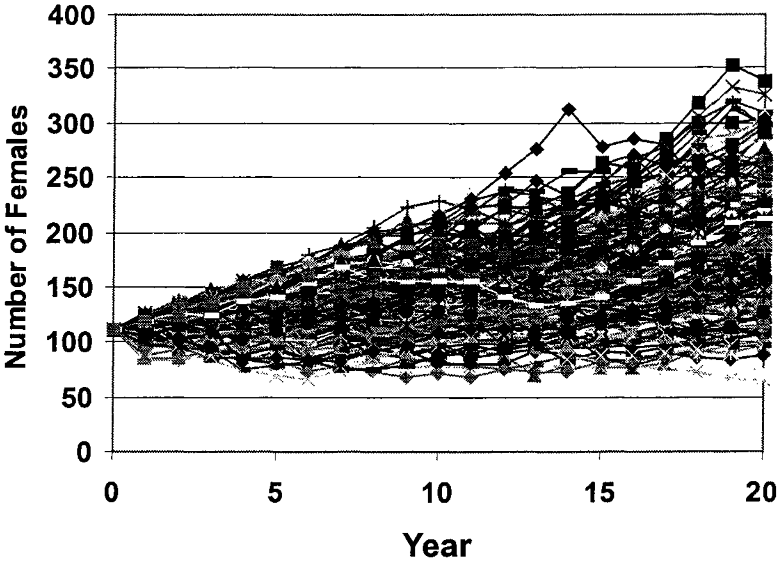


Fig. 2.4. Trajectories of population size projected for 20 years repeated 200 times (a) and the mean (± 1 standard deviation) from 1000 simulation runs (b) of a stochastic model of the Redrock-Prairie Creek woodland caribou herd based on a starting population of 192 females, mean adult survival of 0.873 (SD = 0.100), a mean calf/cow ratio of 0.238 (SD = 0.097) and an assumed calf sex ratio of 50:50. The calf/cow ratio is discounted by 15% to represent overwinter mortality of calves. These projections assume no density dependence.

a



b

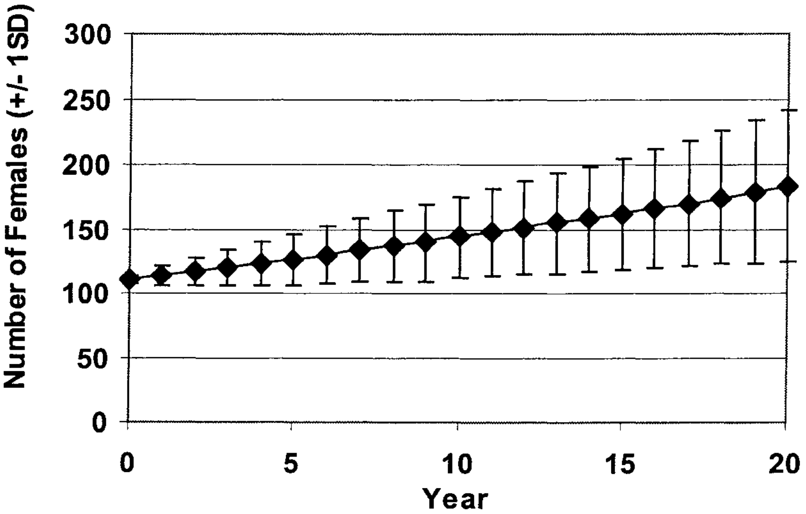


Fig. 2.5. Trajectories of population size projected over 20 years repeated 200 times (a) and the mean (± 1 standard deviation) from 1000 simulation runs (b) of a stochastic model of the A La Peche woodland caribou herd based on a starting population of 111 females, mean adult survival of 0.907 (SD = 0.069), a mean calf/cow ratio of 0.238 (SD = 0.092) and an assumed calf sex ratio of 50:50. The calf/cow ratio is discounted by 15% to represent over winter mortality of calves. These projections assume no density dependence.

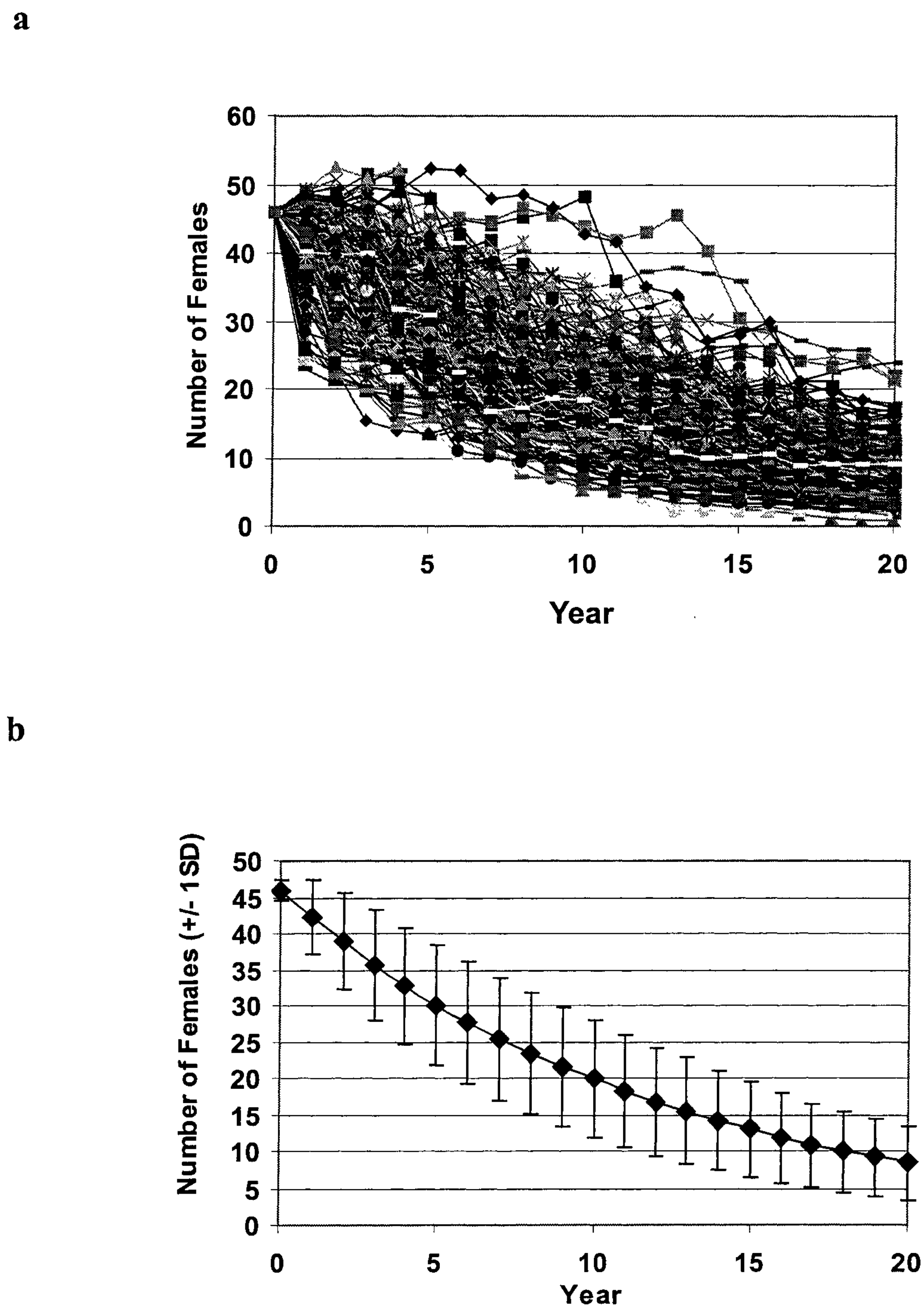


Fig. 2.6. Trajectories of population size projected over 20 years repeated 200 times (a) and the mean (± 1 standard deviation) from 1000 simulation runs (b) of a stochastic model of the Little Smoky woodland caribou herd based on a starting population of 46 females, mean adult female survival of 0.848 (SD = 0.115), a mean calf/cow ratio of 0.140 (SD = 0.120) and an assumed calf sex ratio of 50:50. These projections assume no density dependence.

An alternative approach to the logit transformation of stochastic projections for adult female survival has been proposed which generates survival rates within the unit interval by applying a cumulative distribution function transformation (probit) of the standard normal distribution (Todd and Ng 2001). These authors indicate that this method is particularly effective when survival estimates approach 1. This method maintains the desired means and standard deviations, but results in a beta distribution that does not appear to approximate the range of survival rates documented for woodland caribou (Fig. 2.7). In the examples illustrated in Fig. 2.7 (mean = 0.912), the probit transformation had a similar standard deviation to the original value (0.143 vs 0.139), but the range of survival rates was 0.074 – 1.0 (it would be inconceivable to have 92% mortality of adult females). Moreover, when randomly selecting from this distribution, a survival rate of 1.0 would be chosen 47% of the time. The logit transformation had a lower standard deviation (0.070 vs 0.139), a range of 0.54 – 1.0 and when randomly selecting from this distribution, a survival rate of 1.0 would be chosen 11% of the time. The range of mean annual survival rates between all 3 herds was 0.737 – 1.0. Consequently, the log-odds transformation appears to be more suitable, while accepting that a reduction in variance occurs.

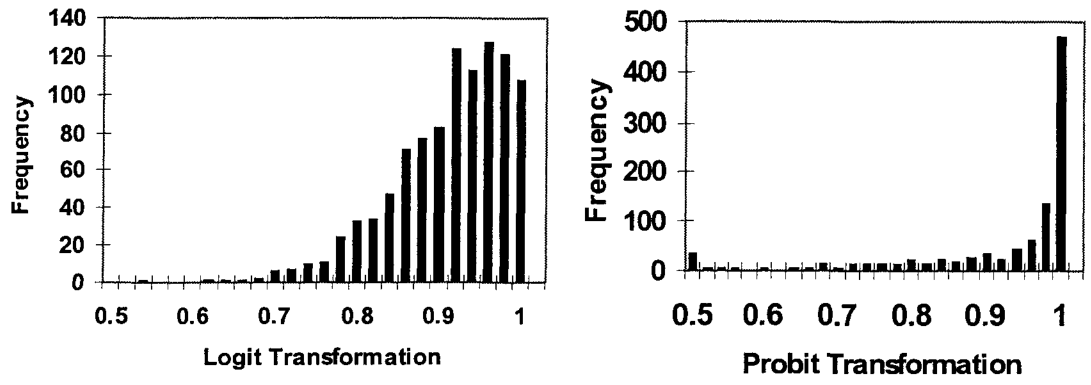


Fig. 2.7. Histograms comparing the inverse log-normal distribution (logit transformation) versus the cumulative distribution (probit transformation) of 1,000 randomly generated adult female survival rates based on a mean of 0.912.

Discussion

Population Demographics

Estimates of adult female survival for the 3 herds (0.848 – 0.919) were within the range reported elsewhere in Alberta (0.85, Fuller and Keith 1981; 0.88: Stuart-Smith *et al.* 1997; 0.86 – 0.93, McLoughlin *et al.* 2003) and adjacent provinces (0.84, in central Saskatchewan, Rettie and Messier 1998; 0.71-0.92, in central British Columbia, Seip 1992). The one exception was the survival rate documented for the 1993-1997 sampling period in the RPC herd (0.947), which slightly exceeded all other reports. Previous survival rates have been calculated for woodland caribou in this study area (0.75, Edmonds 1988), based on both sexes and all herds. Low sample sizes of adult females precluded meaningful comparisons with the present estimates for adult females alone.

The calf-cow ratios obtained during composite aerial surveys (0.14 – 0.28) also were within the range documented for woodland caribou populations with similar adult female survival rates (0.114 – 0.227 in northeastern Alberta, McLoughlin et al. 2003; 0.37 in central British Columbia, Seip 1992; 0.32 in northern British Columbia, Bergerud and Elliott 1998; 0.28 in central Saskatchewan, Rettie and Messier 1998). The majority of the recruitment surveys were conducted when woodland caribou could be readily observed. For the mountain herds (ALP, RPC), most of these surveys were flown in the fall. Additional mortality of calves still could occur before spring, therefore the calf:cow ratios are considered liberal given the timing of these composite surveys (exception - LSM herd where surveys were flown in late spring).

Fall counts during the rut have been questioned as a useful indicator of recruitment in populations of mountain caribou (Hatler 1987), because females with calves were underrepresented in fall rut concentrations. This bias resulted in calf-cow ratios that were higher in samples obtained using radiocollared females than based on visual survey samples. However, composite surveys from this study were not statistically different (with one exception) from calf-cow ratios calculated from the presence of calves at the heel of radiocollared females only (Table 2.10). Moreover, most rut surveys were facilitated by the location of radiocollared animals. Consequently, fall rut surveys are believed to provide a good representation of recruitment if it is recognized that additional calf mortality may occur prior to spring (see Table 2.9, 2.10, and 2.12). To account for estimated mortality prior to spring, an additional 15% reduction in calf-cow ratios was included in the modeling simulations for the 2 mountain herds (ALP, RPC).

The sex ratio of calves was assumed to be 50:50. This assumption is supported by a number of other studies (Kelsall 1968, Bergerud 1971, Dauphine 1976, Adams et al. 1995, Thomas et al. 1989, Parker 1981– Table 2.11). However, the sex ratios of calves could be significantly altered based on the age structure of the female cohort. Older cows have been documented to produce more male calves, while the opposite is true for younger females (Thomas et al. 1989). Consequently, if recruitment was poor for a series of years and older females dominated the reproductive cohort (Festa-Bianchet et. al. 2003) the calf sex ratio could be biased towards males, thereby reducing reproductive capability.

Factors Influencing Population Projections

The population projections did not address the implications of the age structure of each herd on population persistence because this information was not available. This is not considered a significant deficiency of the model because few animals would be alive after 15 years. Boyce (2000) modeled the dynamics of wolves and elk in the greater Yellowstone area, with and without age structure, and he did not find much qualitative difference between the two (also see Dixon et al. 1997). However, senescence isn't factored into the model structure and a few adult females are "allowed" to continue living past 15 years in each estimate. Consequently, the population projections are considered liberal estimates in all examples.

Density-dependent reproduction has been shown in reindeer (*Rangifer tarandus)* (Skogland 1985). Density-dependent survival of juveniles has been demonstrated in caribou (Bergerud 1978) and reindeer (Leader-Williams 1980, Skogland 1984, 1985). Boyce (1992) argues that without the inclusion of a compensatory density-dependent effect on vital rates (i.e. survival and reproductive output), a model's ability to detect the probability of extinction is likely to be overly pessimistic. However, he does suggest that a density-independent model may be acceptable for very small populations or for short-term forecasting. The LSM herd is the smallest of the 3 herds examined and it has the lowest adult survival, the lowest calf/cow ratios and the highest variability in calf/cow ratios, perhaps suggesting the expression of an Allee Effect (Allee 1931; cited in Case 2000). Based on the areal extent of the winter range used, the LSM herd is at the lowest density. This suggests that if compensatory density dependence exists, it is not being expressed (perhaps due to confounding factors such as high predation rates). However, a complication with making comparisons between herds is the non-migratory behavior of the LSM herd. The 2 other herds are migratory and if a composite of both summer and winter ranges were used in calculating density, the LSM herd would be at the highest density.

The LSM herd appears headed for extirpation in the relatively near future. A starting population of 46 females was used in all projections for comparative purposes. Population persistence based on projections is not linear in terms of initial population size (smaller populations are more likely to go extinct than larger ones, given the same demographic variation) (White 2000). As indicated previously, Dzus (2001) suggested that adult survival must exceed 0.85 for population maintenance, given the variability of juvenile survival observed in northern Alberta (0.10-0.40). The population projections appear to be more sensitive to adult female survival than calf recruitment, as would be expected for a long-lived species (see Meyer and Boyce 1994; Fancy et al. 1994; Crête et al. 1996). However, management programs that focus on enhancing calf recruitment may be more practical and effective since there is little variability in adult female survival (see Gaillard et al. 2000). Pregnancy rates for the LSM herd appear normal (94%, N = 19, 1998, unpublished data) suggesting that this factor is not responsible for poor calf recruitment. In 2002-2003 and 2003-2004, radiocollared females were located in June, in mid September and in late March. Results indicated that the bulk of calf mortality occurred during summer (Table 2.12). Calf survival increased between July and September in the Yukon during years of wolf control (Hayes et al. 2003).

The limited work that has occurred to date would suggest that predation is the proximate cause of calf mortality. In Alaska, grizzly bears, wolves and unknown predators (grizzly bears or wolves) accounted for 49, 29 and 16% of calf mortality, respectively (Adams et al. 1995). Intercept feeding of bears and reduction of wolves could temporarily reduce predation rates, although limiting landscape change (habitat loss, fragmentation of habitats, changes in age-structure and composition of the forest) would be the long-term solution.

Variation of most parameters (exceptions being calf sex ratio and proportion of females:males in the adult component) was modeled in a stochastic fashion to provide a better approximation of population growth (Boyce 1977). This provided a means of assessing the potential influence variation in woodland caribou vital rates have on growth-rate variance (Nations and Boyce 1997). If significant variation exists in the sex ratio of calves or if males make up a variable proportion of adults, the effect of variance on population projections would be magnified.

A log-odds (logistic or logit) transformation of adult female survival was applied to restrict these rates to the unit interval (≤ 1.0) (Burnham et al. 1987); however, this resulted in inverse log-normal survival values lower than the original. These transformed values were increased until the mean survival rate was identical to the original values. The log-odds transformation also resulted in lower estimates of variance, but these are considered a better approximation than the assumption of a normal distribution, which is necessarily truncated at 1.0 (Burnham et al. 1987).

Projecting stochastic processes into the future is problematic in terms of errors that are magnified over time, assumptions regarding the lack of change in mechanisms affecting vital rates, and the selection of the correct structural model (Boyce 1992). The land-use trajectories in west-central Alberta are speculative, although it is almost certain that there will be continued industrial activity for the foreseeable future. The projection model could be enhanced in a number of areas, e.g., obtaining a variance estimate for the calf sex ratio, a variance estimate for the adult sex ratio, and relating vital rates to stochastic weather events. Because of uncertainty in the estimates of vital rates, projections should be used only to examine woodland caribou population persistence over short intervals (5-10 years) (Ludwig 1999, Fieberg and Ellner 2000). However, I modeled woodland caribou response over 20 years to include the standard planning horizon for most timber management scenarios.

The approach used in this analysis provides a means of examining the potential influence variation within woodland caribou vital rates has on growth-rate variance, an important population management consideration relative to extinction probabilities (Nations and Boyce 1997). This analysis contributes to a larger-scale assessment of sustainable forest management in Alberta by examining the population viability of a species that appears to be sensitive to changes in the amount, fragmentation and location of old forests.

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CHAPTER 3. EFFECTS OF LANDSCAPE CHANGE ON SURVIVAL OF ADULT FEMALE WOODLAND CARIBOU IN WEST-CENTRAL ALBERTA.

INTRODUCTION

The major industrial activities that result in landscape change of woodland caribou range in west-central Alberta are petroleum exploration and development, and timber harvest. Both activities require a network of roads, which fragments caribou habitats. Petroleum activity also fragments habitats through the development of seismograph lines, pipelines, powerlines and wellsites. In addition to fragmentation, timber harvesting directly removes habitat through clearcutting the forest overstory. Once the canopy is removed, lichens (the main winter diet of caribou; Thomas et al. 1996) desiccate and do not re-establish for many decades (Harris 1996, Kranrod 1996).

Effects of petroleum activity on woodland caribou populations

Oil and gas exploration and development has the potential to negatively affect caribou through disturbance, habitat loss and fragmentation (Fahrig 1997). The longest-term examination of the response of barren-ground caribou (*R. t. groenlandicus*) to oil and gas activity has taken place on Alaska's North Slope over the last 25 years. Cameron et al. (1979) initially documented female caribou with calves avoiding the TransAlaska Pipeline (1.8 m above ground) and this was still apparent 5 years later (Dau and Cameron 1986). Similarly, at Prudhoe Bay, barren-ground caribou densities and the proportion of calves in composition counts declined within 1 km of an oil field access road when it was constructed through a calving concentration area. At the same time, there was a corresponding increase in densities and the proportion of calves at a distance of 5 – 6 km from the road (Cameron et al. 1992). Subsequently, a radiotelemetry study revealed a significantly lower abundance and reduced movements of female caribou within the intensively developed oil field relative to more distal areas (Cameron et al. 1995). Possible consequences of this displacement were thought to include increased competition for forage, reduced productivity and increased risk of predation (Nelleman and Cameron 1998). However, individual pipelines do not appear to present significant barriers to migrating barren-ground caribou (Carruthers and Jakimchuk 1987), yet, pipelines associated with roads may have a cumulative effect (Smith and Cameron 1985, Curatolo and Murphy 1986). Despite the documented avoidance of intense oil field development, Cronin et al. (1998) point out that this Central Arctic herd increased during the same period and they argue that resource extraction and wildlife populations can be compatible if managed properly.

Research concerning the effects of oil and gas field activity on woodland caribou has received attention only recently. Simulated oil and gas exploration activities (i.e. loud noise) in northeastern Alberta resulted in greater mean movement rates and habitat boundaries being crossed more frequently by "disturbed" woodland caribou than control animals, but feeding patterns were similar between the two groups (Bradshaw et al. 1997). These authors suggested that increased movements might have a higher energy cost and later modeled the cumulative influence of disturbance caused by exploration activities experienced in northern Alberta and indicated the potential for an effect on individual energy loss during winter (Bradshaw et al. 1998). The indirect effect of oil and gas access has been linked to increased efficiency of wolf travel and hunting success on woodland caribou (James 1999, James and Stuart-Smith 2000). Woodland caribou have been documented avoiding roads and seismic lines by 250 m and wellsites by up to 1000 m in northern Alberta (Dyer et al. 2001). These authors also indicated that roads provided semi-permeable barriers to movement, while seismic lines did not appear to present a barrier. A similar study in west-central Alberta (Oberg et al. 2003) found no avoidance of seismic lines (possibly because of the older vintage of lines), but did observe avoidance of roads. Given the intensive and extensive nature of petroleum activity in west-central Alberta, the potential for the indirect effect of linear corridors to negatively influence vital rates of woodland caribou continues to be worthy of further examination.

Effects of timber harvest on woodland caribou populations

Timber harvest has the potential to negatively affect woodland caribou by directly eliminating habitat for the period of time that it takes for lichens to re-establish (Kranrod 1996). Indirectly, timber harvest can affect woodland caribou by enhancing habitat for other ungulates sought by wolves (alternative prey – see section on predation in Chapter 1, Introduction). A number of studies have implicated timber harvest with declines of woodland caribou populations (Bergerud 1974, Bloomfield 1980, Hristienko 1985, Cumming 1992, Rettie and Messier 1998). A significant decline in a sub-population of the *mountain-arboreal* ecotype of woodland caribou was documented in an area of extensive timber harvest in southern British Columbia (Kinley and Apps 2001). In this study, caribou mortality was higher in the portion of the study area that had a greater proportion of forests ≤ 40 years, a higher road density and was more fragmented (Kinley and Apps 2001).

Short-term changes in movements and distribution of woodland caribou have been documented in response to active timber harvest on summer range in Newfoundland (Chubbs et al. 1993) and to winter haul-road activity in Ontario (Cumming 1998). In the RPC herd range of this study, woodland caribou were observed at least 540 m further from timber cutblocks than would be expected and they appeared to avoid active cutting areas on winter range by 1200 m over a 15-year period (Smith et al. 2000). In the same area, Stepaniuk (1997) did not record woodland caribou crossing cutblocks during 2 years of backtracking. [Since that time, some animals outfitted with GPS collars have been using the remaining unharvested timber blocks within the cutblock matrix (WCACSC unpubl. data)].

In this chapter, I performed analysis at 2 scales to model survival of adult female woodland caribou relative to oil and gas activity and timber harvest. Females from 3 woodland caribou herds in west-central Alberta were monitored over a 20-year period (1981 – 2001). Industrial activity was examined at the home range and the telemetry location scale. My hypothesis was that adult female woodland caribou survival was negatively correlated with an increased density of industrial activity. In particular, the cumulative effects of these 2 industrial activities were thought to be most detrimental.

Methods

Radiocollared Sample

A total of 141 adult female woodland caribou were captured by helicopter-borne netgun (Barrett et al. 1982) from 3 ranges in west-central Alberta between 1981 and 2001. Animals were either fitted with very high frequency (VHF) or Global Positioning System (GPS) radio-collars (Lotek Engineering Inc., Newmarket, Ontario, Canada). An effort was made to capture all age groups of adult females (≥ 1 year), although yearling females were underrepresented in the collared sample. Survival of caribou was monitored monthly by aerial tracking during the winter and monthly - bimonthly during the summer. The objective was to maintain a sample of at least 20 adult females in each woodland caribou herd range (Mean = 24.1, SD = 7.5, n = 15 herd years) by redeploying radiocollars where mortalities or collar failures were detected. Where possible, cause of death was determined by recovering collars as soon as mercury switches indicated prolonged (>8 hrs) lack of animal activity.

Geographic Information System Analysis

Calculation of Landscape Change

Linear disturbance changes were tracked for each animal and for each herd by examining a series of digitally referenced ortho-photos (1981, 1985, 1991, 1995), and Indian Remote Sensing (IRS) Satellite Images (1998, 2001) covering each of the 3 caribou ranges. (Complete coverage of all ranges was not possible during all periods). The 4 Forest Management Agreement holders with tenure on the caribou ranges provided digital timber-harvest information for the entire study period on an annual basis (1981- 2001). These data were stored as land-use layers in a Geographic Information System.

Landscape change was tracked at 2 different scales. At the home range scale, a minimum convex polygon (100% MCP; Mohr 1947) was generated for each animal by plotting all telemetry locations. This scale was used to represent the cumulative change in risk related to industrial activity over all seasons during the time the animal was monitored (Mean = 585 km^2, Range = 6.5 – 2190 km^2). For the non-migratory, boreal herd (LSM) 1 MCP was generated incorporating all seasons. For the 2 migratory herds (ALP, RPC), I generated only a winter MCP since most summer locations were considered discontinuous intervals of risk and there were no industrial activities in the protected areas used as summer range.

At the individual-location scale, each animal's location was buffered in the GIS by 3875 m. This buffer size represented the 80th percentile of the weekly movements calculated after pooling a sample of 48 GPS-collared adult female caribou (N = 53,789 locations, 1999 – 2003; Fig. 3.1). Each buffered location is designed to represent the landscape that the animal might have used during the interval since the previous location (changes in risk). Additionally, the 3875 m buffer was a compromise between exceeding the mean daily movement distance (2800 m; Szkorupa 2002) while not overestimating the area of use given that daily travel was not necessarily in a straight line. The technique of using a percentile of travel distances for a telemetry interval has been used to define available habitat for caribou in other studies (Servheen and Lyon 1989, Johnson 2000, Rettie and

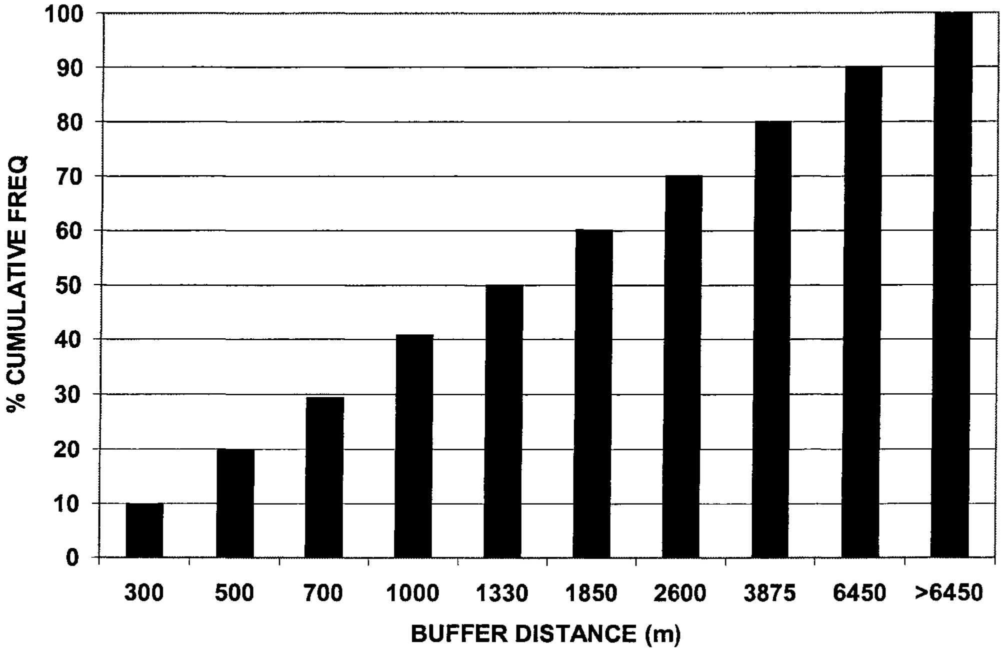

Fig. 3.1. Cumulative frequency of mountain and boreal caribou buffers (combined) illustrating the percent of subsequent locations within 1 week contained within increasing buffer size in west-central Alberta (N = 53789 GPS locations of 48 adult female woodland caribou).

Messier 2000, Apps et al. 2001, Szkorupa 2002) and polar bears (Arthur et al. 1996).

Based on this digital record of landscape changes, the density (m/km^2) of seismic lines, pipelines, powerlines and roads was calculated within the MCP and the buffered location of each animal and summarized by each disturbance type for each period of photo/image availability. New disturbance was estimated by comparing each new photo to the previous one for the same area. The cumulative total of each type of linear disturbance was then calculated. Additionally, all types of linear disturbances were summed to calculate the variable *all linear* *disturbances.* The amount of timber harvest within each MCP was calculated annually and expressed as a density (ha/ km^2) in order to standardize change across home ranges of different sizes. The cumulative total of the density of timber harvest was also calculated.

Industrial activity was tracked annually (for timber harvest) or at 5-year intervals (for linear disturbance), while adult female survival was monitored at more frequent intervals ($\leq$ 45 days). To provide an estimate of change in each land-use variable from one radiotelemetry location to the next, linear regression analysis of disturbance type (dependent variable) relative to date of study (independent variable) was performed. For example, a linear regression for an animal with a MCP road density of 200 m/km^2 on study day 500 and a road density of 300 m/ km^2 on study day 2325 (5 years later) would be calculated with the road densities as dependent variable inputs and the study days as the independent variable inputs. The actual study day of each telemetry location would be entered into the linear regression equation to estimate the road density within the MCP at that time. This approach assumed that industrial activity developed in a linear fashion. Since each photo/image was generally obtained during summer, the annual date of study was assigned as July 31 of that year. The same approach was used for cumulative timber harvest. All GIS measurements were made in ArcInfo GIS® (Version 8.3).

Modeling variation in survival based on landscape condition

If management programs are to be successful, the feasibility of enhancing vital rates that drive population recovery must be ascertained. One means of tracking the implications of habitat change and environmental factors on survival is to use the Cox regression method (White and Garrott 1990). This method uses the "hazard function" (an animals instantaneous probability of death), which can be modeled as

[Eqn. 8] $$h(t) = h_o(t) \cdot \exp(\beta_1 x_1 + \beta_2 x_2 + \ldots + \beta_p x_p)$$

(White and Garrott 1990), where $h(t)$ represents the death rate at time t, $h_o(t)$ is the baseline hazard function (constant: dependent only on time), β's are regression coefficients of independent covariates that could influence survival and x's are the independent covariates. The exponential term guarantees that h_o is positive for any β (S-PLUS-Statistical Sciences 1993).

Variables that could influence female woodland caribou survival were examined through the application of the Andersen-Gill (A-G) model (Andersen and Gill 1982, Therneau and Grambsch 2000), which incorporates Cox regression (or multiplicative hazards). This technique is similar to the Cox Regression; however, it also allows staggered entry of individuals. Like Cox Regression, it can use time-dependent covariates and discontinuous intervals of risk (i.e. disappearance, but eventual location of radiocollared individuals). As a non-parametric technique, it is more robust than similar parametric tests. This technique has been used primarily in medical research (Hosmer and Lemeshow 1999, Therneau and Grambsch 2000), but has received recent attention for wildlife applications (Boyce et al. 2001, Johnson et al. 2004).

Data were updated at each telemetry location. The time variable (or interval) was the number of days since the last telemetry location. All intervals > 45 days were truncated and considered as discontinuous intervals of risk because 45 days corresponds to the estimated maximum length of time a migratory female woodland caribou would take to move between summer and winter range, which would result in a dramatic change in the risk factors. The 45-day interval limit resulted in most summer locations for the migratory mountain herd animals being considered discontinuous intervals of risk (because they were not monitored during this extended period of time and industrial activity may have changed). Consequently, for the migratory mountain herds, only the industrial activity densities on winter range were entered in the A-G model. (Values of 0 for linear disturbance and timber harvest were entered for the few summer locations of mountain animals where intervals were ≤ 45 days).

The start and endpoint of each telemetry interval was the number of days since the capture date of the very first study-animal collared (February 25, 1981). (For example, if the 20th animal was collared one year after the first animal and then located 10 days later, its start point and end point would be 366 and 376 days, respectively). Each individual animal data set was stratified by herd. A total of 6 continuous predictor variables were included in the analysis. These were measured at the end of each individual's telemetry interval:

1) The cumulative area logged (ha/km^2) within each individual's minimum convex polygon (MCP).

2) The cumulative amount of roads (m/km$^{2)}$) within each individual MCP **or**

3) The cumulative total of all linear disturbances (roads, pipelines, powerlines, seismic lines, railroad lines combined) within each individual MCP.

4) The cumulative percent of area logged within a 3875 m circular buffer around each individual location (ha/km^2).

5) The cumulative amount of roads (m/km$^{2)}$) within a 3875 m buffer around each individual location **or**

6) The cumulative total of all linear disturbances (roads, pipelines, powerlines, seismic lines, railroad lines combined) within a 3875 m buffer around each individual location.

Because roads were included in the variable set for *all linear disturbance*, the analysis considered one or the other to avoid problems of multicollinearity (roads were highly correlated with the total of other linear disturbances).

Survival was modeled as a function of these variables. Initially, all variables that were considered a priori to influence survival of adult female woodland caribou were included in a global model. To assess model fit, to identify poorly predicted subjects and influential points, and to test the principal assumption of the multiplicative hazards model (that the hazard or risk ratio is proportional over time), I plotted the scaled Schoenfeld residuals for each variable against time (Therneau and Grambsch 2000). These plots were visually and statistically examined. A line can be fitted to the residuals and tested for a nonzero slope. If the fitted line was significantly different from zero, the proportional hazards assumption was rejected. The variables which violated the proportional hazards assumption were withdrawn from any further analysis.

Candidate models were ranked by Akaike Information Criteria (AIC). Due to the low sample size of woodland caribou deaths (*n*) relative to model parameters (*K*) (where $n/K < 40$), I used the corrected AIC formula ($\mathrm{AIC_c}$) (Anderson et al. 2000). Differences in $\mathrm{AIC_c}$ (Δ) were used to identify the most parsimonious model and to compare between models. $\mathrm{AIC_c}$ weights (w) were calculated and interpreted as the approximate probability that each model is the best within the set of models examined and to generate multi-model averaged values for β coefficients and unconditional sampling variance (Anderson et al. 2000, Johnson et al. 2004). To test for non-linear effects of each variable on survival, I compared a model with a linear term to one with a quadratic term (variable + variable^2) and I included the model with the quadratic term if the $\mathrm{AIC_c}$ score was reduced by > 2 points. All statistical analyses were performed with S-Plus 2000 (Release 2) and Microsoft ® Excel 97 SR-2.

RESULTS

Cumulative Changes in Industrial Activity over Time

At the initiation of the study (1981), industrial activity had begun in all 3 woodland caribou herds (Fig's. 3.2 – 3.4), but linear disturbances were much more significant in the LSM range compared to the other 2 herds (Fig. 3.2 and 3.3). In contrast, timber harvesting activity was more prevalent in the 2 mountain herd winter ranges (ALP, RPC) than the LSM herd from 1981 –1991, after which the density of cumulative timber harvest experienced by this boreal herd surpassed that of its mountain counterparts (Fig. 3.4). Both linear disturbances and timber harvesting activities have created an exceedingly greater "footprint" in the LSM range compared to the ALP and RPC herds since that time.

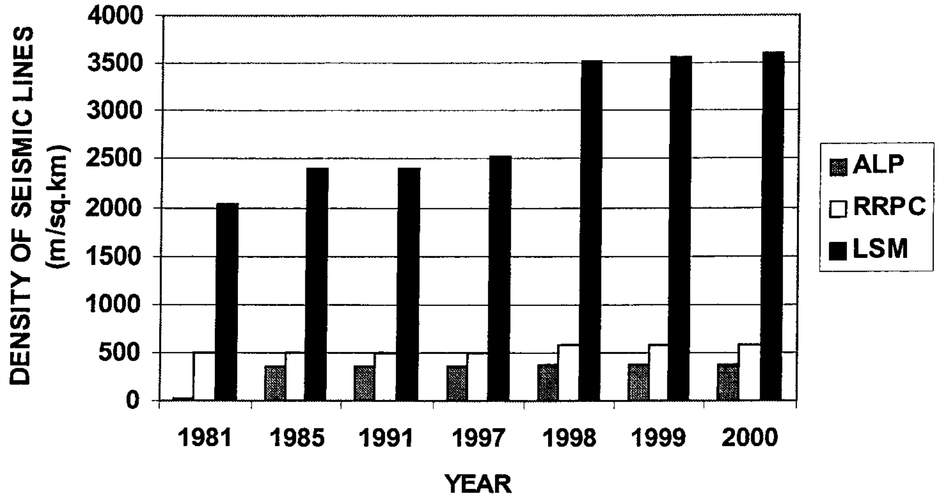

Fig. 3.2. Density of seismic lines (m/km^2) within specific woodland caribou herd Minimum Convex Polygons in west-central Alberta, 1981-2001. (ALP = A La Peche, RPC = Redrock/Prairie Creek, LSM = Little Smoky).

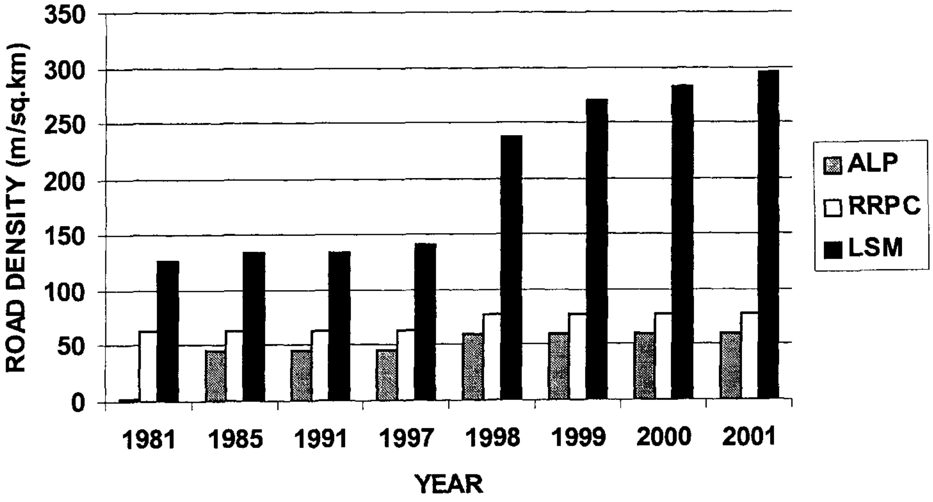

Fig. 3.3. Density of roads (m/km^2) within specific woodland caribou herd Minimum Convex Polygons in west-central Alberta, 1981-2001. (ALP = A La Peche, RPC = Redrock/Prairie Creek, LSM = Little Smoky).

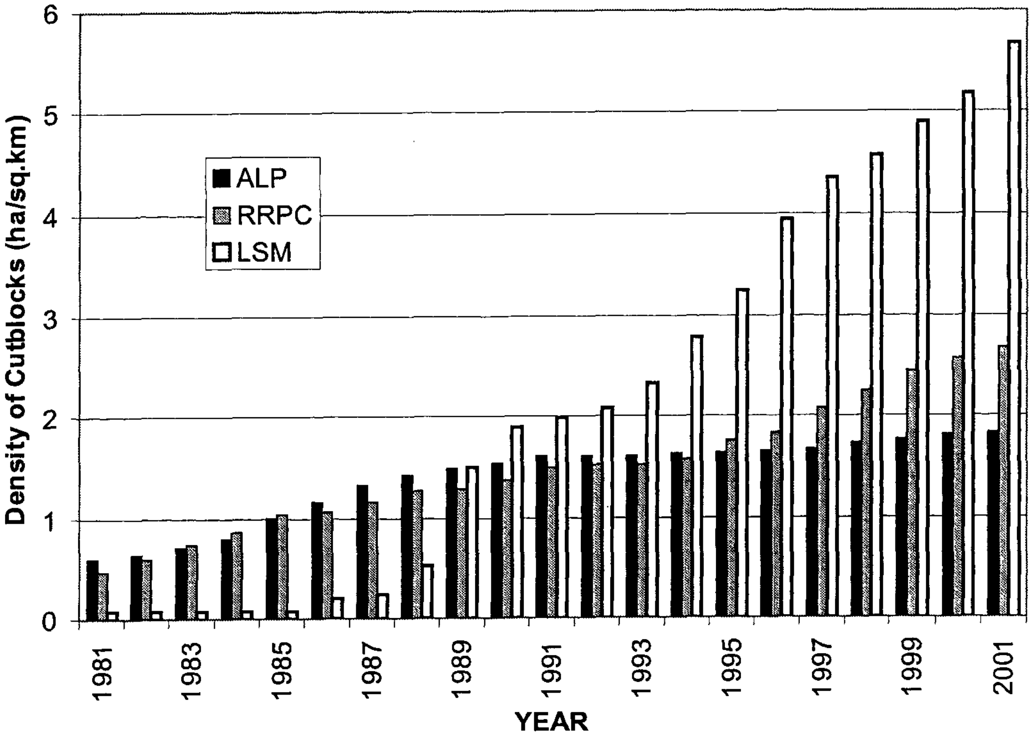

Fig. 3.4. Density of cutblocks (ha/km^2) within specific woodland caribou herd minimum convex polygons in west-central Alberta, 1981-2001. (ALP = A La Peche, RPC = Redrock/Prairie Creek, LSM = Little Smoky).

Andersen-Gill Models

A total of 141 adult female woodland caribou were radiocollared in west-central Alberta between 1981 and 2001. Most were fitted with standard VHF collars; however, a total of 26 Global Positioning System collars were placed on females between 1998 and 2001. Of the radiocollared animals, mortalities were recorded in 48 instances. Both VHF and GPS collared caribou were monitored at different intervals depending on the specific questions being asked in each study period (Median = 14 days, Range = 1 – 45 days) (Edmonds and Bloomfield 1984, Edmonds 1988, Edmonds and Smith 1991, Smith et al. 2000).

A total of 3342 telemetry locations of these individuals were made over the 20 years of monitoring. The variables of Cumulative Timber Harvest, Road Density or All Linear Disturbances within each MCP and Cumulative Timber Harvest, Road Density or All Linear Disturbance Density within each location buffer (Table 3.1) were examined in the first "global" model. The variables *All Linear Disturbances* within each MCP and *Roads* and *All Linear Disturbances* within location buffers violated the proportional hazards assumption (Fig. 3.5 and Fig. 3.6) and received no further consideration for model development. I used the remaining covariates within MCP's (*Roads, Timber Harvest and Timber Harvest*^{2}) and *Timber Harvest within buffers* to generate a second "global" model. The variable *Timber Harvest within buffers* did not approach significance (P = 0.82) in the second "global" model. Consequently, I used the remaining covariates within MCP's (*Roads, Timber Harvest and Timber Harvest*^{2}) to generate 3 additional models. The model that included the variables of Timber Harvest, Roads and Timber Harvest^2 within MCP's had the lowest AIC$_c$ score and was statistically significant ($\chi^2 = 10.4$, 3 df, $P = 0.0157$; Table 3.2).

Table 3-1. Independent variables and description of industrial activity variables used to describe variation in survival of adult female woodland caribou from 3 herds in west-central Alberta, 1981-2001.

Variable	Description
	Within MCP
TimbrHrv	Density of cumulative timber harvest within the MCP (ha/km^2).
Roads	Density of roads within the MCP (m/km^2).
Cutline Density	Density of seismograph lines within the MCP (m/km^2).
Pipeline Density	Density of pipelines, powerlines and rail lines within the MCP (m/km^2).
All Linear Disturbance	Density of all linear disturbances within the MCP (m/km^2).
	Within Buffered Locations
Cumulative Timber Harvest	Density of cumulative timber harvest within a 3875 m buffer of each individual location (ha/km^2).
Road Density	Density of roads within a 3875 m buffer of each individual location (m/km^2).
Cutline Density	Density of seismograph lines within a 3875 m buffer of each individual location (m/km^2).
Pipeline Density	Density of pipelines, powerlines and rail lines within a 3875 m buffer of each individual location (m/km^2).
All Linear Disturbance	Density of all linear disturbances within a 3875 m buffer of each individual location (m/km^2).

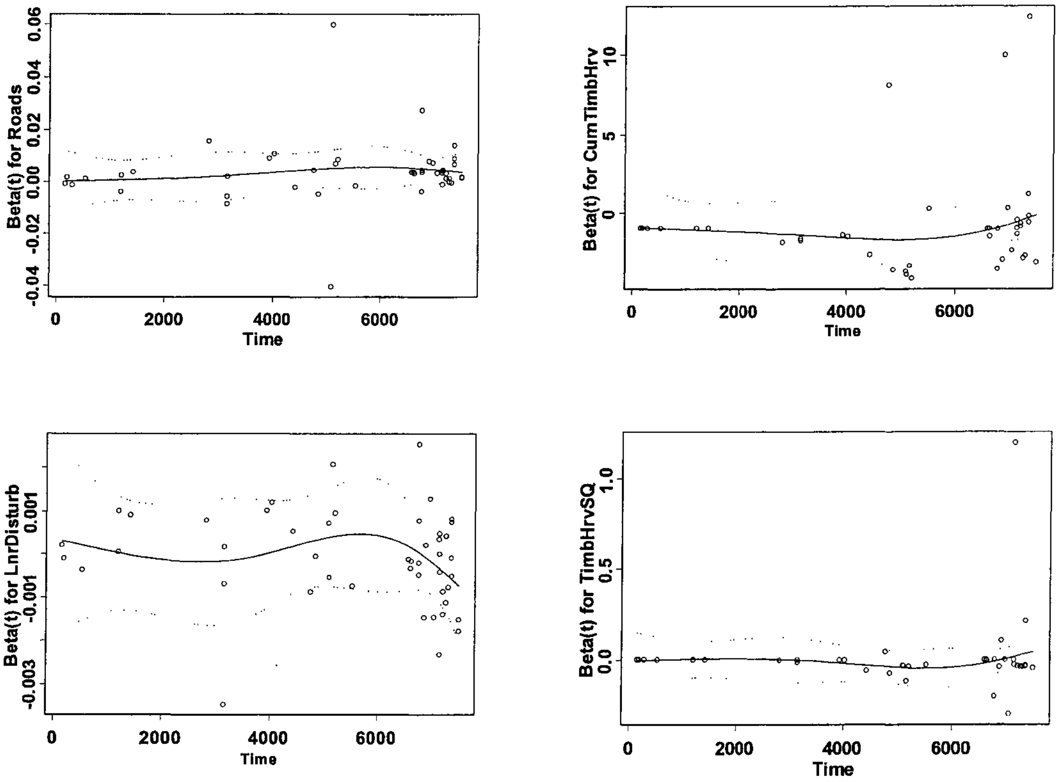

Fig. 3.5. Scaled Schoenfeld residuals vs time of study for *beta coefficients* of road density, cutblocks, all linear features and cutblocks^2 within woodland caribou MCP's.

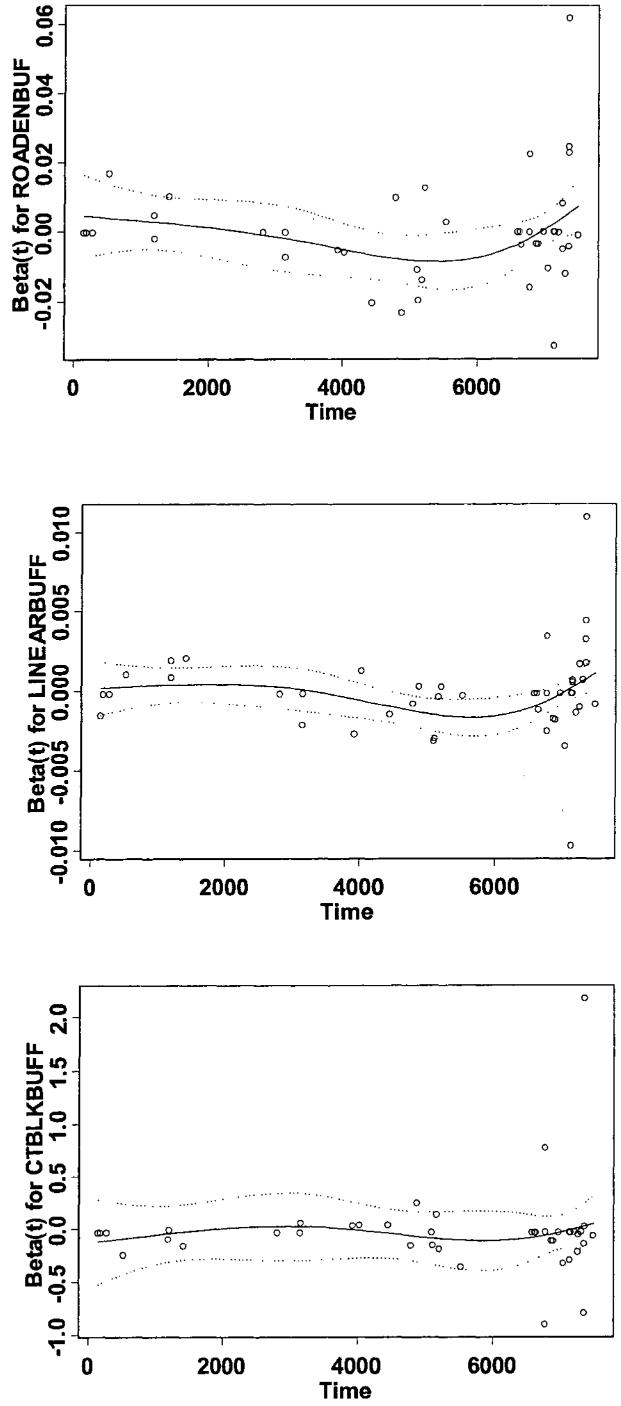

Fig. 3.6. Scaled Schoenfeld residuals vs time of study for *beta coefficients* of road density, all linear features and cutblocks within 3875 m buffers around woodland caribou locations.

Table 3.2. Candidate Andersen-Gill models, number of parameters (*k*), Log-Likelihoods, AIC$_c$ scores, differences among AIC$_c$ scores (Δ) and AIC$_c$ weights (*w*) of industrial disturbance variables for 3 herds of woodland caribou in west-central Alberta, 1981 – 2001. Akaike information criteria (AIC$_c$) were used to order models relative to most parsimonious fit.

Model	*k*	Log-Likelihood	AIC$_c$	AIC$_c$ Δ	AIC$_c$ *w*
Roads + TimbrHrv +TimbrHrv^2	3	-113.846	234.24	0.00	0.729
Roads + TimbrHrv	2	-116.504	237.27	3.04	0.160
Roads	1	-117.957	238.00	3.76	0.111

The exponentiated linear coefficients from these models can be interpreted as risk ratios relative to each covariates influence on adult female woodland caribou survival. To address multi-model inference (Anderson et al. 2000), the coefficient for each covariate was averaged across all candidate models in which it had been included (Table 3.3). The coefficient for Roads suggests that an increase in road

Table 3-3. Original coefficients, adjusted coefficients, standard errors and 95% confidence intervals of Andersen-Gill models representing the effects of industrial activity on survival of adult female woodland caribou from 3 herds in west-central Alberta, 1981 – 2001. AIC$_c$ weights were used to adjust coefficients and variance for model selection uncertainty.

Variable	Original Coefficient	Adjusted Coefficient	SE	Confidence Interval
Roads	0.004	0.004	0.002	0.000 - 0.007
TimbHrv	-0.958	-0.831	0.362	- 1.541 - - 0.120
TimbHrv^2	0.124	0.124	0.055	-0.017 - 0.231

density (within an animals MCP) of 100 m/km^2 would result in a 1.5 (=(exp[100 × 0.004])) times greater risk of dying compared to an adult female that was not confronted by this increase in road density within her home range. When statistically controlling for roads, the quadratic relationship with timber harvest was J-shaped and indicated that adult female woodland caribou experienced lower risks of dying in the initial stages of timber harvest. However, this relationship reversed once cutblock density exceeded 3.4 ha/km^2 (3.4% of the MCP) and a risk threshold was reached when 6.8% (6.8 ha/km^2) of an individual animals MCP was logged (Fig.3.7).

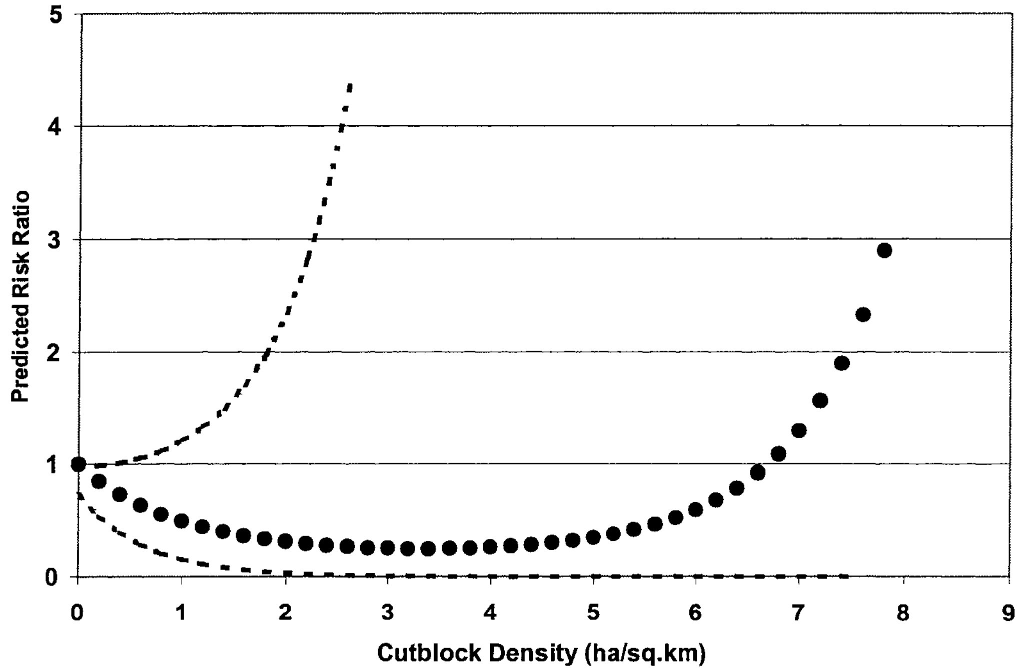

Fig. 3.7. Relationship between the predicted risk ratio of adult female woodland caribou survival and cutblock density within the animals MCP. Hatched lines indicate 95% confidence intervals.

DISCUSSION

The results of this analysis support the hypothesis that cumulative road and cutblock densities within home ranges of woodland caribou are correlated with adult female survival. The indirect mechanisms proposed as being negatively correlated with caribou survival include an increase in predator hunting efficiency (James 1999, James and Stuart-Smith 2000), an increase in alternate prey resulting in higher densities of predators (Seip 1998, Kinley and Apps 2001), the avoidance of industrial activity resulting in reduced home ranges, which might result in increased vulnerability to predators (i.e. less ability to space away; Bergerud et al. 1983, Bergerud and Page 1987, Smith et al. 2000, Dyer et al. 2001) and increased energy expenditure resulting from this avoidance (Bradshaw et al. 1998). In general, all these factors would tend to operate at the home-range scale rather than the individual radiotelemetry location. Studies that have examined woodland caribou behavior at multiple scales suggest that the ability of woodland caribou to avoid limiting factors (e.g., landscapes with high probability of predation) should be most obvious at coarse spatial scales (Rettie and Messier 2000, Johnson et al. 2002).

Minimizing the development of new corridors and making existing corridors in caribou habitat unsuitable as travel routes are recommended to reduce the impacts of industrial development on caribou populations (James and Stuart-Smith 2000). Until the demographic response of caribou herds to industrial activity is known, a conservative approach would be to set limits to industrial development in woodland caribou habitat (Dyer et al. 2001). Limiting the amount and tenure of petroleum access in any given winter, remote production of wellsites (satellite controlled), consolidation of multi-company seismograph programs and using heliportable seismograph operations to reduce the development of access are all interim measures stressed in the West-central Alberta Caribou Standing Committee Guidelines for Industrial Activity.

The hazard or risk ratio was not proportional over time for the variables measured within buffered locations (road density, all linear disturbance, cutblocks). This was also true for the variable *all linear disturbance* measured within MCP's. Apparently, the influence of these covariates was not consistent across the duration of the study. A possible reason for this violation of the proportional hazard assumption might be the high variability of each of these covariates. In terms of the buffered locations, the scale of buffer may have influenced this outcome. The A-G model is designed to detect changes in the probability of survival and as such, landscape scale effects rather than local effects, are much more likely to be measurable. Furthermore, woodland caribou have been exposed to predation for many centuries before man-caused changes to the landscape had ever occurred and, of course, still die in protected areas where human influence is arguably minimal. This factor would reduce the probability that industrial activities in the vicinity of each telemetry location would be significant, given that 2 of the 3 study herds spend their summers (and for the last 6 years, the majority of the A La Peche herd has spent the entire year) in protected wilderness areas. Additionally, although the 3875 m distance used to buffer each telemetry location represented the 80th percentile of the weekly movements pooled across a sample of 48 GPS-collared adult female caribou (N = 53,789 locations, 1999 – 2003), this buffer may have not been the appropriate scale to detect local effects. Given the variability in monitoring intervals and individual animal behaviour, choosing the appropriate buffer size presents a challenge when trying to select the appropriate scale to measure landscape metrics (Porter and Church 1987, McLean et al. 1998, Apps et al. 2001, Rettie and Messier 2000, Johnson et al. 2002).

Although I had *a priori* expectations that correlations between densities of industrial activity and adult female woodland caribou survival might not be linear, the positive relationship between low levels of cumulative timber harvest and survival was unexpected. (This relationship reverses itself at cutblock densities of 3.4 ha/km^2 or greater). One possible explanation is suggested by earlier work on 1 of the study herds (Smith et al. 2000). Woodland caribou in the Redrock/Prairie Creek herd were observed further from timber harvesting activity (1200 m) than would be expected by chance (Smith et al. 2000). This may have resulted in beneficial "spacing away" (Bergerud et al. 1983, Bergerud and Page 1987, James and Stuart-Smith 2000, Rettie and Messier 2000, Stuart-Smith et al. 1997) from alternate prey expected to "pioneer" new cutblocks. Moose are the primary alternate prey found in this area. Increases in moose density have been linked to increased amounts of logging. If predators focused on these alternate prey in the early stages of timber harvest, the combination of caribou "avoiding" timber harvesting areas and predators "selecting" cutblock areas within caribou herd ranges (e.g. wolves: see Kuzyk 2002) might have increased the probability of survival in the early stages of timber harvest. As the density of timber harvest increases, the relationship seems to have the opposite effect. The "lag effect" of predator populations increasing in response to an increase in the alternate prey base would probably not be expressed as an increase in adult female woodland caribou mortality for a period of time that could correspond to higher cumulative densities of timber harvest. At some point the ability of woodland caribou to "space away" from cutblocks (and therefore alternate prey/more wolves) would begin to decrease as cumulative timber harvest increased. This would also be expressed as a reduction in individual home range size as documented for the RPC herd (Smith et al. 2000). In combination with decreasing lichen abundance (through removal of older stands by timber harvest), an increase might be anticipated in the probability of encountering predators as the range of "effective habitat for caribou" was reduced by additional timber harvest. However, the significance of the quadratic relationship between timber harvest and adult female woodland caribou survival must be viewed with caution. There were a limited number of adult females exposed to cutblock densities beyond the inflection point of the quadratic relationship (> 3.4 ha/km^2). [Only 4.3% (6/141) of the radiocollared females were exposed to this level of timber harvest, or greater; however, 2 of the 6 females died]. This resulted in large confidence limits around the mean (Fig. 3.7). In addition, the confidence intervals for Timber Harvest^2 overlapped 0 indicating poor power and weak statistical inference (Table 3.3). More data from animals exposed to the higher range of cumulative timber harvest is required before more definitive conclusions can be drawn. Moreover, all cutblocks require roads and any positive influence of low levels of timber harvest on adult female woodland caribou survival would probably be eclipsed by the negative affect of increased road densities.

The designation of each herd as different strata results in the calculation of herd-specific baseline hazard functions, but common β coefficients (Therneau and Grambsch 2000). The strength of this approach is that the findings are more robust as a function of these 3 replicates (i.e. the designation of herds as strata are somewhat analogous to "blocking" in ANOVA's; Therneau and Grambsch 2000). Consequently, there is strong evidence to support the conclusion that the density of roads within caribou home ranges is negatively correlated with adult female woodland caribou survival. These correlations occur despite different levels of industrial activity on each caribou range.

Previous work in the RPC woodland caribou range (migratory, mountain ecotype) found that caribou appeared to avoid roads (and riparian areas), but not seismic lines (Oberg et al. 2003). These results were in contrast to that for a boreal herd in Alberta where seismic lines were avoided by 250 m (Dyer et al. 2001). However, a major difference between the 2 study areas may have been the age of the lines (and the corresponding vegetation response). The linear footprint on the RPC range was analyzed in 1998 when > 80% of the seismic lines were > 23 years old (see Fig. 3.3: RPC Herd). By this time, vegetation [e.g. green alder (*Alnus crispa)*] may have grown up thereby eliminating the potential for snowmobile activity on the lines (no packed access for wolf travel in the winter). Additionally, the RPC study area is in the Upper Foothills Ecoregion and the topographical relief is much greater than the Boreal Ecoregion where Dyer et al. (2001) conducted their work. This would also tend to reduce human use of these lines and line-of-sight.

Seismic lines are more prevalent on the landscape (compare Fig's. 3.2 and 3.3), but may not offer the travel efficiency for wolves that maintained roads do in winter. Linear disturbances have been linked to increased travel rates for wolves of up to 2.8 times during winter (James 1999). Presumably packed access (either through industrial clearing or recreational snowmobile use) would provide easier travel routes for wolves than adjacent forests. Seismic lines that are not packed/cleared actually might be harder for wolves to travel than adjacent forests in years of deep snow (seismic lines do not have snow-intercepting forest canopies and therefore would have greater snow depths). With the exception of years when seismic programs were being conducted in the 3 study areas, most seismic lines might not be cleared/packed, particularly older ones that had vegetation growth and more "deadfall" restricting human access. There was significant seismic activity in the LSM range during the winter of 1997-1998 (Fig. 3.3), but snowfall was approximately 34% below average that winter (94 cm total snowfall; November 1997-April 1998 vs Mean November-April 1980 – 2003 of 150 cm; Source = Grande Cache Airport). In general, seismic lines would not be expected to have as significant negative long-term (or continual) effect on adult female survival as roads.

The negative association of higher densities of timber harvest and adult female woodland caribou survival also might be attributed to grizzly bear (*Ursus* *arctos*) and black bear (*U. americanus*) behaviour. Both grizzly bears and black bears typically den from late November to mid-April/mid May in west-central Alberta (Stenhouse and Munro 2001). Consequently, bear predation is not a factor in the winter months. However, despite adult female woodland caribou mortality occurring predominantly between fall and early spring (September – April), the months of September and March/April account for over half of the observed deaths; periods when bears still could be active. Mortality factors could not be documented directly given the sampling intervals used in this study exceeded the period when scavenging vs predation could be ruled out (~ 2 days; Adams et al. 1995).

Bear predation is most likely a factor in May/June, when female caribou are calving and therefore more sedentary (see Boertje et al. 1988, Adams et al. 1995, Edmonds 1988, Young and McCabe 1998, Edmonds and Smith 1991). Additionally, calves cannot travel far for the first 2 weeks of their life making their dams more vulnerable. In west-central Alberta, grizzly bears were found in cutblocks more than expected between June 15 and August 7 (Nielsen et al. in press). Additionally, perennial streams (shorelines <20m in width and continual flow) were selected by grizzly bears between den emergence and July 31, while major streams (shoreline width >20 m) and intermittent streams (shorelines <20 m in width and often dry) were used in proportion to their availability (Nielsen et al. 2002). Ungulates (primarily moose calves) dominate the diet of grizzly bear in west-central Alberta throughout the month of June, but the amount of ungulate biomass in the diet begins to increase in late May and there is a small increase again in late October (Munro and Stenhouse 2003). Since cutblocks and perennial streams would provide good forage for alternate prey (moose, elk and deer), woodland caribou that spent the spring near these areas might have a higher probability of encountering bears (and suffering mortality and/or loosing calves). In the central Canadian arctic, barren-ground caribou are the major prey of grizzly bears in the spring (May – mid June) and fall (September – mid October) (Gau et al. 2002). An increase in bear predation on ungulates can occur during the rut (Boertje et al. 1988, Hamer and Herrero 1991). In Yellowstone National Park and vicinity, one study found that predation (as opposed to scavenging) on ungulates by grizzly bears was highest in October but relatively low in April/May (Mattson 1997). However, in the same area, all predation by grizzlies and black bears on elk calves occurred within 28 days of birth (late May – mid June) (Singer et al. 1997).

In summary, the examination of industrial disturbance variables relative to adult female woodland caribou survival (through the A-G model) indicates a negative correlation with the cumulative density of roads. Additionally, there appears to be an initial positive correlation between timber harvest and survival at low densities of cutblocks followed by a negative relationship as timber-harvesting activity accumulates. The proximal cause of the increase in caribou mortality is attributed to wolf and to a lesser extent bear predation. The mechanism for the increase in the probability of predation is linked to; 1) corresponding increases in alternate prey responding to forage along roads and in cutblocks, 2) a reduction in home range size and therefore the ability of woodland caribou to avoid predators and 3) the direct removal of lichen forage through industrial disturbance. Strategies to conserve populations of woodland caribou relative to timber harvest include: 1) the minimization of early seral stage forests adjacent to caribou movement routes (Johnson et al. 2002), 2) localizing timber harvest in terms of time and space (Hervieux et al. 1996, Seip 1998, Smith et al. 2000), and 3) the avoidance of core areas of caribou activity (Darby and Duquette 1986, Cumming 1992, Cichowski and Banner 1993, Smith et al. 2000). Further implications of the correlation between anthropogenic change and adult female woodland caribou survival are discussed relative to population persistence in Chapter 4.

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CHAPTER 4. LANDSCAPE EFFECTS ON WOODLAND CARIBOU POPULATION PERSISTENCE: FUTURE OUTLOOK.

INTRODUCTION

Woodland caribou persist at low densities (Thomas 1995), have low fecundity and late maturation, and are associated with large tracks of older forests. These factors reduce their ability to adapt quickly to large-scale landscape changes brought about by industrial activities. Consequently, they might be expected to be one of the first species to decline when faced with alterations to forests caused by timber harvest and oil and gas exploration and development in west-central Alberta. However woodland caribou can live 12 - 15 years and this long life can make short-term consequences more difficult to track. Moreover, time lags in the demographic response of species faced with habitat loss have been documented (Hanski and Ovaskainen 2002, Carroll et al. 2004). The term *extinction debt* has been coined to describe situations when the habitat loss threshold for a population's persistence has been exceeded, but the species still exists because of this time lag (Tilman et al. 1994).

In west-central Alberta, 2 woodland caribou herds (A La Peche and Redrock/Prairie Creek) have traditionally made seasonal migrations of over 80 km between alpine summer range and forested winter range (mountain ecotype). A 3rd herd (Little Smoky) is considered a boreal ecotype and it remains in the upper foothills and sub-alpine throughout the year (Edmonds 1988, Edmonds and Smith 1991). The advantages of a migratory strategy include the avoidance of predators (Bergerud and Page 1987, Thomas 1995, Fryxell et al. 1988) and insect harassment, and increased access to seasonally available forage. Migratory herds are generally larger and more successful than sedentary ones (Thomas 1995) and loss of this migratory behaviour may be one of the early warning symptoms of impending population decline. In the absence of other limiting factors, movement of caribou to areas where food is inferior indicates avoidance of other negative factors (Thomas 1995). The mountain caribou ecotype in west-central Alberta migrates to calving areas where food quantity and quality is poor relative to that at lower elevations, but where predator pressure is less (Edmonds and Smith 1991).

Fragmentation of landscapes through industrial activity may have detrimental effects on woodland caribou by reducing the amount of lichens available for forage and increasing the probability of encountering predators (see Chapter 3). Additionally, avoidance of linear features (Oberg et al. 2003, Dyer et al. 2002) and timber harvesting activities (Smith et al. 2000) can concentrate woodland caribou, thereby possibly increasing detection rates by predators. Clear-cut logging removes older forests and subsequent regeneration provides better forage for alternate prey (deer, elk, and moose). In turn, this may result in greater wolf numbers on woodland caribou range (see Chapter 3). Predicting the outcome of landscape change relative to population persistence of woodland caribou populations is one of the greatest wildlife conservation challenges in west-central Alberta.

In this chapter, I apply the results of the effects of industrial activity on adult female woodland caribou survival as predicted by the Andersen-Gill analysis from Chapter 3 to the population projection models for each woodland caribou herd from Chapter 2. Specifically, the risk ratios associated with increasing road development and timber harvest (A-G model results) are used to project adult female woodland caribou survival. These survival estimates are entered into the population projection model along with estimated calf recruitment. Additionally, I examine changes in woodland caribou distribution between 1981 and 2001 and discuss factors associated with these changes. I discuss the implications of landscape change caused by industrial activity to adult female woodland caribou survival and population persistence in west-central Alberta. Finally, I review current conservation policies related to woodland caribou, make management recommendations and discuss future research needs.

METHODS

Landscape changes and adult female survival

As outlined in Chapter 2, range-specific adult female survival was estimated from radio-collared samples of adult female woodland caribou using Pollock et al.'s (1989) staggered-entry modification of Kaplan and Meier's (1958) survivorship model. Sampling variance associated with annual survival was removed to properly estimate environmental variance of the series of years (White 2000). Calf-cow ratios were estimated from composite aerial surveys as per Eqn. 2 (see Chapter 2). The mean ± 1 standard deviation of these 2 parameters was randomly entered in a stage-structured stochastic model based on Eqn. 5 (see Chapter 2).

The results of the Andersen-Gill analysis (see Chapter 3) indicate a correlation between adult female survival and the density of roads and timber harvest within the MCP of each animal. To examine the potential implications of additional landscape change on population response of woodland caribou, the *beta* coefficients of these relationships were used to modify adult female survival within the population model discussed in Chapter 2. For example, the beta coefficient for roads was 0.004. This correlation indicates that a 100 m increase in the density of roads/km^2 would result in a 1.5 (=(exp[100 × 0.004])) times greater risk of dying compared to an adult female that was not confronted by this increase in road density within her home range. If annual female mortality was 0.10, the influence of an increase in the density of roads of 100 m/ km^2 would be 0.15 (1.5*0.10) or a corresponding survival rate of 0.85. A series of adult female survival rates modified by these *beta* coefficients were entered into the population model and the results compared to the 2001 baseline for 20 year projections using the following equation:

(Eqn. 9)

$$\begin{bmatrix} n_c \\ n_a \end{bmatrix}_{t+1} = \begin{bmatrix} 0 & CCR \\ s_c & s_a \end{bmatrix} \begin{bmatrix} n_c \\ n_a \end{bmatrix}_t$$

where , n_c is the number of calves entering the population, n_a is the number of adult females in the population at time t, S_c is the calf survival rate, S_a is the adult female survival rate modified by increasing amounts of industrial activity and CCR is half the calf-cow ratio (assuming a 50:50 sex ratio). The matrix is time varying and the elements of CCR and S_c and S_a are randomly chosen from ± 1 standard deviation around the mean. A log-odds transformation was applied in order to restrict random survival variates to ≤ 1.0 (Burnham et al. 1987). Each population projection was initiated with a population of 100 adult females (to facilitate between herd comparisons), replicated 1,000 times for a 20-year period, and the mean of these projections summarized annually.

Tracking changes in distribution and migration routes

Changes in distribution were examined by plotting all locations of VHF radiocollared animals in each of the 3 herds annually (if data were available, less frequently, if not) during the period 1981 – 2002. The RPC herds' migratory patterns were examined through successive locations of GPS-radiocollared animals during the spring and fall migratory periods of 1998 – 2002. The proportion of radiocollared woodland caribou bypassing Caw Ridge was compared to previous data collected during the mining activity encroachment onto Caw Ridge (Jakimchuk et al. 1999). The development of the open-pit coal mine was illustrated relative to the RPC herds migratory path.

Demographic response to changes in distribution and/or migratory behavior

Herd-specific adult female survival and calf-cow ratios (recruitment) were examined annually relative to changes in woodland caribou distribution and/or migratory behaviour between 1998 and 2003.

Snow

Total cumulative snowfall was summarized for the period of November 1 – April 30 each year (1980 – 2003) from the Grande Cache airport, which was the most central weather station to the study area during the study period.

RESULTS

The potential effects of increases in landscape changes on population trajectories of the 3 woodland caribou herds were compared to the 2001 landscape conditions. An increase in road density of 120 m/km^2 is projected to cause a negative population response in the ALP herd (Fig. 4.1). Both the RPC and the LSM herds are projected to decline at current levels of road density and any increase will only exacerbate the projected population declines (Fig. 4.2 – 4.3).

An increase in cutblock density to 7.4 ha/km^2 is projected to cause a decline in the ALP herd (Fig. 4.4.), while timber harvest above a threshold of 6.8 ha/km^2 is sufficient to cause a negative population response in the RPC herd (Fig. 4.5). Similar to the effect of roads, any further change in the cutblock density within the LSM range simply hastens the decline of that herd (Fig. 4.6)

The LSM herd continued to use the same basic annual range throughout the 20-year study (Fig. 4.7 – 4.10). However, a major change in woodland caribou distribution was observed in the ALP herd. The traditional movement onto forested winter range has been abandoned by the majority of this herd since 1996, with the exception of an estimated 30 animals (Fig. 4.11– 4.14). Likewise, the RPC herd appeared to abandon the Prairie Creek portion of its winter range during the same time period (Fig. 4.15 – 4.19). In addition, the percent of radiocollared animals from the herd that bypassed the main migration route over Caw Ridge steadily increased concurrent with the advancement of mining activity towards Caw Pass (Fig. 4.20).

Following these changes in distribution, adult female survival in the ALP herd was relatively constant, but calf recruitment declined (Fig. 4.21). In the RPC herd, adult female survival and calf recruitment dropped in 2000-2001 (Fig. 4.22), while the LSM herd showed poor adult female survival and calf recruitment between 1999 and 2003 (Fig. 4.23).

Annual November-April snowfall was variable between years and averaged 150 cm for the period 1980-2003 (Fig. 4.24) and there were no apparent patterns that appeared to be correlated with the observed changes in woodland caribou distribution.

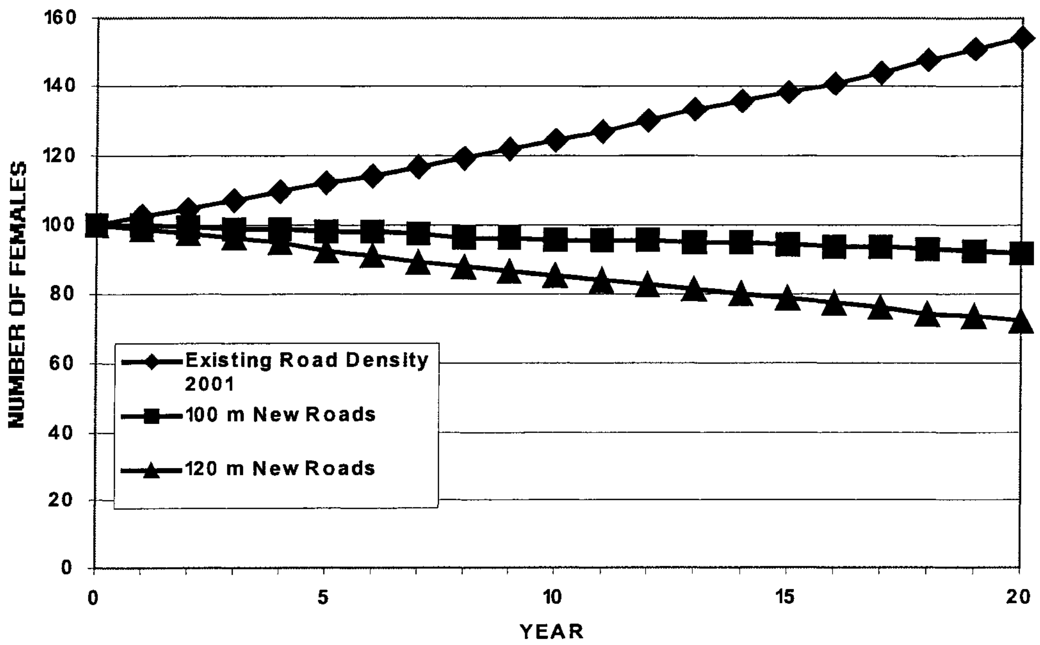

Fig. 4.1. Expected population change for the ALP herd based on different densities of new roads within a woodland caribou MCP.

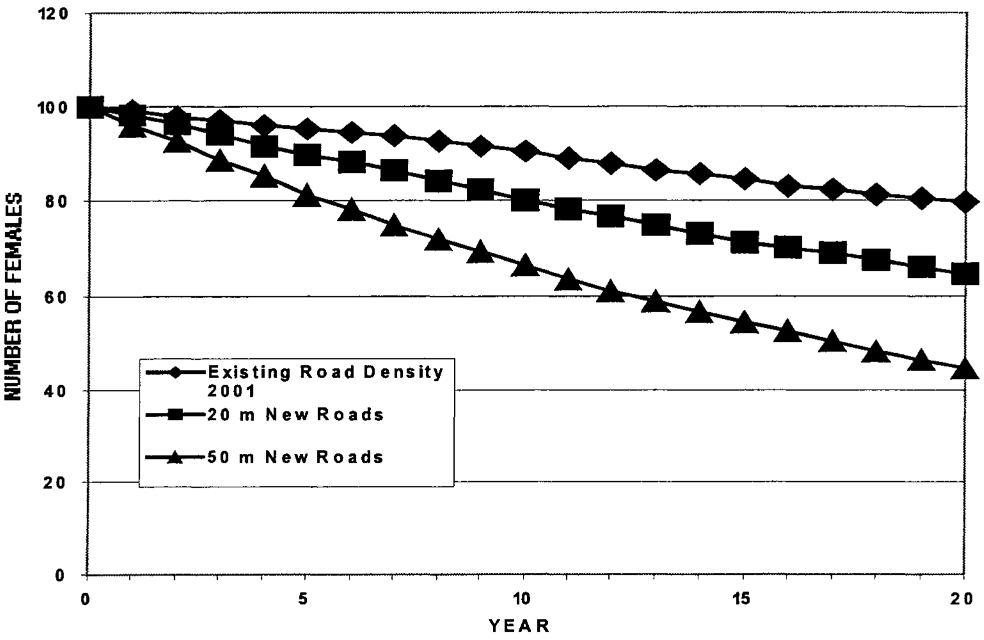

Fig. 4.2. Expected population change for the RPC herd based on different densities of new roads within a woodland caribou MCP.

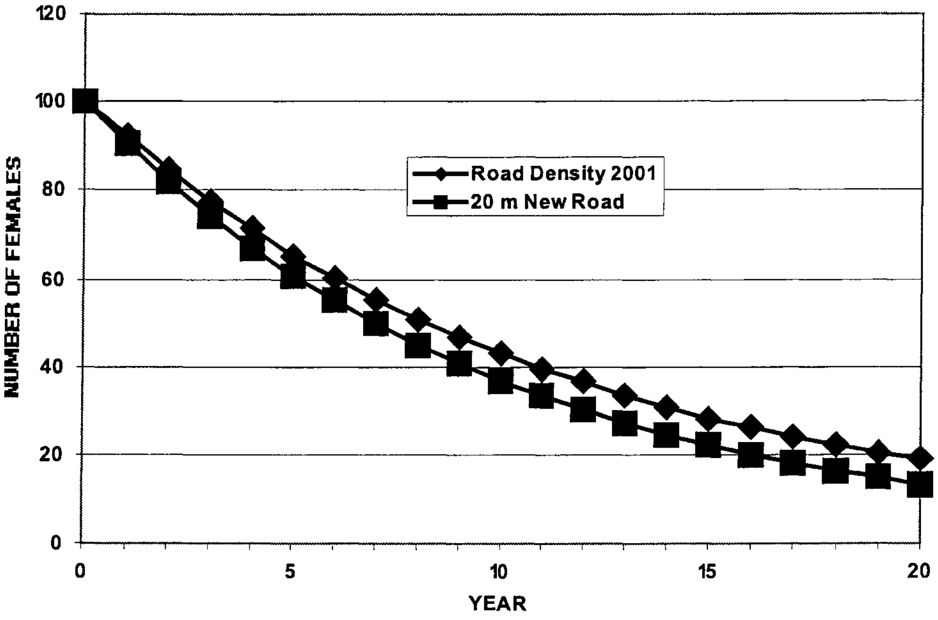

Fig. 4.3. Expected population change for the LSM herd based on different densities of new roads within a woodland caribou MCP.

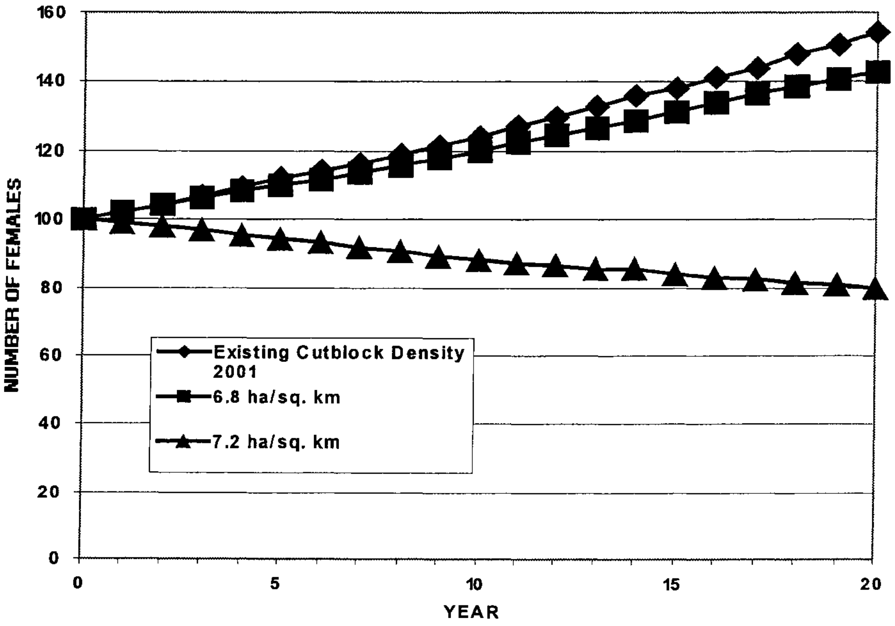

Fig. 4.4 Expected population change for the ALP herd based on different cutblock densities within a woodland caribou MCP.

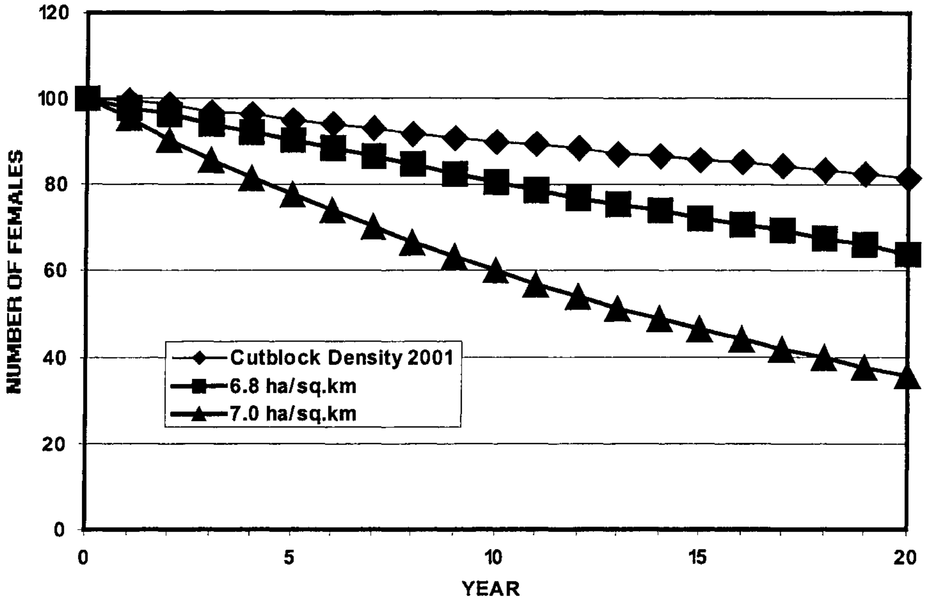

Fig. 4.5. Expected population change for the RPC herd based on different densities of cutblocks within a woodland caribou MCP.

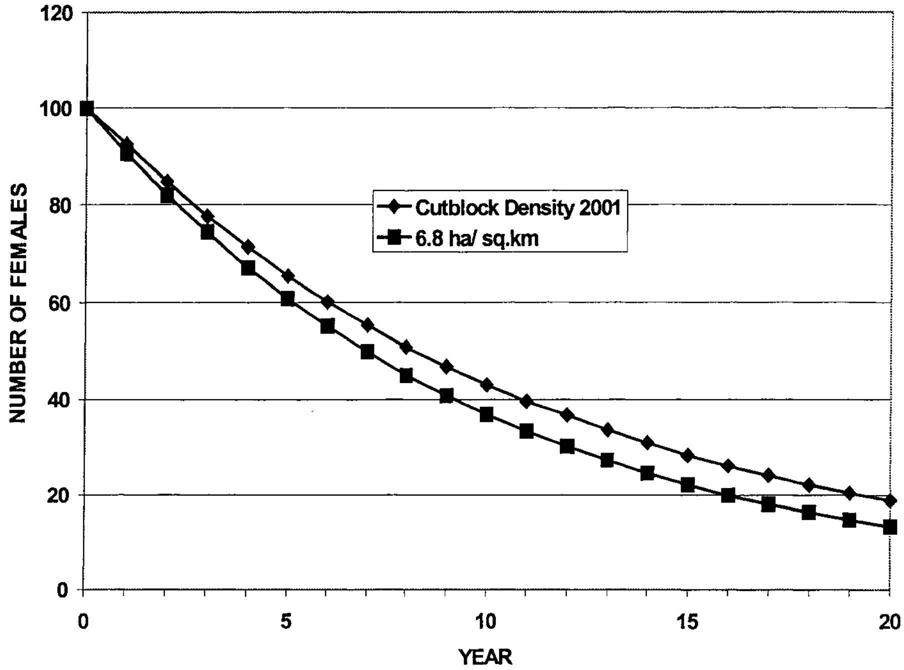

Fig. 4.6. Expected population change for the LSM herd based on different densities of cutblocks within a woodland caribou MCP.

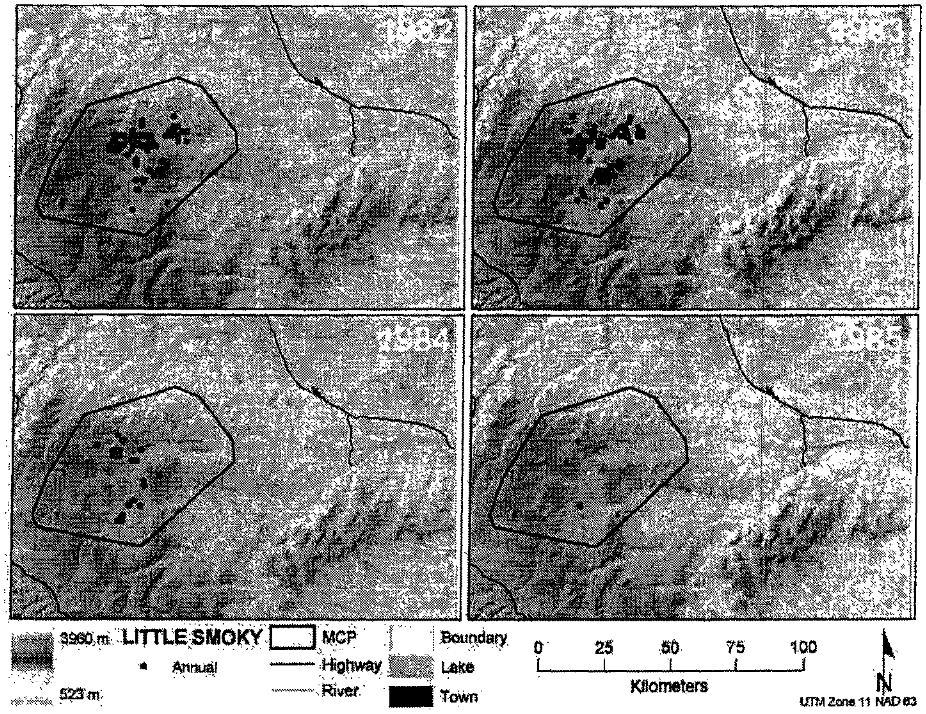

Fig. 4.7. Distribution of the LSM woodland caribou herd based on radiotelemetry locations, 1982 – 1985.

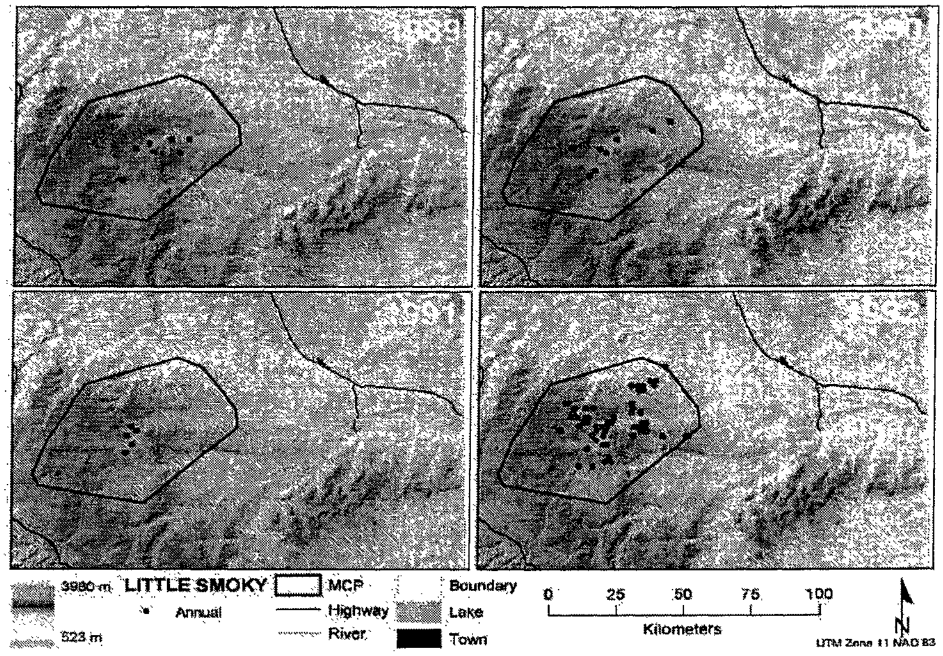

Fig. 4.8. Distribution of the LSM woodland caribou herd based on radiotelemetry locations, 1989 - 1992.

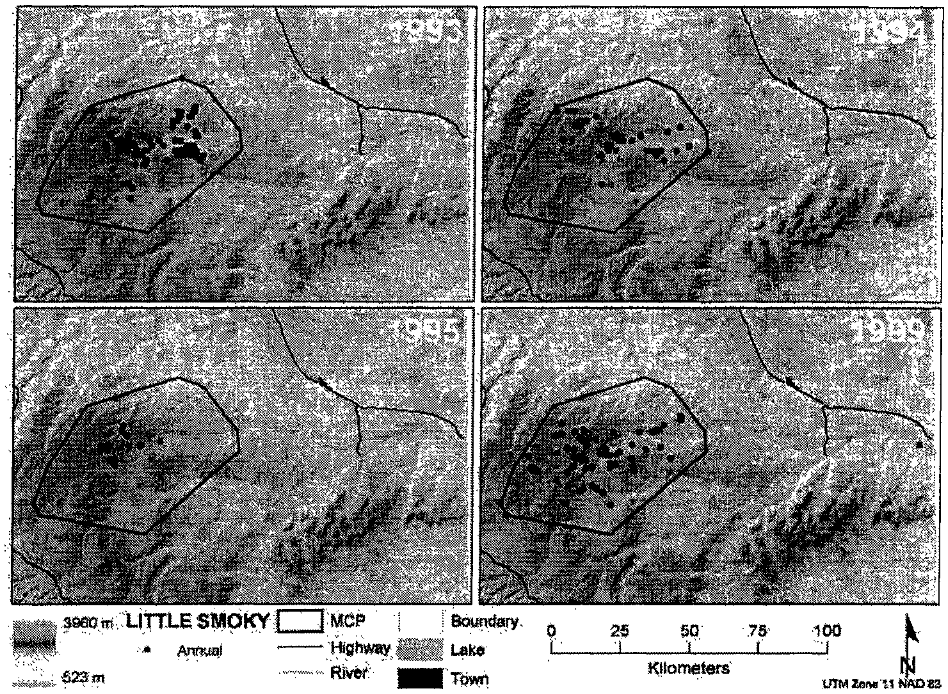

Fig. 4.9. Distribution of the LSM woodland caribou herd based on radiotelemetry locations, 1993 – 1999.

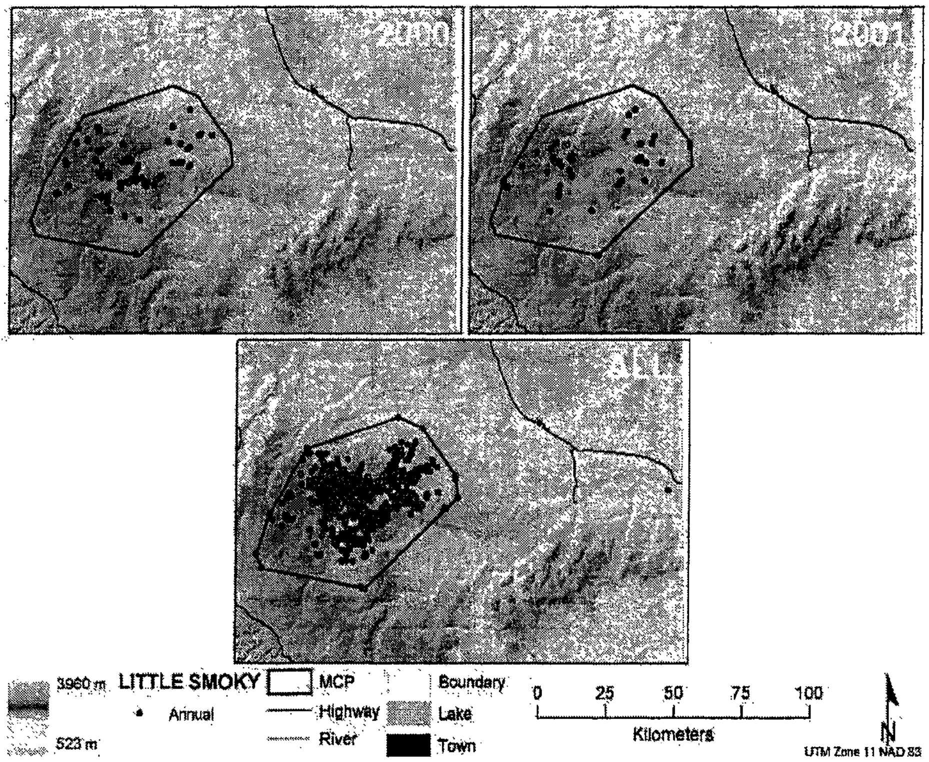

Fig. 4.10. Distribution of the LSM woodland caribou herd based on radiotelemetry locations, 2000 – 2001, plus all locations depicted, 1981 – 2001.

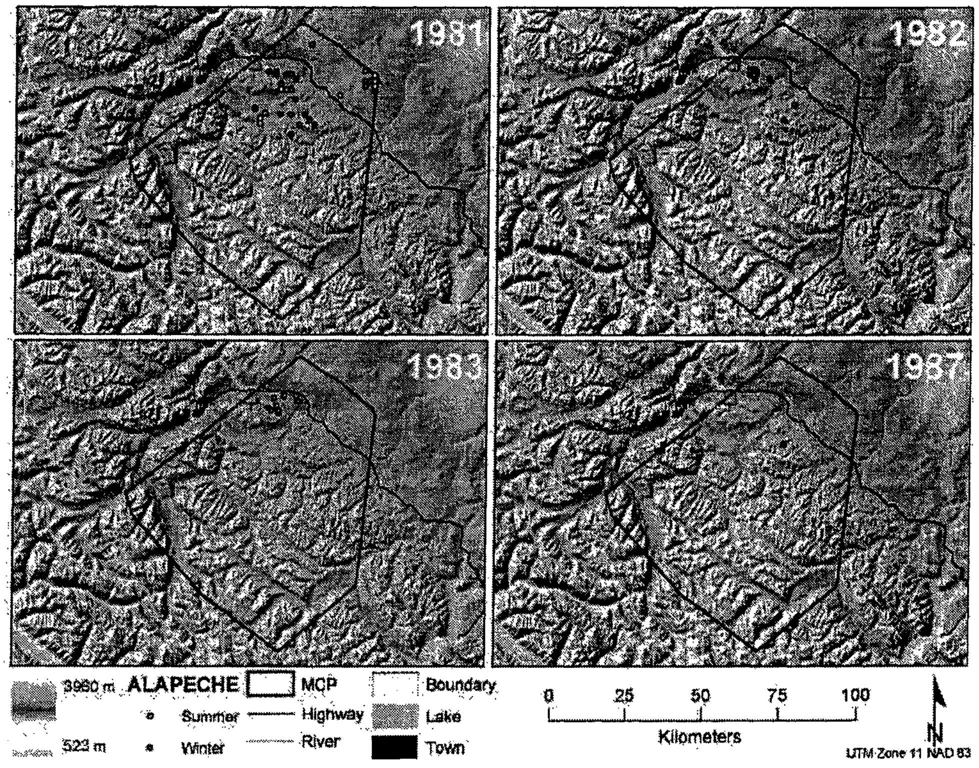

Fig. 4.11. Distribution of the ALP woodland caribou herd based on radiotelemetry locations, 1981 – 1987.

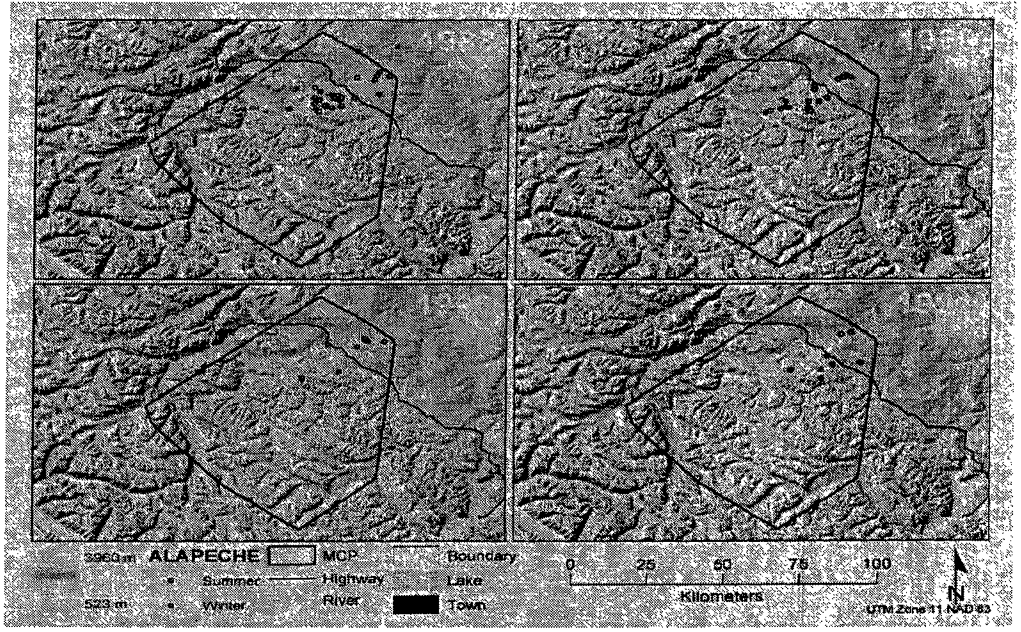

Fig. 4.12. Distribution of the ALP woodland caribou herd based on radiotelemetry locations, 1988 – 1991.

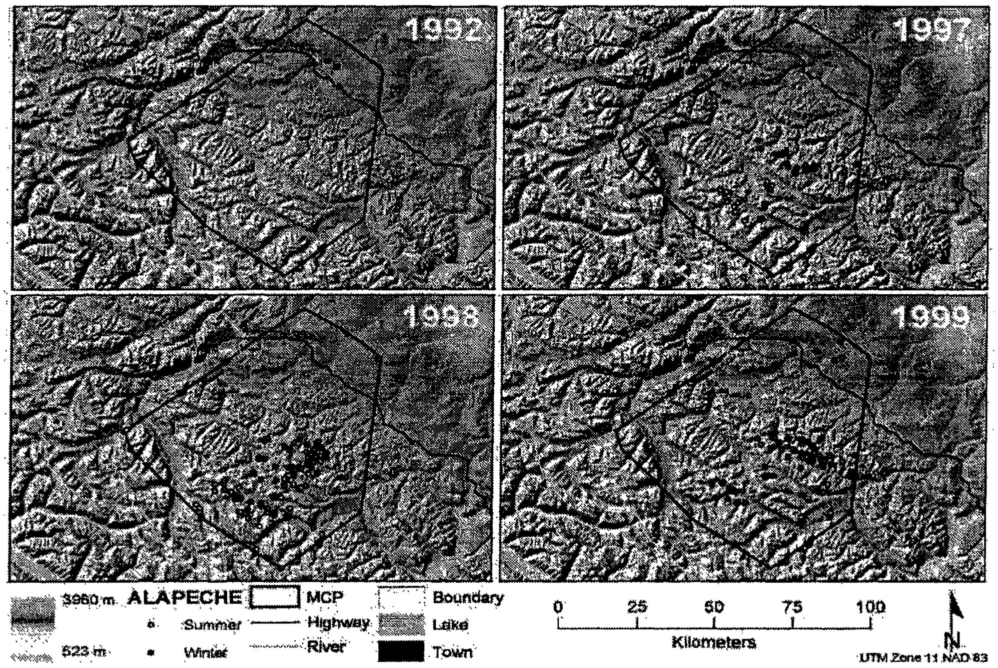

Fig. 4.13. Distribution of the ALP woodland caribou herd based on radiotelemetry locations, 1992 – 1999.

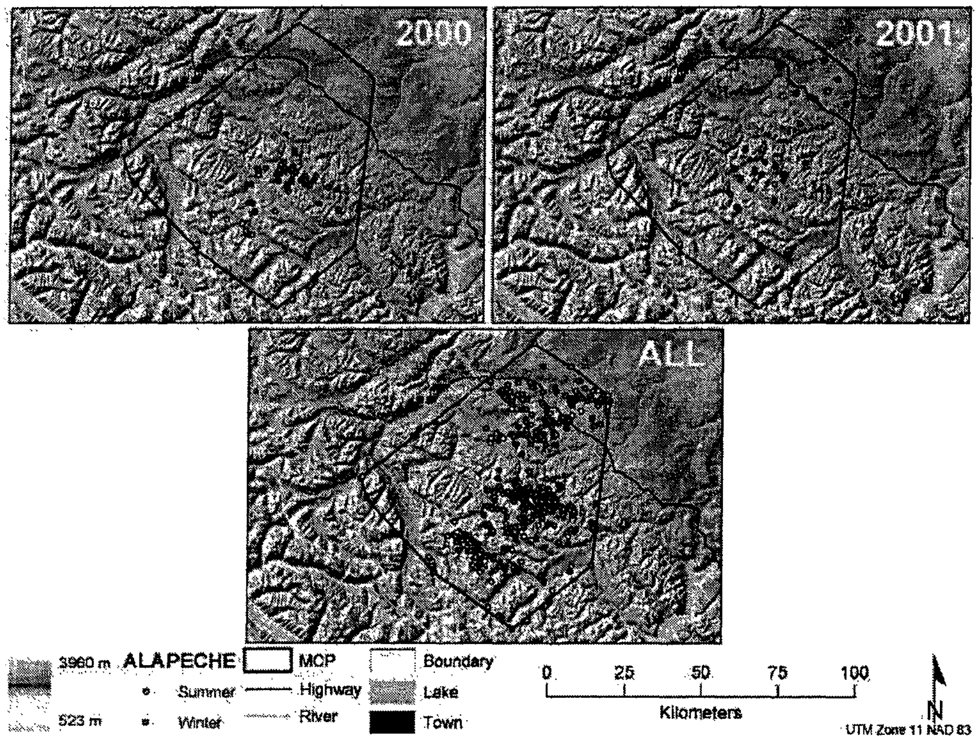

Fig. 4.14. Distribution of the ALP woodland caribou herd based on radiotelemetry locations, 2000 – 2001, plus all locations depicted 1981 – 2001.

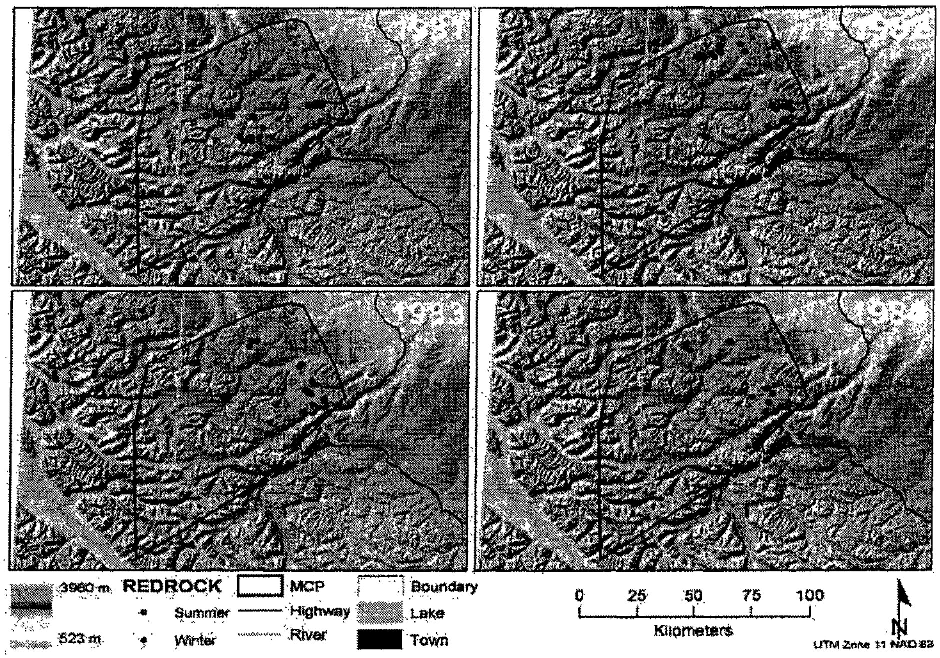

Fig. 4.15. Distribution of the RPC woodland caribou herd based on radiotelemetry locations, 1981 – 1984.

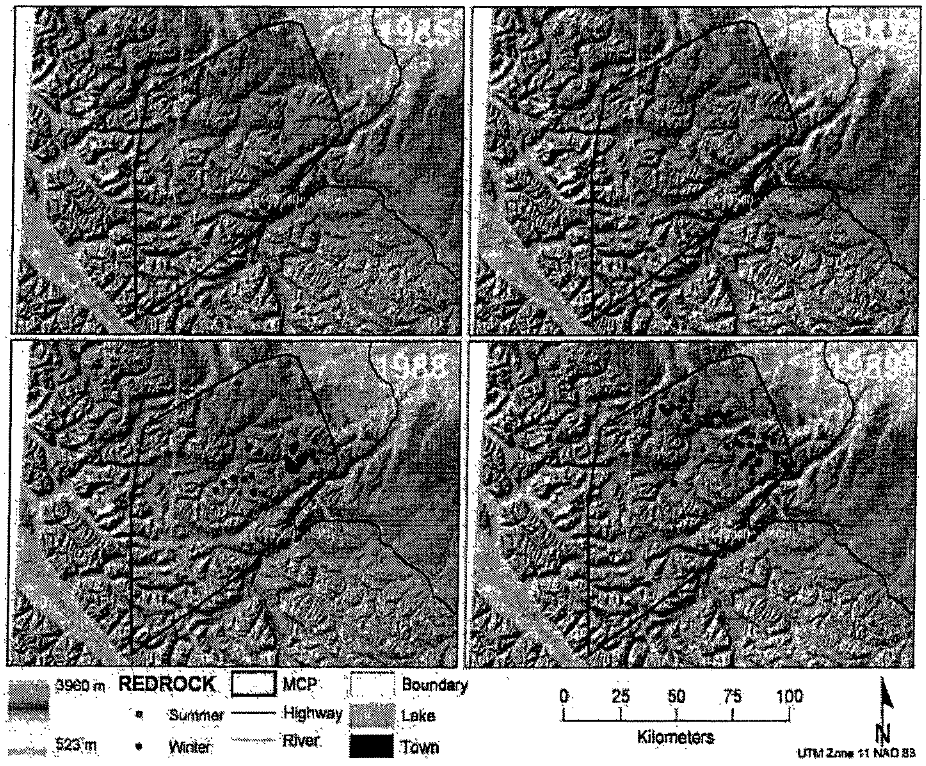

Fig. 4.16. Distribution of the RPC woodland caribou herd based on radiotelemetry locations, 1985 – 1989.

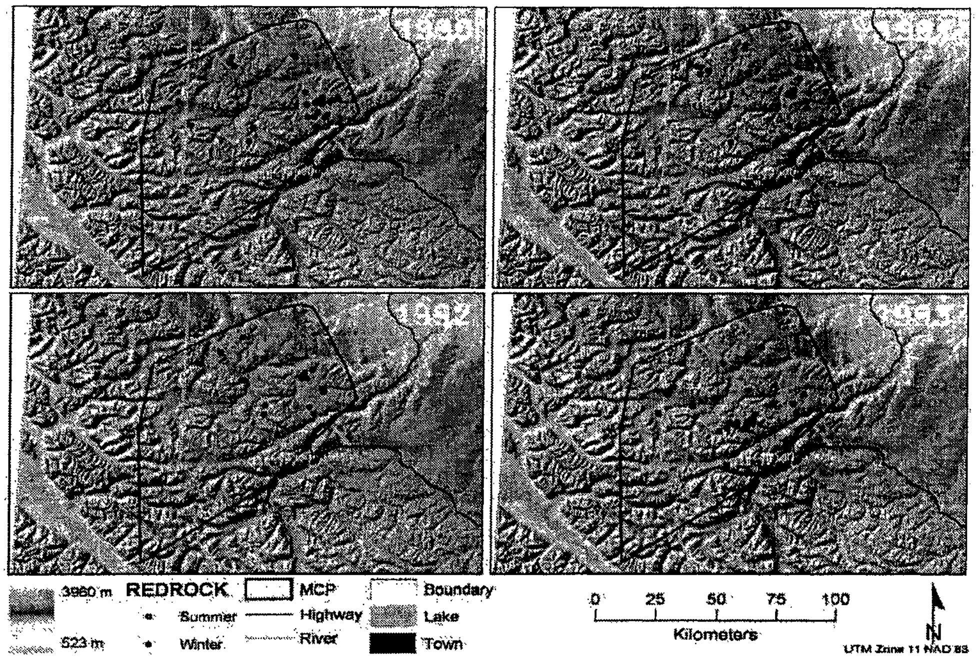

Fig. 4.17. Distribution of the RPC woodland caribou herd based on radiotelemetry locations, 1990 – 1993.

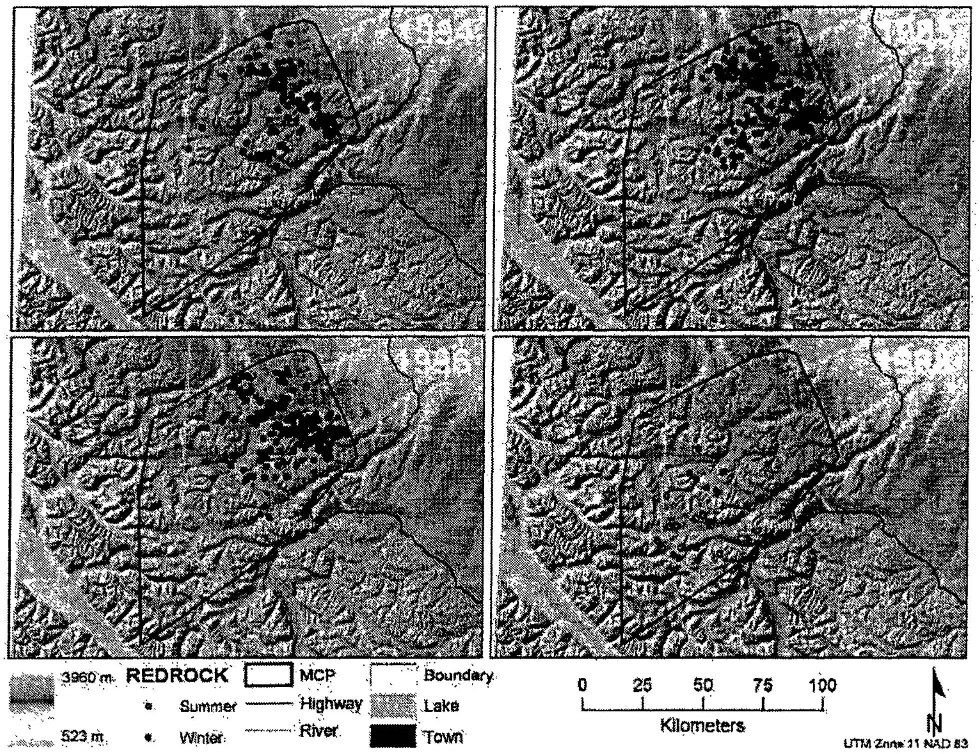

Fig. 4.18. Distribution of the RPC woodland caribou herd based on radiotelemetry locations, 1994 – 1998.

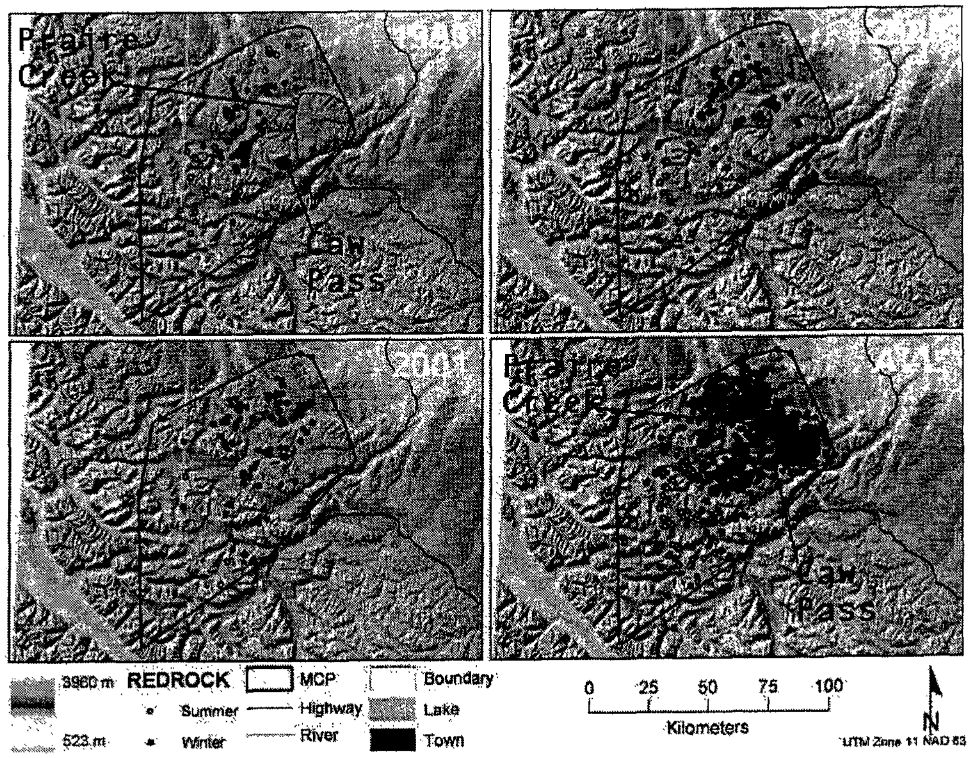

Fig. 4.19. Distribution of the RPC woodland caribou herd based on radiotelemetry locations, 1999 – 2001, plus all locations depicted, 1981 – 2001.

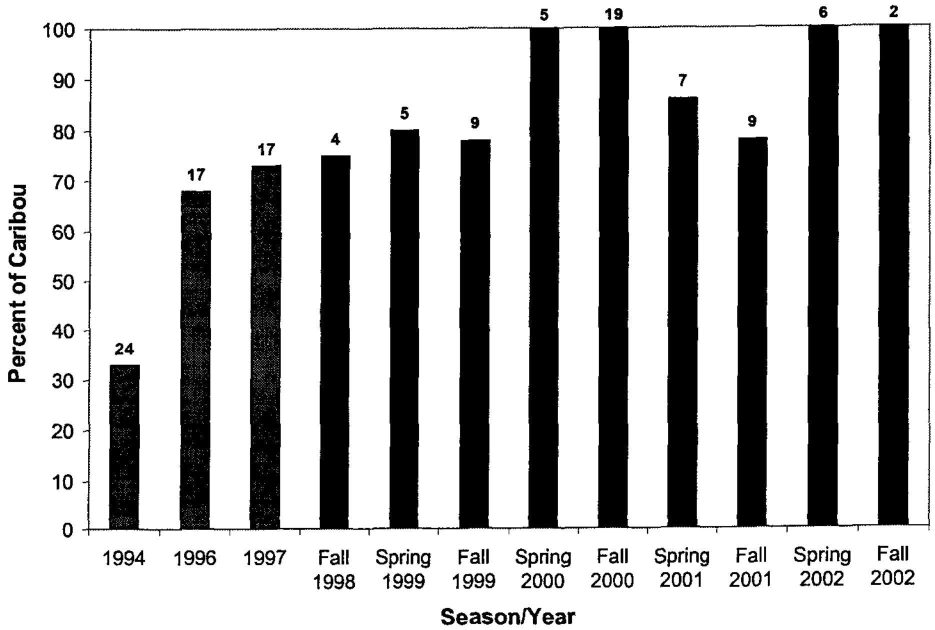

Fig. 4.20. Percent of radiocollared woodland caribou bypassing Caw Ridge during migration, 1994 - 2002. Numbers above bars represent sample size of collared animals. Grey bars for 1994 – 1997 from Jakimchuk et al. (1999).

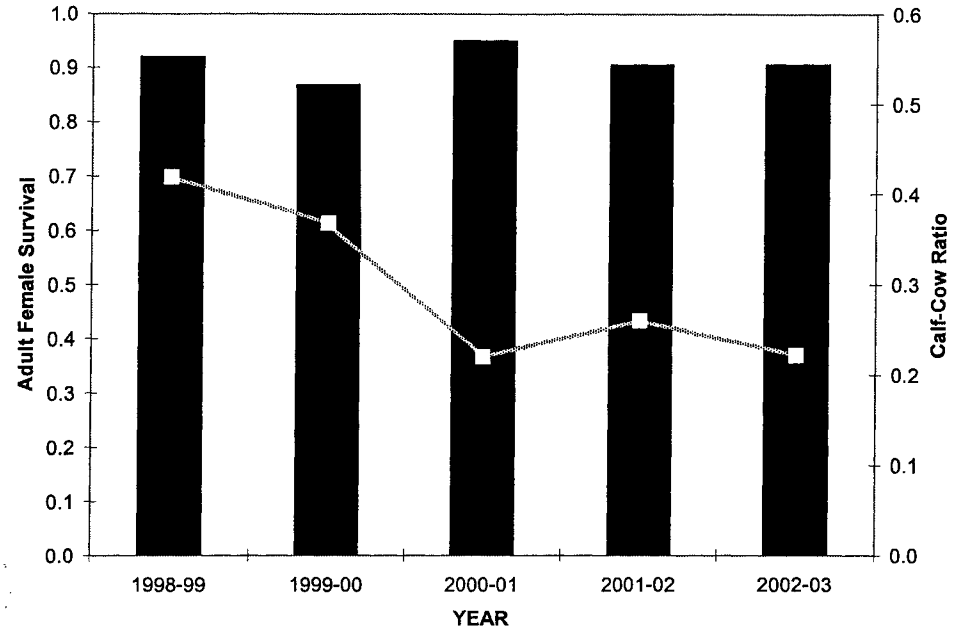

Fig. 4.21. Annual adult female woodland caribou survival and calf recruitment in the ALP herd of west-central Alberta, 1998 – 2003. Solid bars and hatched lines represent adult female survival and calf-cow ratios, respectively.

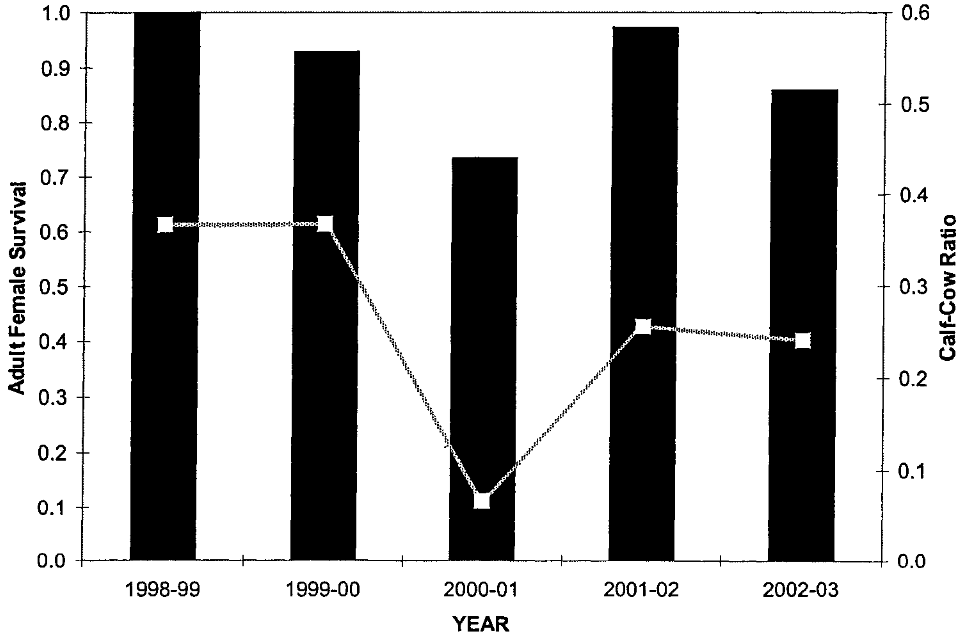

Fig. 4.22. Annual adult female woodland caribou survival and calf recruitment in the RPC herd of west-central Alberta, 1998 – 2003. Solid bars and hatched lines represent adult female survival and calf-cow ratios, respectively.

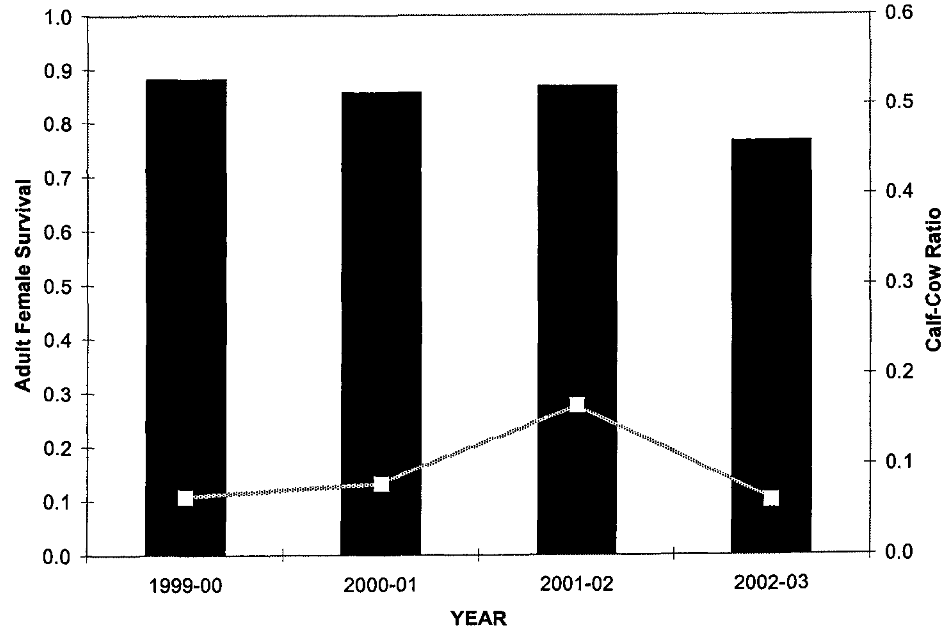

Fig. 4.23. Annual adult female woodland caribou survival and calf recruitment in the LSM herd of west-central Alberta, 1999 – 2003. Solid bars and hatched lines represent adult female survival and calf-cow ratios, respectively.

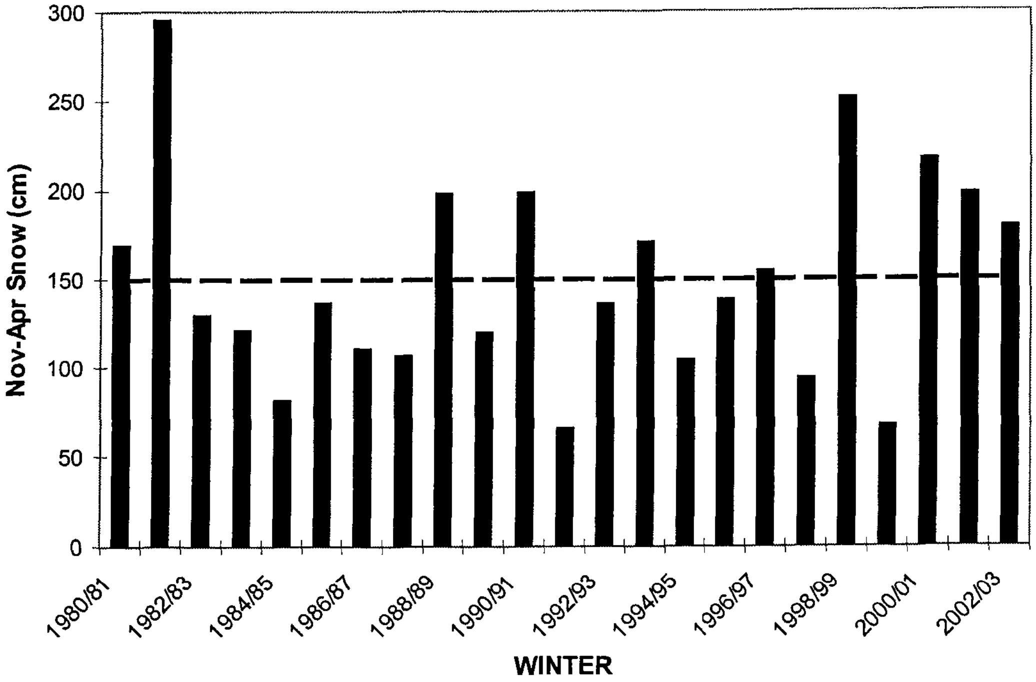

Fig. 4.24. November-April total cumulative snowfall at Grande Cache, AB, 1980 – 2003. The dashed line represents the mean annual snow accumulation for that time period.

Discussion

The potential effects of increases in the density of roads and timber harvest on adult female survival of the 3 woodland caribou herds illustrate how sensitive population response can be to industrial development. In southern British Columbia, a sub-population of the *mountain-boreal* ecotype (lives in the mountains year round) of woodland caribou was projected to face extirpation within a decade. The portion of this populations range that exhibited the highest mortality of woodland caribou contained road densities of 1.0 km/km^2 and a higher proportion of disturbed habitats (23%) (Kinley and Apps 2001). Simulation experiments based on data from northern Alberta suggested that dramatic declines of woodland caribou would occur when the density of linear features exceeded 1.22 km/km^2 (Weclaw and Hudson 2004). These authors argue that food is not limiting to woodland caribou over most of their range and that caribou and wolves could coexist at low densities in much of the boreal forest if the density of seismic lines did not exceed 0.3 km/km^2 and there were no roads. In my study, the LSM herd is projected to continue its decline even without added industrial activity. Likewise the RPC herd's decline is projected to continue with as little as a 20% increase in road density (20 m/km^2 of new road added to the existing mean density in the overall range of 80 m/km^2). The ALP herd would appear to be able to withstand an increase in the density of new roads of 100 m/km^2 before suffering a decline (providing there wasn't any additional timber harvest). However, it should be noted that the majority of the ALP herd has not returned to the industrialized portion of their range for the last 7 years (and therefore most animals in the ALP radiocollared sample have not been exposed to industrial activity). Additionally, timber harvest would have an additive impact on population projections. Consequently, this projection (based on only roads) is considered overly optimistic.

Industrial linear features have been associated with increases in wolf traveling efficiency and the increased probability of caribou mortality in woodland caribou ranges (James and Stuart-Smith 2000). Avoidance of these features by woodland caribou has been documented (Oberg et al. 2003; Dyer et al. 2001) and the activity associated with these linear features has been implicated in increased movement rates (Bradshaw et al.1997), increased energy expenditures due to disturbance (Bradshaw et al. 1998) and barriers to with-in range movements (Dyer et al. 2002). Avoidance of similar activities by barren ground caribou has also been documented (Dau and Cameron 1986, Cameron et al. 1992, Cameron et al. 1995, Nelleman and Cameron 1998). However, despite avoidance of these features, the Central Arctic herd increased during this same time period (Cronin et al. 1998). My study documented population demographics consistent with a significant decline in the herd that was exposed to the greatest amount of linear disturbance.

The A-G analysis suggests that the effects of timber harvest results in an increased risk to adult female woodland caribou once 6.8% of the animals MCP is logged. I have not examined the spatial relationship of timber harvest within MCP's in this study (i.e. do caribou confronted with logging distributed over their entire range face a greater risk than those that experience the same amount of logging, but the timber harvest is localized?). However, given the avoidance of cutblocks documented in the RPC herd (Smith et al. 2000) it seems possible that the ability of adult females to avoid predation continues to decline as timber harvest progresses beyond this point. A reduction in individual winter range size was documented concurrent with increased cumulative timber harvest (Smith et al. 2000). If predators learned to focus on this reduced area it could increase the probability of woodland caribou being detected. The risk ratio appears to increase exponentially as more timber harvesting activity occurs. The winter ranges of the migratory herds are currently less impacted by timber harvest than the LSM range, but a negative impact is projected for both the RPC and ALP herds as timber harvest progresses. However, as discussed in Chapter 3, these results should be viewed with caution due to the low sample size of animals exposed to the upper range of cutblock densities.

During the course of this study, two changes in woodland caribou distribution have been detected. The most significant change has been observed in the ALP herd where it appears that the traditional movement onto forested winter range has been abandoned since 1996 (Fig. 4.15 – 4.16). In contrast to the previous 15 years of monitoring, the majority of animals in this herd now move from their summer range in the higher mountain ranges in north Jasper National Park (JNP) and south Willmore Wilderness Park (WWP) to the central and front ranges of WWP in the winter. [Recent data indicates that a small group of woodland caribou (~ 30) still use the forested winter range adjacent to Highway 40]. Only 2 of 6 GPS collared females moved into the mountains during the summer, while the other 4 remained in the forest year round (WCACSC unpublished data)].

There are a number of possible explanations that have lead to woodland caribou abandoning their historical winter range in the forested foothills. The herd may have been avoiding either the industrial activity or the alternate prey and accompanying predators associated with the fragmentation of their former winter range. This documented range shift coincided with new timber harvest on the winter range along Highway 40.

This range shift might be correlated with woodland caribou density. In the early 1990's, a significant number of animals (up to 10% in one year) were lost to vehicle collisions on the major highway (Highway 40) that bisects the forested winter range. A reduced number of animals may be able to obtain sufficient forage in the alpine wintering area until the population recovers to a larger number. A higher number of animals may need to winter in the forest where forage is more plentiful, but alternate prey that support predators are more numerous in this area (see Bergerud and Page 1987). In British Columbia, caribou inhabiting alpine areas were subjected to lower predation risk (Johnson 2000).

Snow conditions also could be a factor influencing the distribution of the ALP herd. However, snow accumulations appear to be as variable during the period that the ALP herd has remained in the mountains (1996-2003) compared to the period when they were wintering in the forest (1980-1995) (Fig. 4.24). Although mean November-April snow accumulations have been greater since the migration to the forested winter range was abandoned (162.7 cm, 1996 – 2003;

143.2 cm, 1980 – 1995) the difference is not significant ($t = 0.758$, $P > 0.457$). Snow depth does not appear to be a variable influencing this observed change in the winter distribution of the ALP herd. There was no relation between snow depth and woodland caribou herd demography in the Yukon (Hayes et al. 2003). Given the significance of snow to woodland caribou ecology, it will be important to monitor if global warming results in more variation of this parameter in the future.

A lesser change in distribution has been observed in the RPC herd. During the 1990's, the Smoky River Coal Mine expanded onto Caw Ridge. This ridge was one of the main migration routes between the summer and winter range for the RPC herd (Jakimchuk et al. 1999). Additionally, oil and gas development began to proliferate on the Prairie Creek portion of the RPC range. Concurrent with this mining activity, the radiocollared sample of the RPC herd increasingly avoided Caw Ridge and appeared to abandon the Prairie Creek portion of its winter range. Since the coalmine ceased operation in the spring of 2000, woodland caribou have not re-established their use of the Prairie Creek winter range (Fig. 4.20 – 4.21). Adult female survival was lower (0.873) during the period that the coal mining activity came in direct contact with Caw Pass (1998-2002) when compared to the earlier monitoring period (1993-1996; 0.947) when mining activity was further down in the valley and not directly in the migratory path (see Chapter 2; Table 2.1); however, due to limited sample sizes this difference was not significant ($F_{8,34} = 1.306$, $p > 0.25$).

A restriction in distribution and migratory behaviour is of concern relative to genetic diversity of woodland caribou. Despite over 700 individual caribou radio-telemetry tracking years in west-central Alberta, there has only been one instance where an animal from one range moved to another range. Moreover, this happened after the rut and the animal returned to its former range the following summer. Consequently, genetic interchange between herds may become less probable if additional industrial activity limits movements.

The combination of low adult female survival and poor recruitment suggests current and projected declines in the LSM herd. The ALP herd is currently increasing and is projected to increase in the near future. The RPC herd is currently showing a slight increase, but is projected to decline based on the stochastic model. Factors contributing to poor recruitment have not been documented directly. No calves were radiocollared, so the causes of calf mortality are unknown. However, the majority of calf mortality occurs during summer (Edmonds 1988, Edmonds and Smith 1991, this study). Predation, particularly by wolves, has been reported as a major mortality factor by most researchers (Bergerud et al. 1983, Bergerud and Elliott 1986, Bergerud and Page 1987, Van Ballenberghe 1985, Thomas 1995). In addition, grizzly bears can be major predators of woodland caribou where their ranges overlap (Seip 1992, Adams et al. 1995, Young and MacCabe 1998). Black bears also have been implicated in declines of woodland caribou (Rettie and Messier 1998). Density estimates of 4.6 grizzly bears/ 1000 km^2 and 5.7 black bears/ 1000 km^2 were documented in the LSM range and the eastern portion of the ALP winter range during 1981 – 1985

(Nagy et al. 1989). Although wolf densities on caribou winter range were estimated to be similar between the LSM herd range and 1 of the 2 mountain herds (RPC) (Kuzyk 2002), the 2 mountain herds leave the wintering area to summer in mountainous areas with lower wolf densities. In contrast, the LSM animals remain in the same area year-round and probably suffer greater predator-caused mortality of adult females. This is probably also the case for calves as well (see Adams et al. 1995). High wolf densities (11 wolves/1000 km^2) on the LSM annual range and the RPC winter ranges (Kuzyk 2002) were presumably maintained by higher densities of alternate prey (Edmonds and Smith 1991). Wolf reduction did not enhance adult woodland caribou survival in the Yukon (Hayes et al. 2003). (Because bears are not obligate carnivores, their densities may not have been affected by the presence of alternate prey).

In summary, woodland caribou in west-central Alberta have been exposed to the cumulative effects of oil and gas activity and timber harvest. One of the 3 herds exhibits demographic responses consistent with a significant decline projected to lead to eventual extirpation. This herd has the highest density of linear disturbance (roads, seismic lines) and the greatest proportion of range that has been logged. Additionally, the LSM herd does not migrate to protected areas during the summer. Migratory herds are generally larger and more successful than sedentary ones (Seip 1991, Thomas 1995) and the 3 herds in west-central Alberta appear to reflect this observation. Both the ALP and RPC herds are predicted to decline if levels of timber harvest and oil and gas development increase at current projected rates. Given the time lag between habitat loss thresholds and population extirpation (extinction debt) documented for other species (Hanski and Ovaskainen 2002, Carroll et al. 2004), the industrial activity thresholds calculated in this study may be overly optimistic.

The mechanisms that are, or have the potential to negatively affect all 3 herds include:

1. an avoidance of industrial activity (both oil and gas development and timber harvest),
2. an increase in predation rates (both in terms of more predators and increased hunting efficiency by predators). The probability of encountering predators could be linked to a greater density of alternate prey (elk, moose and deer).
3. an inability to "space away" from alternate prey and predators as woodland caribou restrict the size of their home ranges in response to habitat fragmentation.
4. a direct loss of habitat through landscape change. Habitat loss can include both a loss of lichen forage and a continual reduction in the amount of contiguous old forest that is favored by caribou and not selected by alternate prey.

The combination of all these factors can contribute to a decrease in adult female survival and calf recruitment.

Alberta Government Policy Direction and Commitments

The conservation of woodland caribou is supported by a number of Alberta Government Policy documents and directions. At the broadest level, the Government of Alberta supports Canada's commitment to the Canadian Biodiversity Strategy (Canadian Biodiversity Strategy Working Group 1995), which includes goals for the maintenance of biodiversity. At the provincial level, *Alberta's* commitment to the *Accord for the Protection of Species at Risk* and to the *National Framework for the Conservation of Species at Risk* reinforces the conservation of woodland caribou in the province. Woodland caribou are classified as threatened in Alberta and the Committee on the Status of Endangered Wildlife in Canada have listed the "southern mountain" (includes the ALP and RPC herds) and "boreal" populations of woodland caribou (includes the LSM herd) as *Threatened* nationally within Canada (COSEWIC 2002, 2003). Alberta's *Wildlife Act* and the federal *Species at Risk Act* provide requirements for a provincial recovery program. The overall goal of a recovery program is to restore species identified as *Threatened* or *Endangered* to viable, naturally self-sustaining populations within Alberta. The Fish and Wildlife Policy for Alberta (Alberta Public Lands and Wildlife 1982) also stipulates that viable wildlife populations are to be maintained. In addition, the Canadian Council of Forest Ministers first criterion for Indicators of Sustainable Forest Management is the *conservation of biological diversity* (Canadian Council of Forest Ministers 1997). A 1999 Alberta government policy (Information Letter 91-17) stipulates that "industrial activity can occur on caribou ranges as long as the integrity and supply of caribou habitat is maintained to permit its use by caribou". The 1999 Alberta Government document, Alberta's Commitment to Sustainable Resource and Environmental Management includes a commitment that species diversity shall be protected and maintained. Consequently, failure to maintain viable populations of woodland caribou in the Province of Alberta would violate a number of provincial and federal mandates and contradict the cornerstone of sustainable forest management.

Management Recommendations

1. Establish range-specific thresholds for industrial activity (number and location of roads, amount of timber harvest) in woodland caribou habitat as part of a recovery plan that is designed to ensure viable populations of woodland caribou in west-central Alberta.
2. Concentrate industrial activity in localized areas rather than over extensive areas. Coordinate the oil and gas industry and the timber industry so that these 2 industries are restricted to the same area at the same time. This could be achieved by selling oil and gas rights only in areas of timber-harvest activity.
3. Plan for effective amounts and configurations of woodland caribou habitat (large contiguous blocks of older coniferous forests) over meaningful planning horizons (e.g., 150 years).
4. Manage alternate prey populations (deer, elk and moose) through hunting to minimize predation affects on woodland caribou populations. Weclaw and Hudson (2004) argue that most caribou ranges in Alberta contain sufficiently high densities of linear features to preclude wolf reduction as a feasible strategy for recovering woodland caribou. They suggest reducing alternate prey populations (moose, elk and deer) as a viable alternative.
5. Transplants **should not** be considered as a means of mitigating woodland caribou populations suffering from habitat fragmentation. With the exception of

islands with little or no predation, most transplants of woodland caribou throughout North America have failed to re-establish populations that were extirpated or failing (Audet and Allen 1996, Compton et al. 1995). The exception warranting transplants would be if in-breeding depression was thought to be a factor limiting the recovery of a herd. In this instance, importing a few animals to enhance genetic diversity should be considered.

6. Coal mining should not be allowed on Caw Ridge or any point north of the existing Smoky River Coal mine footprint. This recommendation is designed to allow woodland caribou in the Redrock/Prairie Creek herd to re-establish their migratory route over Caw Ridge.
7. In order to reduce caribou/vehicle collisions, the speed limit along Highway 40 should be reduced to 90 kph at night between the Berland River and the settlement of Muskeg. Flashing signs should be established to warn motorists of caribou activity.
8. A Road Corridor Sanctuary (identical to the one on Highway 40) should be established along the Berland Resources Road in the Little Smoky caribou range.
9. In order to minimize the effects of fragmentation, the Berland Resources Road and the Smoky Mainline should not be joined in the future. [This avoids creating a direct route between the town of Grande Cache and communities to the east (Fox Creek, Whitecourt, Edson)].

Future Research

1. Data collected from GPS-collared woodland caribou should be analyzed in a similar fashion to increase the sample size of relocations relative to landscape changes.
2. Habitat variables should be included in the Andersen-Gill analysis as additional covariates [including Resource Selection Functions – (see Johnson et al. 2004)].
3. Wolf and grizzly bear distribution should be monitored relative habitat changes that alter predator/prey dynamics.
4. The location and timing of predation events should be monitored relative to habitat change.
5. Population responses of alternate prey (moose, elk, deer) should be closely monitored in relation to landscape change.
6. Genetic profiles of each herd should be analyzed to determine the likelihood of in-breeding depression becoming a conservation problem.
7. Controlled experiments should be conducted to establish cause-and-effect relationships between woodland caribou demographics, alternate prey response, predator response and industrial activity (see Boutin 1992).
8. Home range size should be examined over time (with the GPS collared sample available over the last 6 years) to determine if a reduction in woodland caribou range is occurring.

There are obviously a number of areas where more information is required to make sound management decisions. However, the implementation of many of the findings obtained to date is far more important than simply continuing to conduct research. Radiocollaring animals is intrusive, dangerous and sometimes fatal. If management agencies do not have any desire to implement research findings, it is of little consequence to continue research simply to make it appear to the public that adaptive land management is being practiced.

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