FINAL REPORT

Feasibility of Some Direct Management Options to Recover Populations of Boreal Caribou

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ABSTRACT

Over the last half century, populations of boreal caribou (Rangifer tarandus caribou) in north-eastern British Columbia have shown significant declines. In addition to habitat restoration efforts, direct management options (active predator control, maternity penning, predator exclosures and population augmentation) are being designed and tested to ensure these small populations can be stabilized and/or increased in the short-term while longer-term solutions (e.g., habitat restoration) are tested and implemented. To assess the feasibility of these options, we implemented a stochastic age-structured caribou population dynamics model assuming no density-dependence. Using demographic parameters estimated for this population, we explored potential scenarios for involving different levels of investment in these actions, applied singly or in combination. We found that for declining populations of caribou there is at least one and usually more than one feasible management action that if implemented over multiple years could result in improved probabilities of short-term population growth. Interactions between different management options and population demography can be complex. Achieving increases greater than 50% in the probability that annual population growth will increase above the “no management” scenario requires at least two and perhaps three concurrent management actions, and usually larger annual levels of investment. Management of wolves to population densities of <5/1000 km2 appears to significantly affect the apparent success of all other caribou population management options across all six boreal caribou populations. More realistic levels of management given costs might stabilize the populations but at lower levels. The implications of population recovery options, given constraints on feasibility of implementing these actions over a wide scale, are discussed.

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INTRODUCTION

Boreal caribou (Rangifer tarandus caribou) populations continue to exhibit widespread declines in both population size and in distribution across Canada, particularly in western Canada (Environment Canada 2012). Recent reviews link these declines to indirect effects of increasing resource development in the boreal forest, which together with climate change interact to alter relationships in the predator-prey food webs in which caribou are embedded (COSEWIC 2002; Vors and Boyce 2009; Festa-Bianchet et al. 2011; Hervieux et al. 2013; Dawe et al. 2014). In particular, disturbance-induced increases in the numerical and functional responses by predators[1], which lead to increased coincidental mortality rates in caribou (Seip 1992, Wittmer et al., 2005, Whittington et al. 2011), is widely considered to be the primary threat to woodland caribou populations. Other potentially important threats to boreal caribou may occur primarily through loss of habitat (directly through vegetation removal, or indirectly through fragmentation and/or loss of access to habitat areas), localized hunting in some herds, and a heightened risk of disease transmission through contact with expanding populations of white-tailed deer (Odocoileus virginianus) and other ungulates (Environment Canada 2012).

The demographic characteristics of boreal caribou limit their potential to recover from population declines (Environment Canada 2012). Their general demographic characteristics are: (1) a late age-at-first-calving by female caribou, typically at 3 years of age; (2) low per-capita calving rate of only one calf per year (Bergerud 2000); and (3) generally high predation-caused calf mortality rates, particularly within the first 30 days of life (Bergerud and Elliot 1986; Gustine et al. 2006). This pattern can lead to low first-year calf survival rates which are frequently insufficient to compensate for annual adult mortality (also primarily due to predation), a situation that is especially problematic in already small and declining populations (Seip 1992, Wittmer et al. 2005). Therefore, small subpopulations containing few adult females (and hence few births) may be subject to an increased likelihood of extirpation.

The current recovery goal for boreal caribou is to achieve self-sustaining populations in all local population units (LPUs) within their current distribution, and this goal is considered technically and biologically feasible although many small populations are currently not self-sustaining (Environment Canada 2012). It is clear that coordinated management efforts linking multiple jurisdictions, including First Nations and a variety of other stakeholders, will be required to achieve this goal, particularly for the smallest and most vulnerable populations. While restoring ecosystems to provide habitat conditions that would support caribou populations over the long-term is an overall goal of current planning and recovery efforts, creating these habitat changes (even if successful) requires years or decades to achieve. In cases of small, declining populations, some

[1] Numerical responses by predators can occur as a consequence of disturbance events that convert old-seral vegetation to young-seral vegetation that supports other ungulate species which act as primary prey for many predators (Serrouya et al. 2011). Functional responses can occur as a consequence of activities that cause removal of vegetation and/or maintenance of vegetation in a young-seral stage condition (e.g., linear features) leading to an increased probability of encounters between predators and their prey, including caribou (Whittington et al. 2011).

more immediate management actions will be needed to bridge the temporal gap (Wittmer et al. 2010).

Here, we describe a stochastic age-structured caribou demography/management model designed to help assess the efficacy of targeted population management actions, singly or in combination, to achieve the desired recovery trajectory for a given herd. As a case study for this analysis, we chose the five populations of boreal caribou in northeastern British Columbia (BC) as they are currently the focus of recovery action planning by various stakeholders. Using the model, we asked: (1) how effective could the three types of aggressive population interventions, taken singly or in combination, be in achieving recovery targets for the case study herds; (2) which options are most likely to be feasible given current and/or potential conditions of predator pressure, where feasibility is defined by operational constraints; and (3) if costs are taken into account, does the selection of potential management options change?

METHODS

Overview of the Case Study Populations

Study Area

All five populations considered in this study (Calendar, Chinchaga, Maxhamish, Snake-Sahtaneh, and West-side Fort Nelson) are located in northeastern BC (Figure 1). Ecologically, all of the population ranges are within the Boreal White and Black Spruce biogeoclimatic zone (Meidinger and Pojar 1991) and are within the Taiga Plains ecozone (except the eastern portion of the Chinchaga herd's range, which overlaps the Boreal Plains ecozone). The general climate in this region can be characterized as having frequent inflows of arctic air masses, with long very cold winters and short growing seasons. The mean annual temperature for long-term climatic stations within the zone is -2.9 to 2oC. Annual precipitation averages between 330 and 570 mm with 35-55% of this occurring as snowfall (Meidinger and Pojar 1991). Vegetative cover is dominated by white spruce (Picea glauca) and with lodgepole pine (Pinus contorta) on drier sites, with many wetlands and muskeg that are surrounded by black spruce (Picea mariana) and tamarack (Laric laricina)(Demarchi 2011). Fire impacted areas often result in regeneration of trembling aspen (Populus tremuloides) and willow (Salix spp.) patches (Demarchi 2011).

All of the study ranges have been affected by both anthropogenic disturbance and wildfires. The proportion of buffered disturbed area in each range is estimated to be between 34% (part of the West-side Ft. Nelson range) to 87% (Snake-Sahtaneh), with the majority being anthropogenic in origin (see Environment Canada 2011 for estimation methods).

Study Population Demography

Boreal caribou in the Taiga Plains ecozone use open coniferous forests with abundant lichen in both summer and winter (Environment Canada 2011). Breeding takes place in late September to mid-October, with dominant bulls breeding several cows. During

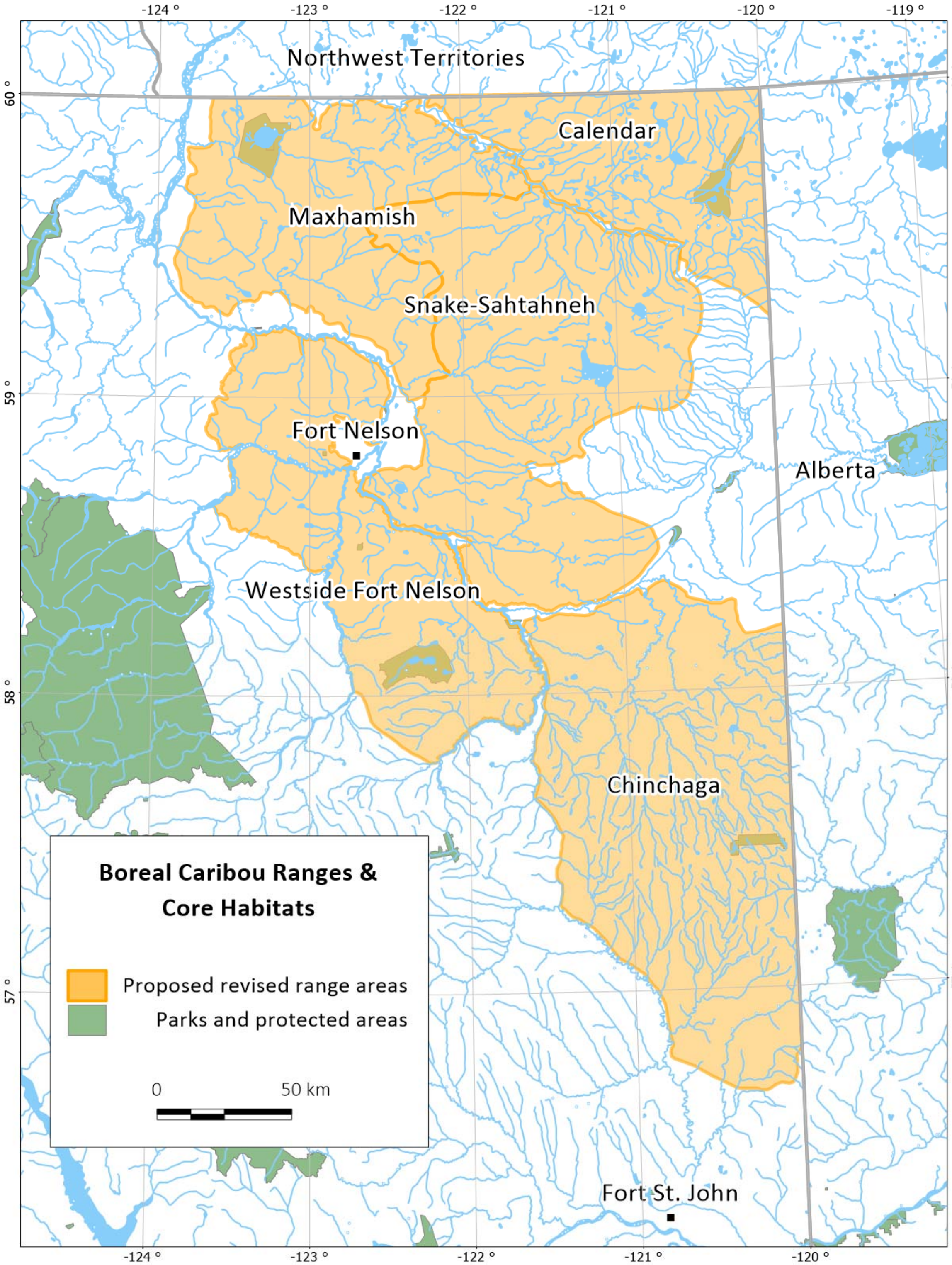

Figure 1. Geographic location of the boreal caribou ranges included in this study. Source by S. Wilson (pers. comm.).

calving in late May or early June, female boreal caribou are often solitary and seek areas of deciduous thickets along lake shores or small islands of black spruce or mixed forests within peatlands, to avoid predators. Annual reproductive output of boreal caribou is low. Wolves (Canis lupus) are the primary predator of boreal caribou (calves and adults); however, predation by black bears (Ursus americanus) and grizzly bears (U. arctos) may also be significant in some areas, especially during spring and early summer (Zager and Beecham 2006; Pinard et al. 2012)

Current population-specific demographic parameters for the five populations in this study are given in Table 1. Each of the study populations appears to be independent (i.e., no effective immigration from, or emigration to, the other populations). Recent published estimates of annual recruitment rate (# calves: 100 cows) in populations of boreal and mountain caribou ecotypes are 28.4 (DeCesare et al. 2012), as well as a 10% coefficient of variation (CV) (Whittington 2014). Similarly, estimates of female survival rates from these studies are 0.844 (Wittmer et al. 2010) and 0.874 (DeCesare et al. 2012), with an estimated annual CV of 8.3% (Whittington 2014). Note that the current rates of adult female survival in the five NE herds are consistent with the corresponding estimates from the literature, while their annual rates of recruitment are lower than these estimates (compare the weighted means and CV in Table 1 with the above values)

Overview of the Caribou Population-Management Model

We implemented a stochastic, age-structured caribou population dynamics model assuming no density-dependence, for this study. In the model, each caribou population was explicitly represented by annual age classes (up to 19 classes, including calves of the year) and both sexes, tracked in a number of "pools" (see Figure A-1). For each study population, the model projects annual numbers by sex, age, and "pool" for a given projection period[2]. A "pool" is a component of the population affected by a direct management activity. Only females were modeled in pools, other than the "wild" pool, as we assumed that males are unlikely to be managed. We excluded potential density-dependent effects on reproduction or mortality from the model because the current sizes of the focal caribou herds are well below the sizes at which density-dependent factors are likely to exert a significant influence on population dynamics over the projection time horizon, even assuming an eventual successful recovery of the population, as defined by the BC government (BC MoE 2011). The model is implemented in R 3.3.1 (R Core Team 2016).

Reproduction was modeled using age-specific parturition rates. Age-, sex- and "pool"-specific mortality was specified by rates that can be partitioned among two time periods within the annual cycle: calving (May-June) and post-calving (July-March) periods. The effects of wolf-caused mortality on calves or adults was modelled as an option of either: (1) the consequence of a target wolf density or (2) an estimated wolf density derived from current recruitment and survival rates (if no target density is specified) using the calf recruitment and adult female survival relationships in Bergerud and Elliot (1998; Bergerud 2007). Other potential sources of mortality, including by other predator species (e.g., bears, wolverines [Gulo gulo]; cougars [Puma concolor]), malnutrition, accidents, or unknown causes, were modeled by their respective proportions of the overall annual

[2] For this project, the projection time period is 10 years.

Table 1. Current (circa 2015) demographic estimates for the five designated boreal caribou herd ranges in north-eastern BC, including the provisional West-side Fort Nelson population range. Sources for these estimates are given in the footnotes to the table.

Range	Current pop'n size N_0 (CV)[1]	# Calves: 100 Cows[2]	Annual survival rate (adult females)[3]	Bull:Cow ratio	Empirical Lambda[4]
Calendar	290 (33 %) min. est.: 81	22	0.96	.25	1.07
Chinchaga[5]	250 (10 %) min: 189	8	0.81	.23	.84
Maxhamish	300 (25 %) min: 81	21	0.81	.38	.90
Snake-Sahtaneh	360 (15%) min: 258	18	0.87	.43	.95
West-side Fort Nelson[6]	79 (79%) min: 60	7	0.87	0.23	0.89
Means (5 pop'ns)	228	16.9	0.86[7]	0.35[7]	0.93[7]
SD (5 pop'ns)	n/a	2.17	0.03	0.03	0.02
Wt'd %CV (5 pop'ns)	40%	12.83	3.35	3.87	2.11

[1] Population size estimates are taken from Table 6; BC MoE (2010) and are similar to estimates shown in Table 11, Environment Canada (2011). Coefficients of variation (CVs) are taken from standard deviations presented in Table 3; Wilson et. al. (2010). The most recent minimum population sizes are taken from Diversified Environmental Services (2015).
[2] Data for 2015 from Diversified Environmental Services (2015). Data for 2014 from S. Wilson (*pers. comm.*)
[3] Data for 2015 from Diversified Environmental Services (2015).
[4] These are population growth estimates for 2015 calculated using the formula developed by DeCesare (2012).
[5] These are pooled estimates (S. Wilson, *pers. comm.*).
[6] Includes data for the Prophet and Parker RRAs for 2014 and 2015.
[7] Geometric means.

mortality rate as determined from field investigations of mortality on radio-collared individuals (Diversified Environmental Services 2015).

Year-to-year stochasticity in demographic rates were modelled using coefficients of variation estimated from field studies and/or obtained from literature sources, and were applied to the individual estimates of demographic parameters using random draws from specific distributions (see Appendix A). Summary statistics of projections are therefore based on multiple replicates of population projections (see below).

Direct management actions and interventions to reduce projected source- and age-specific mortalities from one or more of the above sources were applied to individuals in each "pool". The effects of the following action on population parameters were modeled as: (1) annual levels of reductions in mortality due to management of predators and/or in other sources of mortality; (2) annual maternal penning of pregnant females during pre-calving and calving months of the year, with selected individuals drawn either from the focal population, from external populations, or both; and (3) augmentation of the pool of pregnant females via translocation of pregnant females obtained from external sources (e.g., other populations, captively-bred animals). Among-year patterns of management actions can be specified (see Scenario Descriptions for examples).

Population parameters needed to initiate and run the model were obtained from annual recruitment surveys, from long-term studies of mortality rates on GPS/telemetry collared animals, from recent studies of captive breeding (Traylor-Holtzer 2015), and from on-going maternal penning experiments[3]. Stochastic effects were modelled using coefficients of variation (estimated from field studies and/or obtained from literature sources) and applied to the individual parameter estimates using random draws from specific distributions (see Appendix A: Annual Cycle and Order of Events). Multiple replicates of population projections were therefore used to create the following summary statistics of the projections:

1. Outputs as text file summaries (csv format) of time-series: (1) expected total population size at the end of each projection year, stratified by stage (calves, year 1+ individuals), and sex; (2) more detailed projections by age (year); and (3) projected survival rates by age/sex as a result of management.
2. Post-processed summaries of the time series output files containing probabilities of observing different types of population growth trends in response to the management actions by the modeled populations.

The present model structure is described in more complete detail in Appendix A, including definitions of the parameters for estimating population dynamics, and the effects of different management actions on specific aspects of caribou demography.

Estimates of Demographic Parameters

Population parameters used in the model consist of three sets: (1) initial population size $\boldsymbol{N_0}$ and age distribution (Table B-1); (2) demographic parameters related to annual survival by age and sex (Table B-2) and parameters defining how management affects these (Table B-3); and (3) parturition rates by age (Table A-3). While estimates of the initial population size for each population were available from annual population surveys, the age-structure of each population was not. Therefore we generated an initial age distribution by assigning the observed number of individuals to ages 1-18 at random. Sexes were assigned using the observed bull:cow ratios for each population.

Design of Management Scenarios

The model currently implements three types of management actions that can be applied according to an annual schedule: (1) a reduction in wolf density; (2) protective maternal

[3] See http://rcrw.ca/ and https://vimeo.com/161226373 (accessed 161210).

penning for breeding females and (3) population augmentation (modeled as a number of pregnant females from one or more external sources added to the population). A proportion of augmented females can also be penned in this model. A potential fourth action, management of other predators (e.g., cougars, bears and wolverines) can also be simulated in this version of the model, but was not implemented for this analysis, because: (i) cougars are not known to be present in the case study area and (ii) current estimates suggest caribou mortality by bears and wolverines in the study region is low (Diversified Environmental Services 2015).

We explored a range of methods and schedules for implementing these management options to assess the relative efficacy of the actions (singly or in combination) to influence population growth trends, as well as to give a range of possible economic and operational constraints to consider. In general, implementation of direct management actions was for eight years in these scenarios, or 80% of the 10-year projection horizon.

The scenarios we undertook are outlined below.

1. No management: This default case specified no management action, and current estimates of survival and recruitment were used for the "wild" (unmanaged local population) pool.
2. Predator management: Here we focused entirely on wolf management. We characterized management of wolf populations by:
    a. Intensity. Six levels of wolf density target (low to high: 3.0, 5.0, 7.0, 9.0, and 11.0 wolves per 1000 km2). Based on Bergerud (2007), we expected that wolf densities of <6.5 wolves/1000 km2 should permit stable caribou populations. Densities lower than 6.5 were included because caribou populations should have a high chance of sustained growth at low predator density if wolf predation is the primary cause of declines.
    b. Pattern (2 types):
        i. continuous for eight years
        ii. pulsed (3-years on; 2-years off, 3-years on). Note that pulsed requires less overall management effort (75% of the continuous pattern)
3. Augmentation: We increased population size by translocating female animals from a source external to the herd. We characterized augmentation efforts by:
    a. Source (2 levels): from external "wild" populations or from captive breeding. In scenarios, translocations from either or both sources can be specified. If both sources are specified, 50% of the total number of females (see below) are derived from each source.
    b. Intensity: Total numbers of translocated females/year (3 levels): 0, 15, 30.
    c. Percent of the females translocated that were also penned (4 levels): 0, 15, 50, 80.
    d. Pattern of translocations (2 types):
        i. continuous (for 8 years)
        ii. pulsed (translocations for first 3 years, no translocations for 2 years, translocations again for 3 years)
4. Maternity Penning: We protecting cows and calves from predation during the calving season in a protected maternity pen facility. We characterized penning by:
    a. Intensity: Percent of females penned (4 levels): 5/yr, 15/yr, 50/yr, and 80/yr.
    b. Pattern of penning (2 types):

i. continuous for eight years
ii. pulsed (penning for first 3 years, no penning for 2 years, penning again for 3 years)

Note that existing maternity pens are frequently reused from year-to-year, although the annual set-up and operating costs are nearly the same among years (McNay et al. 2016).

In addition to the defined management scenarios, we included parameter adjustments to address post-translocation survival depression (PTSD) which appears to be an important issue in ungulate management (Friar et al. 2007; Whittington 2015). PTSD can occur due to translocated animals having naivité about both the structure and location of local habitats and the predator community. Survival of caribou in past translocations have been variable (from 33% to 67%; Cichowski et al. 2014), and for elk may be lowered by as much as 50% in their first year after translocation (Friar et al. 2007). Because recent field data from the Klinse-Za maternity pen experiment has not yielded PTSD results as low as these (although sample sizes are small) (McNay et al. 2016), and because the diversity of predators on boreal caribou are expected to be lower than in some of the cited studies, we were slightly more optimistic for our modeling. We chose two possible PTSD levels to be applied to translocated females (low: 10%; high: 27% depression in the year after translocation).

Steps in the Modelling and Analysis

We undertook the analysis in three main steps. First, we projected each of the four scenarios (above) for each population using a factorial design (i.e. all combinations of all actions applied at least once). We projected a total of 4,559 potential scenarios, with 50 replicates of each, to build a database of demographic outcomes from which demographic outcome indicators were calculated. Second, we applied feasibility criteria to select the feasible scenarios for each population, and calculate costs for that subset of scenarios. Third, we calculated costs for each management action and expressed these in relation to the population indicators projected by the model.

Demographic Outcome Indicators

As described above, the primary indicators of projected demographic outcomes calculated for the female component of the population are:

1. the probability that annual population growth (λ) ≥ 0.99 (i.e., the probability that the size of the population is approximately stable or increasing through the time horizon);
2. three probabilities for annual population growth consistent with those calculated by Environment Canada (2011):
    a. probability that annual population growth (λ) > 0.99 (termed the probability that the size of the population is approximately stable or increasing through the time horizon);
    b. the probability that annual population growth (λ) ≥ 1.0 (termed the probability that the size of the population is approximately increasing through the time horizon); and
    c. the probability that annual population growth (λ) > 1.0 (termed the probability that the size of the population is definitely increasing through the time horizon).

3. the probability that λ for the population is ≥ 1.0 over successive 3-year periods. That is, the projected population can demonstrate consistent positive growth for multiple years in succession;
4. the probability of avoiding an extirpation threshold (e.g., dropping below 10 reproductive females at least once during the time horizon);
5. the probability of reaching a specified target size of reproductive females. This target was set at 250 females for each population, a size consistent with a low likelihood of catastrophic extirpation (e.g., Environment Canada 2011).

Note that indicators 2a-c are hierarchically nested, and represent successively more stringent criteria for determining the likelihood that the population can continue to persist and grow over the time period.

Each of these outcomes can be considered a criterion for evaluating the utility of each management action in achieving the specified recovery goals for boreal caribou populations. As such, they form candidate criteria or components that could enter in to a decision model.

Assessment of Feasibilities and Costs

Operational Feasibility

In this analysis, we considered operational feasibility of the modeled management actions at the scale of each individual population. In particular, we set feasibility thresholds on the following three management actions: (1) annual capacity (total number of females) for a maternity pen (one per population); (2) potential numbers of females from other external populations that could be translocated into the focal population in any one year (assuming source females are available), and (3) similarly, potential numbers of captively bred females that could be translocated into the focal population in any one year (assuming a captive breeding facility was available).If any one of these constraints is exceeded during the time period in which management actions are applied for a given scenario and simulation replicate, then that particular management scenario was marked “infeasible”. As it is possible for different populations to vary in terms of the feasibility of these different factors, we applied different feasibility thresholds for each of them (Table 2). These levels were applied factorially to the outcomes of the scenario projections.

Costs of Implementing Management Actions

Each direct management action has its own set of costs, both capital and operational. We estimated six different costs for the mix of scenarios undertaken in this study (Table 2). We interpreted these costs as annual costs, apportioning capital investment costs (if any) among the total number of years a management action is applied, in addition to the estimated annual operating cost of that action. Costs were calculated in 2016 dollars, and were not discounted. The sources used to estimate each cost are given in Table 3.

Note that there are potential constraints that could apply at the regional level (i.e. among-populations, such as the total number of females available for augmentation from either source (external population or captively-bred) that we did not consider.

Table 2. Proposed levels of operational feasibility of different management actions that could be applied at the level of each population. Units for each factor are numbers of females/year.

Annual Capacity of Maternity Pen[1]	Availability of Females (external) for Translocation (augmentation)	Availability of Females (captively-bred) for Translocation (augmentation) [2]
0	0	0
15	10	10
25	20	20
35	30	30

[1] Variables constraining capacity of maternity pens include: size of area that can be enclosed each year; food supply; number of years a pen has been established in the same location; and number of trained personnel that can oversee the pen through the entire penning period.
[2] These numbers are within the values currently being discussed for boreal caribou (see IUCN 2016).

Table 3. Estimated annual costs of implementation of each management action. The source or background for each cost estimate is given in the comments section of the table.

Cost Type	Annual cost ($/yr)	Description and Sources
Predator Management	$2,000 per wolf removed[1]	Source: Seip *pers. comm.* (Jan. 2016)
Maternity Pen	$550,000/pen	Sources: current experience with the Klinse-Za maternity pen indicates $550,000 as a recurring annual cost (including set-up and operating and monitoring costs, but excluding predator control costs). Other sources: Hayek et al. (2016), Smith and Pittaway (2011).
Translocation from external population into focal population	$7,2623/female; pulsed implementation scenarios). $9,163/female; continuous implementation scenarios.	Source: Hayek et al. (2016); see also Kinley (2009). Costs combined 1st year costs (translocation), with subsequent years costs (post-release monitoring of the fates of each translocated individual.)

Captive-breeding and release into focal population	$21,163/female; pulsed implementation scenarios). $23,263/female; continuous implementation scenarios.	Hayek et al. (2016). Costs are approximate, and based on an assumed yield of 100 females/year from a centralized facility. Current assumptions at the facility level are: $10 million to install a facility, and $4 million/yr to operate it. Additional costs are translocation/female and monitoring post-release).

[1] Predator management cost was estimated as cost per wolf removed/yr. The annual expected number of wolves removed from the population per year was estimated as the difference between the current estimated density of wolves for the population[4] and the target wolf density as set on the management scenario.

In the analysis, these costs were summed for each scenario as follows: (1) total cost per managed female (penned, translocated, and/or captively-bred); (2) total cost per managed calf that recruited into the population (i.e. recruiting calf originating from a penned female, or either a translocated or captively-bred female), or (3) total cost for all recruited calves (managed or unmanaged).

RESULTS

As described in Methods, we first briefly present overall results of the demographic model, before presenting results that consider both feasibility and costs.

Overall Demographic Response Patterns

We found that most management scenarios (94%), across all population states and wolf densities, resulted in projected population trends achieving at least some (i.e. > 10%) probability of showing a stable or increasing trend over 10 years. Of these scenarios, a substantial percentage (84%) yielded > 90% probabilities of a stable or increasing trend over 10 years (Figure 2). Differences between the shape of the cumulative distributions of the three indicators of annual population growth trend at a given wolf density were relatively small (compare panels vertically). However, it is clearer that fewer scenarios

[4] As current densities of wolves was not known at the time of this analysis, we estimated density using the Bergerud equation (Fig. 1; Bergerud 2007) with the current estimated annual recruitment and annual adult survival rates, and averaging the resulting interpolated density consistent with those rates. This density was translated into an expected number of wolves on the basis of the area of each population unit.

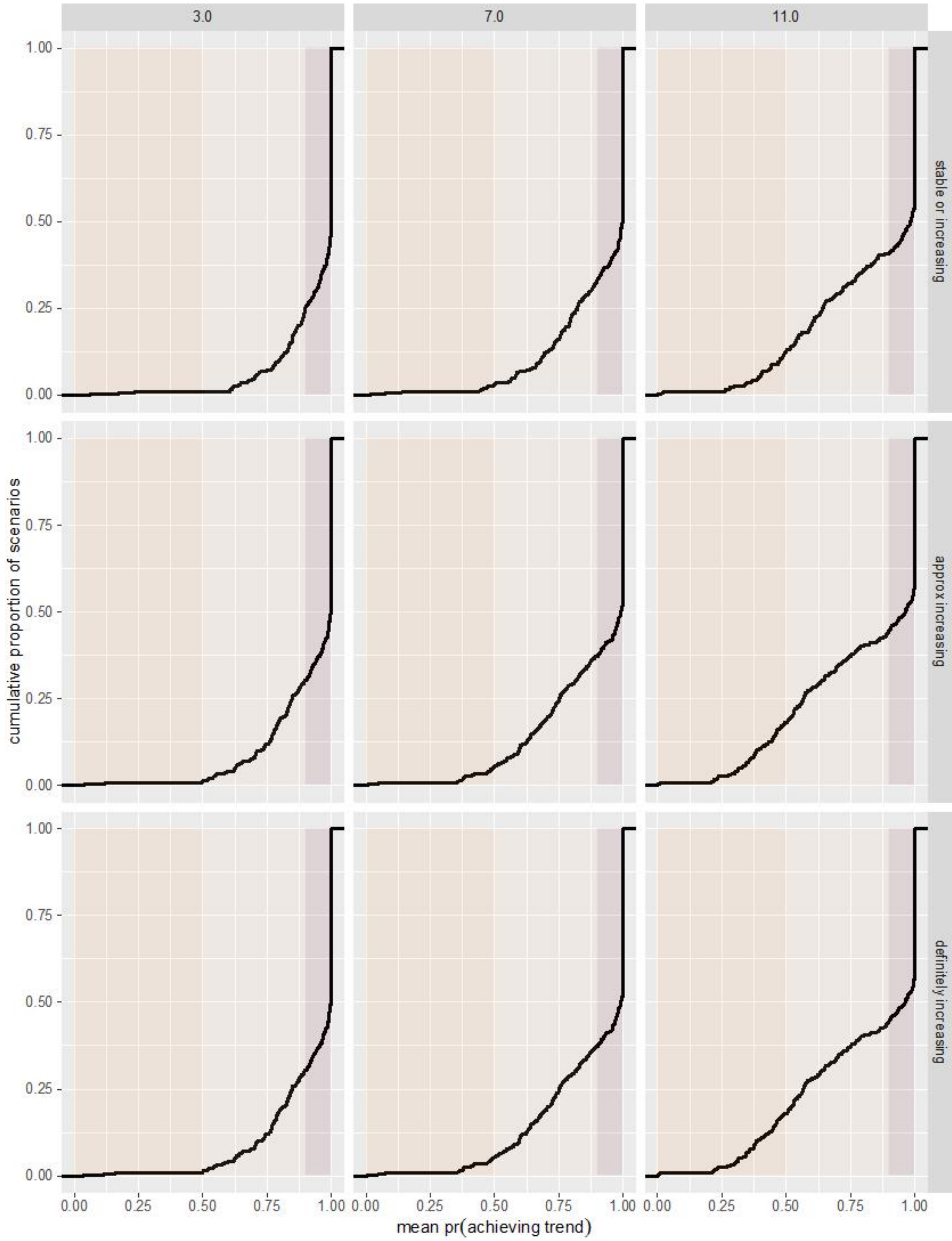

Figure 2. Cumulative proportions of the management scenarios (y-axis) that cause the projected populations to attain the three different trend conditions (panels from top to bottom) with the given probability (x-axis) for three levels of wolf density (#wolves/1000 km2; panels from left to right). Shading indicates different "zones" of probability (low: 0.0-0.5 [tan], moderate: 0.5-0.9 [light tan], high: 0.9-1.0 [dark tan]. Mean projected population trends are calculated averaging over each population x management scenario (i.e. across all demographic assumptions).

(combinations of management actions) appear to be required to achieve a given state of annual population trend with high probabilities under conditions of low wolf density compared with high wolf densities (compare graphs horizontally).

We expected that the probability that populations have a lower probability of achieving a given management objective for population growth (i.e. as defined by one of of the three demographic population growth indicators; see Methods for definitions) over the 10-year time horizon would be negatively related to a decreasing value of target wolf density, inclusive of all other management actions modelled. Management of wolves to lower population densities appears to significantly affect the apparent success of all other caribou population management options across all six boreal caribou populations (Figure 3). These general patterns held across all modelled populations, although populations differed considerably in their mean probabilities of becoming approximately stable or increasing at densities of wolves < 7.0 wolves/1000 km2 (i.e. near to or below the 6.5 wolves/1000 km2 density level that Bergerud (2007) estimated was required for stabilizing a population of woodland (boreal) caribou (Table 4). In addition, applying management of predator density continuously yields a small improvement in the probability of stabilizing or increasing caribou populations compared with the less intense pulsed pattern of applying predator management actions (Figure 4). This suggests that continuously managing predators yields a small improvement in the probability of stabilizing or increasing caribou populations compared to the pulsed pattern of predator management (Figure 3).

See Appendix C for examples of the distribution of demographic parameters as projected by the model under the "no management" scenario.

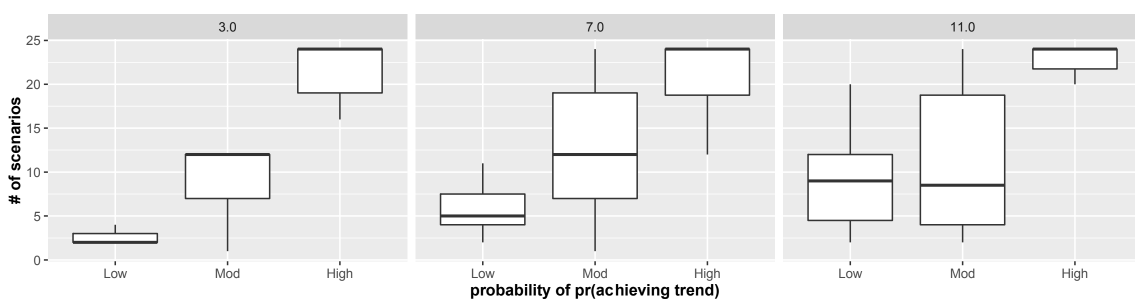

Figure 3. Boxplots of the frequencies of scenarios (y-axis) that cause the projected populations to achieve the trend conditions (combined here) with the given probability "zone" (x-axis: see Fig. 2 caption) across three levels of wolf density (#wolves/1000 km2; panels from left to right).

Table 4. Population specific mean values for key output indicators projected across all management scenarios modeled.
For simplicity of presentation, we show below the projected mean values for each modelled parameter at a constant wolf density target (e.g., 7.0 wolves/1000 km2).

Population	Outcome Indicator		
	Population Trend[1]		Vulnerability
	pr($\lambda \geq 0.99$)	pr($\lambda > 1.0$)	pr($N \leq Q_{Ext}$)
Calendar	0.96	0.93	0.0
Chinchaga	0.69	0.65	0.0
Maxhamish	0.86	0.81	0.0
Snake-Sahtaneh	0.84	0.79	0.0
West-side Fort Nelson	0.97	0.95	< 0.001

[1] Because for most populations the results for the annual population trend indicators pr($\lambda \geq 1.0$) and pr($\lambda > 1.0$) similar, we illustrate only the latter indicator in the above table.

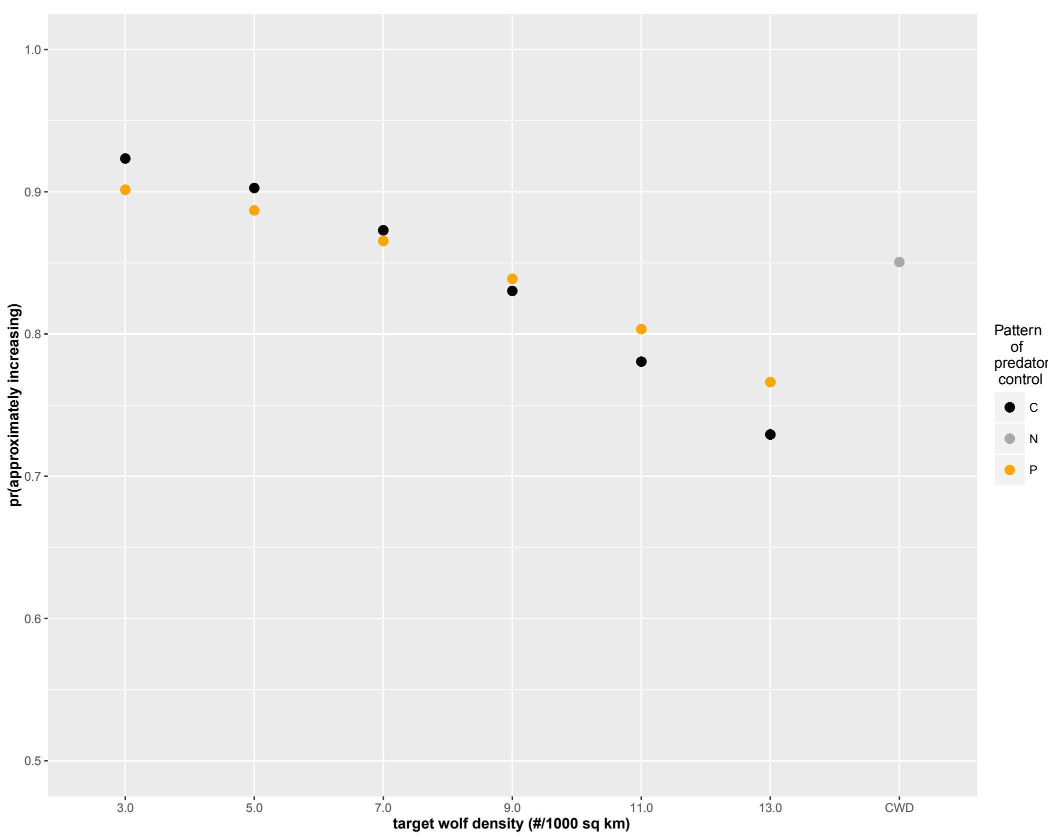

Figure 4. Mean probability that projected population growth over the 10-year time horizon is approximately increasing [pr($\lambda \geq 1.0$)] as a function of target wolf density (wolves/1000 km2) and pattern of predator management applied across all scenarios of managed and unmanaged conditions. C=”continuous”; P=”pulsed”; N=”none”. The grey point for “N” is where assumptions for currently assumed wolf density (averaged over all populations) were used. Label CWD = “currently assumed wolf density”.

Effects of Feasibility and Costs on Selection of Management Actions

Applying feasibility thresholds (Table 5) reduces the number of scenarios that may be feasible to implement for a given population by an average of 84.3%. We therefore focus on this "feasible" subset of the scenarios in the remainder of our analyses. We selected the probability of annual projected population increase [pr(annual $\lambda > 1.0$)] as our nominal management objective for the summaries below, as it is the most stringent of the population growth indicators[5].

Effects of Operational Feasibility Constraints on Outcomes for Populations

We found that all modeled management actions can be feasibly applied to one or more of the study populations to improve population outcomes under certain conditions, whether those actions are implemented singly or in combination with other actions. For currently declining populations (which includes all our study populations except Calendar; see Table 1), each of the management actions, applied singly or in combination, leads to an increase in the mean probability of achieving the annual population growth management objective we selected [pr(annual $\lambda > 1.0$); see above] relative to "no management" (Table 5). Model results suggest that any one management intervention by itself is not usually sufficient to ensure more than a small increase in probability of achieving this objective (performance). While the range of mean improvement in performance is quite wide (0.010-0.762), most of the improvements in performance that are substantial (e.g., mean probability gain > 0.5) involve at least two of the management actions applied together (e.g., maternity penning + predator control; translocation of captively-bred females + predator control) with at least one of those actions implemented at medium to high levels of effort. For example, just managing predator density down to a low level is unlikely to result in a large increase in the probability of ensuring that currently declining populations will increase annually, while predator management in combination with any of the other actions (maternity penning, translocations from either wild or captively-bred sources) substantially increases the probability of population growth within these herds. Unsurprisingly, any management action that involves repeated augmentation of the declining population has, on average, a more pronounced positive benefit on population performance than do scenarios without augmentation (i.e. predator control only, maternity penning only). Only at large maternity pen sizes (i.e. 35 females) will this action, implemented in combination with predator control, overlap with the projected benefit values obtained under options involving translocations.

[5] Note that this performance target is more difficult to achieve than the standard defined by Environment Canada (2011): probability of populations being approximately stable or increasing, or pr(annual $\lambda \geq 0.99$).

Table 5. Mean change (gain) in the probability of growing populations that are currently declining each year, calculated for feasible direct management action combinations relative to no management. The actual mean value of this performance indicator [pr(annual λ > 1.0)] for these declining populations under the "no management" scenario is shown in bold. Note that for clarity, we do not show all values for the different management actions that we projected in scenarios.

Maternity Pen Size	# Females Translocated (external)	# Females Translocated (captively-bred)	□pr(annual λ > 1.0)		Benefit of predator mgmt
			no predator mgmt	with pred mgmt[2]	
none	none	none	> 0.001	0.010	0.010
none	none	10	0.210	0.372	0.162
none	none	30	0.450	0.644	0.194
none	10	none	0.215	0.373	0.158
none	10	10	0.329	0.524	0.194
none	10	30	0.410	0.609	0.199
none	30	none	0.447	0.641	0.195
none	30	10	0.407	0.607	0.200
none	30	30	0.465	0.666	0.200
15	none	none	0.275	0.311	0.036
15	none	10	0.313	0.458	0.145
15	none	30	0.483	0.661	0.178
15	10	none	0.331	0.453	0.121
15	10	10	0.384	0.557	0.173
15	10	30	0.447	0.628	0.181
15	30	none	0.476	0.658	0.182
15	30	10	0.440	0.626	0.186
15	30	30	0.490	0.678	0.188
35	none	none	0.361	0.424	0.063
35	none	10	0.510	0.605	0.096
35	none	30	0.618	0.747	0.128
35	10	none	0.517	0.606	0.089
35	10	10	0.572	0.686	0.114
35	10	30	0.606	0.730	0.124
35	30	none	0.617	0.744	0.126
35	30	10	0.605	0.729	0.124
35	30	30	0.633	0.762	0.129

[1] Predator densities are left as estimated for current conditions (i.e. "current wolf density").
[2] Results here are for predator densities managed to a density of 3.0 wolves/1000 km^2.

Patterns in Costs to Achieve Population Growth Benefits

Among the feasible management scenarios for declining populations that we examined, the costs of management varied widely (Figure 5), as might be expected given the range of scenarios we modelled. The trend across all feasible scenarios is that as the relative expected gains in the chosen performance indicator increase, so do the costs required to achieve those benefits. The main exception to this pattern is at low expected gains, where the high investment costs of establishing maternity pens and/or captive breeding facilities are significant enough to affect the utility of those options for small numbers of animals handled via those options (see Figure 5). Second, the general pattern of management costs incurred (including the cost of predator management) is lower at lower wolf densities, especially for relative gains in the population performance indicator >0.75 (compare left and right panels of Figure 5). Third, many of the management options that yield higher relative gains in performance are also “inefficient” in terms of their cost at any level of wolf density (i.e. lie well above the trend line in Figure 5).

Interactions among Actions that Achieve Different Objectives for Population Growth

For declining populations, there are trade-offs in the combinations of management actions that are likely to achieve different levels of gain in the probability that such populations will show annual growth over the time period relative to no management, depending on the options that are actually available. We illustrate this in Table 6, which shows a subset of scenario results yielding three different levels of gain (0.25 [low], 0.50 [moderate], 0.75 [high]). It is evident from the results that achieving wide gains in population growth are possible if females are available for translocation (from both external populations and/or a captive breeding facility and can potentially reduce the need for concurrent maternity pens especially if wolf densities can be kept relatively low and the objective for population growth is also low. However, in general, if the pool of females available for translocation is small and no captive breeding facility exists (a more realistic situation), then larger numbers of females need to be penned each year and predator management is required to achieve higher levels of gains in likelihoods of population growth.

DISCUSSION

Evidence continues to build that on-going declines in many populations of boreal caribou, particularly those whose ranges overlap with anthropogenic disturbances, are being driven by anthropogenically induced increases in apparent competition, mediated by wolves and broad-scale habitat fragmentation and loss (Hervieux et al. 2013). Given that the stated conservation goal for caribou at both federal and provincial levels is recovery of declining populations (B.C. MoE 2011; Government of Alberta 2011; Environment Canada 2012), a commitment to short-term interventions to prevent extirpation of small populations and effective long-term habitat conservation and restoration will be needed (Hervieux et al. 2013). Understanding the potential benefits of making large-scale investments in conservation management options has implications for both the public and private sector in ensuring conservation goals have a possibility of being realized.

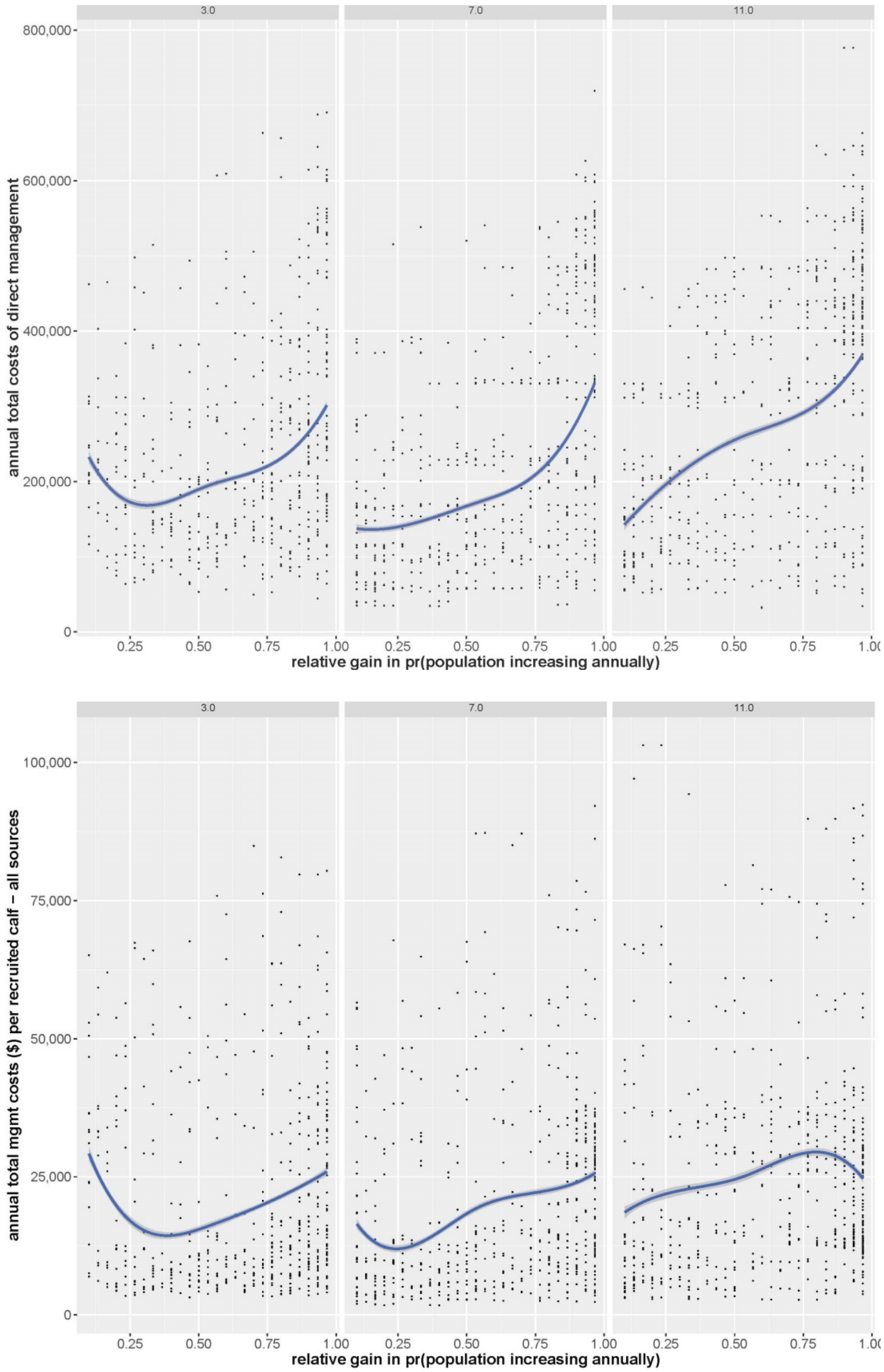

Figure 5. Total annual costs of direct management (top panels) and annual costs per recruited calf (bottom panels) required to achieve different levels of benefit (i.e. relative gain in the probability that the population increases annually (x-axis) for feasible scenarios for all declining populations. A b-spline-smoothed line through all points is shown in blue. The different panels (left to right) show results at three levels of wolf density (#wolves/1000 km2).

Table 6. Top-ranked combinations of management actions that meet three different probability target values for annual population growth > 1.0 for declining populations of NE boreal caribou. Shown are the scenarios whose frequency was in the top 5% of the list of scenarios meeting each target value. A) all possible management actions are available; b) table excludes scenarios specifying annual translocations of > 10 females and the presence of a captive-breeding facility. Costs are average total management costs (in millions of $) over the projection period for a single population.

0.25 ± 0.025						0.50± 0.025						0.75± 0.025					
TWD[1]	Pred. man.	Mat. Penning[1]	Translocation #/yr	Translocation Src[2]	Avg. total cost/pd	TWD	Pred. man.	Mat. penning	Translocation #/yr	Translocation src	Avg. total cost/pd	TWD	Pred. man.	Mat. penning	Translocation #/yr	Translocation src	Avg. total cost/pd
A) All management options available																	
13.0	N	<20(C)	15(C)	ext	4.97	3.0	Y (P)	<20	30	ext	6.20	5.0	Y (C)	<20	15	cbf	5.40
9.0	N	0	15(P)	cbf	1.44	5.0	Y (C)	<20	15	ext & cbf	5.98	7.0	Y (P)	<20	15	cbf	4.78
9.0	Y	<20(C)	15(C)	cbf	6.06							7.0	Y (P)	<20	15	ext	3.87
5.0	Y	0	15(P)	cbf	1.66							CWD	N	<20	15	cbf & ext	5.75
B) translocation/year ≤; no captive breeding facility																	
13.0	N	<20	15(C)	ext	4.97	13.0	N	25	0	---	3.33	5.0	Y	25(C)	0	---	4.53
13.0	N	25	0	---	3.30	9.0	Y	20	0	---	4.40	3.0	(C)	25-30 (C)	0	---	3.49 – 4.51
CWD	N	30	0	---	3.83	5.0	Y	<20	15	ext	5.67						

[1] see caption for Figure 4 for definitions of "TWD" values, including "CWD"
[2] see caption for Figure 4 for definitions of "C" and "P". See also Methods.
[3] "ext" = source is from one or more external ("wild") population(s); "cbf" = source is from a captive-breeding facility.

Overall patterns identified in this study suggest that across a wide range of demographic situations for the BC boreal caribou herds, there are likely to be one or more management actions that can be taken, singly or in combination, to improve (possibly substantially) the projected likelihood of achieving recovery goals going forward. In addition, the results suggest that no single option applied at the population scale is likely to ensure a high probability of population growth in the short-term. Such favourable population outcomes are only likely if two or more management options are applied in a coordinated fashion and sustained over multiple years. Availability of females for translocation into a focal population (either from an external population or from a captive-breeding facility, despite its added cost, enhances the opportunities of being able to achieve greater gains in the likelihood of annual population growth under a wider range of wolf densities. However, options do exist for achieving gains in population growth even if the options for translocations are limited.

Our projections suggest that broad-scale implementation of predator management by itself is unlikely to achieve substantial increases in the probability of annual population growth over a 10 year projection period for most populations. However, predator management in combination with one or more other actions does increase the probability of annual population growth by 10-20% over the same scenarios conducted without predator management. This suggests that predator management is an important supporting management tool. In addition we found that larger capacities of maternity pens are as effective as small annual translocations (all else being equal) in improving probabilities of annual population growth. Finally, translocations from captively-bred sources are the single most effective action relative to other actions (although translocations from other “wild” populations are nearly as effective), although there is uncertainty around the post-translocation survival probability of translocated females. However, availability of translocated females, and its high cost may limit the applicability of this management action to any but very specific circumstances.

The feasibilities of each management option (e.g., predator management, variable-capacity maternity pens, translocations from populations of a similar ecotype of caribou, or captive-breeding facilities) have constraints that can apply both at the population level and the regional level. For example, (1) the total number of females available regionally for augmentation from either source (external population or captively-bred) may constrain the numbers that could be translocated into one or more population being managed, (2) the number of trained teams available for conducting successful translocations; and (3) the number of trained teams available for predator management. We did not assess the effects of these factors upon operational feasibility in the present study. Clearly, great care is required to select options that may ultimately be realistic and sustainable for a given population.

Wolves are widely considered to the most important current limiting factor of caribou populations throughout Canada, primarily because of their effects on calves (McLoughlin et al. 2003; Gustine et al. 2006; Whittington et al. 2011). Current estimates of wolf densities in the ranges of two of our study populations range from 7.0 (Calendar) to 15.6 (Chinchaga) (Serrouya et al. 2016), and estimates from nearby foothills and mountains range from 9.7 – 22.3 wolves/1000 km2 (Webb 2009), well above the meta-analysis estimate 6.5 wolves/1000 km2 that was consistent with stable caribou population sizes (Bergerud and Elliot (1998); Bergerud 2007). Results from long-term predator management experiments (e.g., Little Smoky, Alta) suggest that this management option can potentially stabilize populations (Hervieux et al. 2013), despite its high social cost.

However, it is also clear from those experiments and from the results of our study that this strategy alone is unlikely to achieve sustained growth in caribou populations, although it can operate as an important supporting action to increase the effectiveness of other types of interventions.

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LITERATURE CITED

Bergerud, A.T., and J.P. Elliot. 1986. Dynamics of caribou and wolves in northern British Columbia. Canadian Journal of Zoology 64: 1515-1529.

Bergerud, A.T., and Elliot, J.P. 1998. Wolf predation in a multiple-ungulate system in northern British Columbia. Can. J. Zool. 76(8): 1551–1569. doi:10.1139/98-083.

Bergerud, A.T. 2000. Caribou. Chapter 11, pages 658-693 in Ecology and Management of Large Mammals in North America. S. Demarais and P.R. Krausmann (eds). Prentice Hall, New Jersey.

Bergerud, A.T. 2007. The need for management of wolves – an open letter. Rangifer, 17:39-50.

B.C. Ministry of Environment. 2011. Implementation plan for the ongoing management of Boreal Caribou (Rangifer tarandus caribou pop. 14) in British Columbia. Victoria, B.C. 17 pp.

COSEWIC. 2002. COSEWIC assessment and update status report on the woodland caribou, *Rangifer tarandus caribou* in Canada. Committee on the status of endangered wildlife in Canada (COSEWIC), Environment Canada, Gatineau, Que.

Dawe, K.L., E.M. Bayne, and S. Boutin. 2014. Influence of climate and human land-use on the distribution of white-tailed deer (Odocoileus virginianus) in the western boreal forest. Canadian Journal of Zoology, 92: 353-363,

DeCesare, N. J., M. Hebblewhite, M. Bradley, K. G. Smith, D. Hervieux, and L. Neufeld. 2012. Estimating ungulate recruitment and growth rates using age ratios. The Journal of Wildlife Management 76:144-153.

Demarchi, D. 2011. The British Columbia Ecoregion Classification. Third Edition. Ecosystem Information Section, Ministry of Environment, Victoria, BC.

Diversified Environmental Services Ltd. 2015. BC Boreal Caribou Implementation Plan: Mortality Investigation Summary Report No. 20: April 2015. Prepared for the SCEK-REMB.

Environment Canada. 2011. Scientific assessment to inform the identification of critical habitat for woodland caribou (Rangifer tarandus caribou), boreal population, in Canada: 2011 update. Ottawa, Ontario, Canada.

Environment Canada. 2012. Recovery Strategy for the Woodland Caribou (Rangifer tarandus caribou), Boreal population, in Canada. Species at Risk Act Recovery Strategy Series. Environment Canada, Ottawa. xi + 138pp.

Festa-Bianchet, M., Ray, J.C., Boutin, S., C.t., S.D., and Gunn, A. 2011. Conservation of caribou (Rangifer tarandus) in Canada: an uncertain future. Can. J. Zool. 89(5): 419–434. doi:10.1139/z11-025.

Government of Alberta. 2011. A woodland caribou policy for Alberta. Government of Alberta, Edmonton. Available http://open.alberta.ca/dataset/debdf4b3-d004-42a1-ab1a-6ac456362915/resource/b72d06b1-e3ec-4bbf-92e9-dfb67e601f49/download/2011-WoodlandCaribouPolicy-Alberta-Jun2011.pdf [accessed Dec. 11, 2016]

Gustine, D., K. Parker, R. Lay, M. Gillingham, and D. Heard. 2006. Calf survival of woodland caribou in a multi-predator ecosystem. Wildlife Monographs, 165:1-32.

Hayek, T., N. Lloyd, M.R. Stanley-Price, A Saxena, and A. Moehrenschalger. 2016. An exploration of Conservation breeding and translocation tools to improve the conservation status of boreal caribou population in Western Canada: pre-workshop document. Centre for Conservation Research, Calgary Zoological Society, Calgary, AB.

Hervieux, D., M. Hebblewhite, N.J. DeCesare, M. Russel, K. Smith, S. Robertson and S. Boutin. 2013. Widespread declines in woodland caribou (Rangifer tarandus caribou) continue in Alberta. Canadian Journal of Zoology, 91: 872-882.

McLoughlin, P.D., Dzus, E., Wynes, B., and Boutin, S. 2003. Declines in populations of woodland caribou. J. Wildl. Manag. 67(4): 755–761. doi:10.2307/ 3802682.

McNay, R.S., L. Giguere, B. Pate, and E. Dubman. 2016. Enhancing calf survival to help avert extirpation of the Klinse-Za caribou herd. Wildlife Infometrics Inc. Report No. 527. Wildlife Infometrics Inc., Mackenzie, British Columbia, Canada.

Meidinger, D. and J. Pojar. 1991. Ecosystems of British Columbia. Special Report Series 6. Research Branch, B.C. Ministry Forest, Victoria, B.C. 280pp plus appendices.

Pinard, V., C. Dussault, J.-P. Ouellet, D. Fortin, and R. Courtois. 2012. Calving rate, calf survival rate, and habitat selection of forest-dwelling caribou in a highly managed landscape. J. Wildl. Manage. 76:189–199.

R Core Team (2016). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Seip, D. 1992. Factors limiting woodland caribou populations and their interrelationships with wolves and moose in southeastern British Columbia. Canadian Journal of Zoology 70:1492-1503.

Serrouya, R., B. McLellan, S. Boutin, D. Seip and S. Nielsen. 2011. Developing a population target for an overabundant ungulate for ecosystem restoration. Journal of Applied Ecology 48:935-942.

Serrouya, R., H. van Oort, C. DeMars, and S. Boutin. 2016. Human footprint, habitat, wolves and boreal caribou population growth rates. Unpublished report.

Smith, K.G., and L. Pittaway. 2011. Little Smoky woodland caribou calf survival enhancement project. Rangifer, 31: 97-102.

Vors, L.S., and Boyce, M.S. 2009. Global declines of caribou and reindeer. Global Change Biol. 15: 2626–2633. doi:10.1111/j.1365-2486.2009.01974.x.

Webb, N.F. 2009. Density, demography, and functional response of a harvest wolf population in west-central Alberta, Canada. Ph.D. dissertation, University of Alberta, Edmonton.

Whittington, J., Hebblewhite, M., DeCesare, N.J., Neufeld, L., Bradley, M.,Wilmshurst, J.W., and Musiani, M. 2011. Caribou encounters with wolves increases near roads and trails: a time-to-event approach. J. Appl. Ecol. 48:1535–1542. doi:10.1111/j.1365-2664.2011.02043.x.

Whittington, J. 2014. Caribou captive breeding release scenarios and population projections. Unpublished report prepared for Parks Canada, April, 2015.

Wilson, S.F., C. Paztor, S. Dickinson. 2010. Projected boreal caribou habitat conditions and range populations for future management options in British Columbia. Unpublished report prepared for the B.C. Ministry of Mines, Energy and Petroleum Resources and the B.C. Ministry of Environment.

Wittmer, H., A.R.E. Sinclair, and B. McLellan. 2005. The role of predation in the decline and extirpation of woodland caribou. Oecologia 144:257-267.

Wittmer, H. R.N.M. Ahrens, and B.N. McLellan. 2010. Viability of mountain caribou in British Columbia, Canada: Effects of habitat change and population density. Biological Conservation, 243:86-93.

Zager, P. and J. Beecham. 2006. The role of American black bears and brown bears as predators on ungulates in North America. Ursus 17:95–108.

APPENDIX A. DESCRIPTION OF CARIBOU DEMOGRAPHY MODEL USED TO EXPLORE MANAGEMENT ACTIONS

Model Description

The model used in this study is a stochastic, age-structured, non-spatial caribou population dynamics model which assumes no density-dependence. The model is designed to model a single population with no emigration/immigration, except through the management action of augmenting the population. Within-year correlations in vital rates are generally assumed to be low (e.g., Whittington 2015), and therefore are not included in population projections made by the model. For each population (called the "focal" population), projections of annual population sizes by sex, age, and "pool" (i.e. the component of the population affected by a direct management activity) are made for a given projection period[6]. Model projections are annual, over a user-defined time horizon (i.e. 10 years for the present study). Details of the demographic calculations are given below.

The present version of the model is Version 3.92a (alpha version). The model itself is implemented in R 3.3.1 (R Core Team 2016).

Age, Sex and Breeding Age Classification

We classified caribou into 19 age classes ranging from age 0 (young-of-year born within the year), and from 1-18 years old (yearlings and year 2+ adults). Sexes are tracked separately, resulting in 38 age-sex classes that are used to update the model population size from N_t to N_{t+1}. Both male and female calves are tracked as a single pooled class within their first year, and then the survivors are split into separate sexes at year-end using the population-level parameter defining sex ratio at birth.

The probability of breeding for year 2+ females is controlled through the parturition rate parameter (see below).

Annual Cycle and Order of Events

Caribou in British Columbia breed in the mid-September - early October rut season, with calves born approximately eight months later in May-June. Most calf mortality occurs in the first two months after birth, and calves must survive both summer and autumn to be included in herd composition surveys (usually conducted in March, depending on snow conditions). Therefore, the model defines the population "year" as running from survey period to survey period, which also approximates the period of capture of females for maternal penning. The population size and composition at this point in the year is taken to be the model end-of-year (**time**$_t$) population status, and this status is used to initialize the next year's (**time**$_{t+1}$) population.

Within this annual cycle, the demographic events are modelled in the following order:

[6] For this project, the projection period is 10 years (see Main report).

1. The numbers of animals in each age class and sex class at the start of the simulation (**N_0**) or the ending population of the previous year after aging to **N_{t+1}** – see step 2.g below) are transferred into the **N_t** data structure representing the population at the start of this year's annual cycle. This data structure is subdivided into a number of "pools" (Figure A-1) as described below. Each pool is explicitly tracked for the year in which it is created. At the end of the year, animals in each pool are aggregated back together again into a single pool (by age and sex) again to represent the focal population. See below for a description of how among-year survival effects are treated in this model given this structure.
2. For each of the projection years, the following steps are simulated:
    a. If specified, management actions involving penning and/or translocation (augmentation) are applied first. If females from the focal population are to be removed (i.e. translocated to a different population), or penned, these are chosen at random from the age classes of the pool of "wild" females available at the beginning of the cycle:
        i. Females to undergo protective penning are modelled as a separate "penned female" pool. If there are insufficient females to meet the penning target, all of the available females are penned. Thus 2 potential pools of "wild" females tracked for the focal population: (1) unpenned females, and (2) penned females.
        ii. Translocated females (if specified) are handled next. Translocated females are either added from an external "wild" population or from a captive breeding source. If specified in a management scenario, a portion of the translocated females may also be placed in maternity pens. This results in 4 potential pools of translocated females:
            - translocated from an external source, unpenned;
            - translocated from an external source, penned;
            - translocated from a captive breeding source, unpenned; and
            - translocated from a captive breeding source, penned;
    b. Reproduction is calculated for wild and penned females based on the input population parameters and the age-specific parturition rates. If a penning action is specific, the resulting calves are tracked separately as two calf pools ("wild calves" and "penned calves") until the end of their first model year. Otherwise, calves are treated as a single pool.
    c. Annual realized mortality rates are then calculated for both the "wild" and "penned" pools of females and calves respectively, using the differential survival rates for each class and length of penning period specified in the population parameters. Realized rates are the result of a stochastic draw from a distribution[7], and any adjustments required by management actions.
    d. Annual survival of each management pool ("wild"/"penned"), age class and sex is applied at the end of the model year. Here, if differential survival of translocated females in their second year is modelled, the proportion of the wild pool that consists of surviving translocated females

[7] Stochastic survival and recruitment rates are drawn from a log-normal distribution using the mean and SD of the specific parameter value, constrained to be within the range 0-1 (recruitment: King et al. 2010; survival: Holmes 2004).

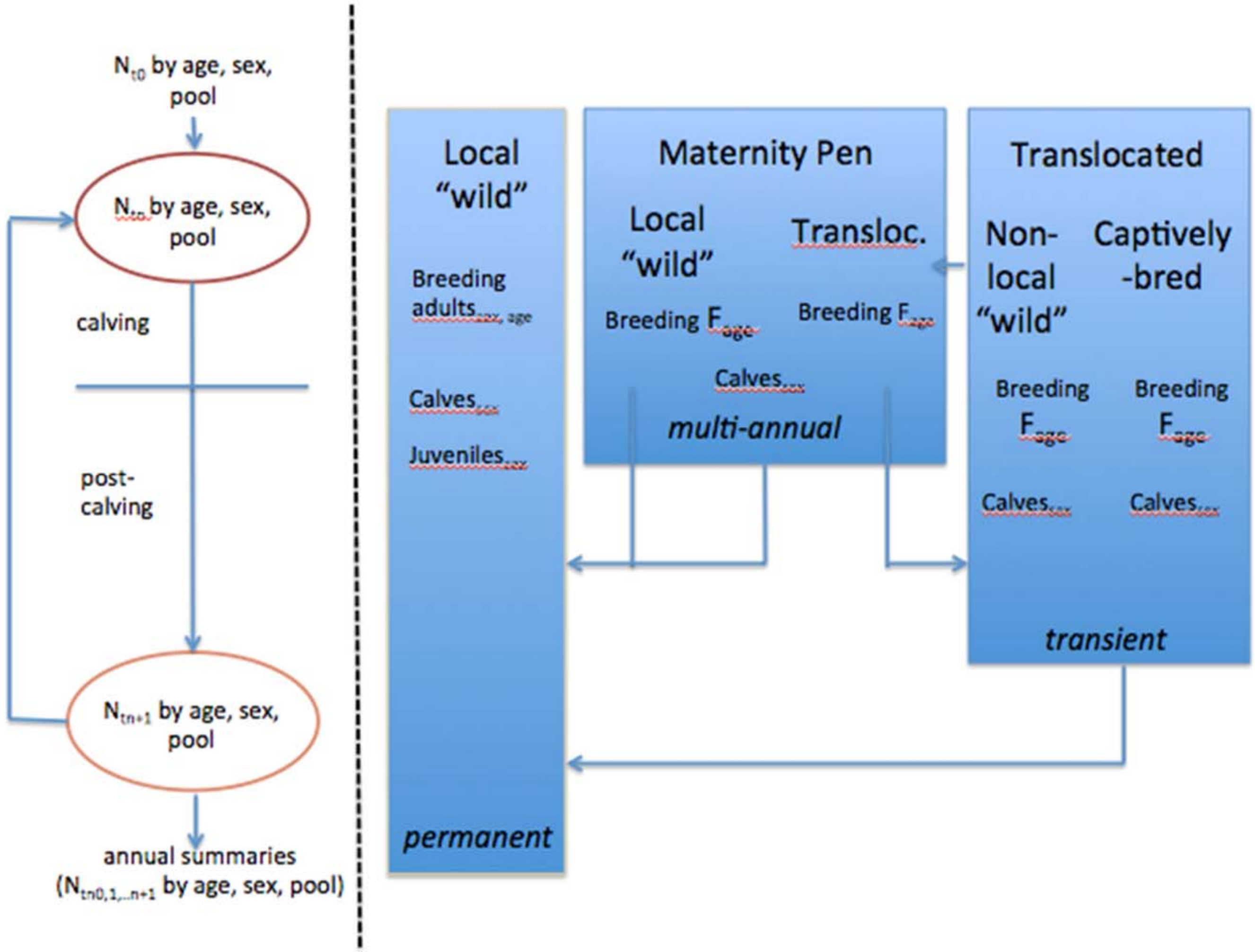

Figure A-1. Basic outlined structure of the demographic model. The flowchart on the left of the vertical dashed line gives an overview of the basic calculation structure within and among years. The three main model "pools" are schematically shown on the right of the dashed line. See text for more details on the "pools" and model calculations.

from the previous year is treated to the combined multiplier representing this differential survival rate and the realized survival rate for this pool. The realized survival rate for non-translocated females is applied to the remaining females in the pool.

e. The survivors are collected from their respective management pools ((“wild”/”penned”) back into their respective age and sex classes. Recruited calves are split into males and females.
f. Output summaries are calculated and saved in an output data structure.
g. The year counter is incremented, the age and sex classes are aged 1 year and are placed into the $\boldsymbol{N}_{t+1}$ data structure, representing the population by age and sex at the beginning of the next model year.
h. The annual model cycle repeated by looping back to step 2.a.

3. At the end of the time horizon, the output data structures are saved for later processing and analysis.

Calculation of Effects of Management on Survival Rates

A key function of the models is the calculation of the different age- and sex-effects of mortality by source of that mortality, and also by the management actions of population augmentation (via translocation) and maternal penning. We describe these calculations in more detail below.

Modelling Management of Source-Specific Mortality

As described above, the model incorporates the proportion of the overall expected mortality rate for a given age class and sex that is caused by each of a number of possible sources. These sources are: wolf predation, predation by other predator types (cougars, bears and wolverines), accidental deaths, and mortality from unknown causes. The present version of the model calculates mortality from each of these causes as follows:

1. Wolf mortality: The model assumes that management of wolf-caused mortality is primarily accomplished through a program of reducing wolf density. Accordingly, the model calculates the expected overall survival rate, and percent recruitment of calves into the population, as a function of given levels of wolf density, according to the equations in Bergerud (2006). If no management action to control densities of wolves is modeled in a scenario, then the current parameter values for calf recruitment and survival of adults are used. This simplifying assumption implies that (1) wolf density is primarily determined by factors other than the availability of caribou in these populations, and (2) wolf densities can recover quickly to conditions pre-wolf management within 1 or 2 years. (e.g., D. Hervieux, pers. comm.).
2. Other sources of mortality: Because the predicted mortality rate estimated from the Bergerud (2006) equation is calculated from an empirical meta-analysis from many populations (Bergerud 2006; Bergerud and Elliot 1998), the realized survival rate from this equation is assumed to include all sources of mortality. If management interventions that reduce the effect of these other sources of mortality are specified, they are applied as a proportional reduction to the original proportion of the overall mortality for each age and sex attributable to that source.

For example, if the unmanaged overall mortality rate of adult female caribou is 0.260 (i.e. an annual unmanaged survival rate of 0.740), and 0.85 of that mortality is due to wolf predation during the year, then the overall annual mortality rate for adult female caribou can be partitioned into two parts: 0.221 due to wolves, and 0.039 from all other causes (residual). If 0.4 of this residual mortality is due to bear and wolverine predation, then an annual management effort that reduces bear and wolverine mortality by 0.60 of its original level will result in a reduction of this component of the overall adult female caribou mortality rate to 0.001. Thus, the resulting overall mortality rate under this type of management action is: 0.039 - 0.001 = 0.038, resulting in an annual managed survival rate of 0.741 – a barely measureable effect.

Note that in this model, the effects of predators on caribou are not modelled using functional or numeric response functions (including interacting multi-species functions; see Serrouya et al. 2015). Rather, the effect of each component of the overall mortality rates on caribou (including wolves and other predator species) is specified as an overall average parameter value held to a constant mean value among years, except when management is applied as above.

Modelling Management of Reproduction (Penning) and Population Augmentation

As described above under *Annual Cycle and Order of Events*, the effects of the management options of maternity penning of local females, and augmentation of the population with females from other populations (i.e. from other wild populations or from captive breeding sources) can also be explored with this model, as follows:

Maternity Penning
A number of local females (i.e. females from the resident population) can be specified as being penned each year using multi-annual maternity pens. These females are assumed to all be pregnant at the time of penning. Females in maternity pens, or ‘predator exclosures’, are removed from the wild population (i.e. selected at random from the available females in the adult age classes) for a specified portion of the year (the penning period) as specified in the population parameters file. Differential mortality rates for penned females and their resulting calves are applied according to the population parameters (see below), and these differential rates are applied for the respective proportion of the year represented by the penning period for females and for calves born in pens. For the remaining proportion of the year, penned females and their calves are subject to the same mortality rate as wild (unpenned) females and calves.

Augmentation via translocation from “wild” or captively bred sources
Adding females from an external population source can augment the focal population. These sources can be a different, wild herd, or a captive bred population. A portion of these females may also be penned as an additional management action. Translocated females are assumed to be of unknown age (and therefore are assigned to an age at random), and are assumed to all be pregnant at the time of penning. Following their addition to the penned pool of females, the assumptions follow those described above for penned females, except for post-translocation survival depression (described below). At the end of each model year, after differential survival has been calculated but before outputs are saved, surviving penned and translocated females are returned back into the wild population pool.

Caribou translocated from other areas, including captivity, may have lower survival rates than local females until they learn how to avoid predators and where to find high quality habitat (Whittington 2015). This effect, termed post-translocation survival depression (PTSD) has been documented in other B.C caribou translocations (South Mountain caribou: Cichowski et al. 2014), and in other ungulate species (e.g., elk: Friar et al. 2007). In this model, PTSD effects (if any are assumed) are applied only in the year in which translocated females are tracked as a separate pool. Any residual PTSD effects occurring in subsequent years are not considered.

Parameters in the Model

Population Parameters

To project annual estimates of population size by age in the model, we used four sets of parameters: (1) initial population size $\boldsymbol{N_0}$ and age distribution; (2) parameters related to annual survival by age and sex in a wild population; (3) parameters related to specific management effects on survival and age; and (4) parturition rates by age. Each of these sets is described in more detail below.

Initial population size ($\boldsymbol{N_0}$) and age distribution
While initial population sizes may be known or estimated from annual population and/or composition surveys (i.e. minimum population size; an extrapolated estimate of total population size), the age-structure is not. Therefore assigning the observed number of individuals to ages 1-18 at random creates an initial age distribution. Sexes are assigned using the observed bull:cow ratio for each population. Note that the same initial age distribution is used for all replicates in this version of the model.

Survival and recruitment parameters for wild populations
For each modelled population, there are 13 parameters describing the vital rates of the population (Table A-1). Estimates for these parameters are derived from field data available for the population.

Management effects on survival and recruitment parameters
Many of the management actions influence specific demographic parameters. In the present model, there are 18 parameters specifying the effects that different management actions may have on one or more vital rates (Table A-2). Some rates are imposed constants defined as part of a scenario, while others may be estimated from recent management case studies or experiments.

Table A-1. Demographic parameters related to estimating survival and recruitment for a population.

Parameter Description	Parameter Name
Proportion of females at birth.	Prop_FemAtBirth
Current proportion of calves in the total population.	Current_Propn_Calves_InPopn
Coefficient of variation (CV) for *Current_Propn_Calves_InPopn* (a stochastic parameter representing process error).	Current_CV_Calf_Birth_Rate
Annual survival rate of wild calves with no predator management. Estimated from calf/cow ratio & average parturition rate.	Current_Annual_Calf_Surv_Wild
Coefficient of variation (CV) for *Annual_Calf_Surv_Wild* (a stochastic parameter representing process error).	Current_CV_Calves_SR_Wild
Proportion females in adult population. Derived from composition surveys of adults (e.g., bull:cow ratio).	Current_Prop_FemAdults
Annual adult female mortality rate.	Current_WildMortRate_Females
Coefficient of variation (CV) for *WildMortRate_Females* (a stochastic parameter representing process error).	Current_CV_Female_WildMortRate
Annual adult male mortality rate. Estimates of this rate are made from the bull:cow ratio to obtain the proportion of females over the age distribution.	Current_WildMortRate_AdMales
Proportion of non-wolf caused natural mortality due to cougars[1].	WildMortPropn_Cougars
Proportion of non-wolf caused natural mortality due to bears and wolverines.	WildMortPropn_BearsWolverines
Proportion of non-wolf caused natural mortality due to accidents.	WildMortPropn_Accidental
Proportion of non-wolf caused natural mortality due to unknown causes.	WildMortPropn_Unknown

[1] See text for an explanation of how these proportions are used to adjust realized mortality if management is specified.

Table A-2. Management & demographic effect parameters related to estimating survival and recruitment for a population under different management actions.

Parameter Description	Parameter Name[1]
# months that females are penned (used in SR calculations).	Months_FemalesPenned
# months that calves are penned (used in SR calculations).	Months_CalvesPenned
Survival rate of unpenned females from translocated (external wild source) pool in year after translocation[1].	Females_SR_UnpennedAug_Ext
Survival rate of unpenned females from translocated (captively bred source) pool in year after translocation.	Females_SR_UnpennedAug_CBF
Coefficient of variation (CV) for female survival rate for unpenned translocated pool (stochastic parameter).	CV_Females_SR_UnpennedAugmented
Survival rate of Penned Females while in the maternity pen.	Females_SR_WhilePenned
Coefficient of variation (CV) for female survival rate while in a maternity pen (stochastic parameter).	CV_Females_SR_Penned
Survival rate of penned females from local population pool post-penning period.	Females_SR_PostPenning_PennedFromLocal
Coefficient of variation (CV) for penned females from local population after the penning period (stochastic parameter).	CV_Females_SR_PostPenning_PennedFromLocal
Post-penning survival rate of penned females from translocated (external source) pool in the period after penning.	Females_SR_PostPenning_PennedFromAug_Ext

Post-penning survival rate of penned females from translocated (captively-bred source) pool.in the period after pennins.	Females_SR_PostPenning_PennedFromAug_CBT
Coefficient of variation (CV) for post-penning female survival rate for penned from translocated - either type of source (stochastic parameter).	CV_Females_SR_PostPenning_PennedFromAug
Post-translocation survival depression[2] (proportional reduction): applied to translocated females from an external (wild population) source.	PTSD_ExtF
Post-translocation survival depression (proportional reduction): applied to translocated animals from a captively bred source.	PTSD_CBF
Calf survival while in the maternity pen.	Calf_SR_WhilePenned
Coefficient of variation (CV) for *Calf_SR_WhilePenned* (stochastic parameter).	CV_Calf_SR_WhilePenned
Calf survival post-penning – up to 12 months of age.	Calf_SR_PostPenning
Coefficient of variation for captive (penned) calf survival post-penning to 12 months (stochastic parameter).	CV_Calf_SR_PostPenning

[1] These names are referred to in Table B-2 and Table B-3 (Appendix B).
[2] Accounts for translocation stress, and unfamiliarity with the location of high quality habitat. This parameter may or may not account for naiveté with the local predator community.

The model uses age-specific parturition rates for female caribou based on a study of Northern Caribou by McNay and Giguere (in prep.). The rates used are given in Table A-3:

Table A-3. Parturition rates for female caribou used in the model and applied to all populations.

Age (yrs.)	Parturition Rate
1	0.0
2	0.48
3-8	0.8
9+	0.625

Management Scenarios and Effects on Parameters

As described in the main report, the model currently implements three general types of management interventions:

1. Predator and other mortality management activities designed to reduce the predation rate on calves and female caribou. The types of predator management activities that can be specified are:
    - changes in wolf predation rate (implemented as an annual target mean density of wolves in the population's range); and
    - changes in other sources of mortality (cougars, bears and wolverines, accidents and unknown), implemented as proportional reductions in these effects on overall survival rates applied to each pool.

   See the Modelling Management of Source-Specific Mortality section above for details on how these predator/other mortality management effects are implemented.
2. number of pregnant females translocated from an external source that are added to the population each year (i.e. augmentation). Translocated females are assumed to be from one or both of two sources:
    - an external wild population; and
    - a captively bred source (non-specific).
3. number of females penned per year. Different target values for implementing the numbers of females from each pool that can be penned are specified in scenarios:
    - percent of the local population of females that can be penned; and
    - percent of translocated females that can be penned per year.

Each of these three management actions can be implemented annually (i.e. are explicitly modeled year by year).

There is an additional management action that is possible in this model – that is, females from the local population can be removed (e.g., if the focal population was a source of translocated females for another population), although this option was never applied in this study.

Limitations of the Model

1. The model internally uses floating point (real) values for population sizes, not integer values. Annual results use rounded values. This may cause a slight over-estimate of projected population sizes, especially at low population sizes.
2. This version of the model does not ensure that there are breeding bulls in the population prior to reproduction. Therefore it is possible at very small population sizes to model reproduction occurring in the absence of breeding bulls, which would evidently produce non-representative results.
3. Although very large maternity pens can potentially be specified in this model, these are not equivalent to true predator exclosures, as all females from maternity pens are released back into the wild at the end of each maternity penning period, in late summer. Modelling of predator exclosures would require the addition of a separate set of pools into the model.

REFERENCES

Bergerud, A.T. 2006. The need for the management of wolves – an open letter. Rangifer, 17:39-50.

Bergerud, A.T., and J.P. Elliot. 1998. Wolf predation in a multiple-ungulate system in northern British Columbia. Canadian Journal of Zoology, 76: 1551-1569.

Cichowski, D., G. Sutherland, and S. McNay. 2014. Purcells South Mountain Caribou Herd Augmentation Viability Assessment. Unpublished report submitted to BC Ministry of Forests, Lands, and Natural Resource Operations, Cranbrook, B.C.

Frair, J. L., E. H. Merrill, J. R. Allen, and M. S. Boyce. 2007. Know thy enemy: experience affects elk translocation success in risky landscapes. Journal of Wildlife Management 71:541-554.

Holmes, E. E. 2004. Beyond theory to application and evaluation: diffusion approximations for population viability analysis. Ecological Applications, 14: 1272-1293.

King, R, B.J.T. Morgan, O. Gimenez, and S.P. Brooks. 2010. Bayesian analysis for population ecology. Chapman and Hall/CRC, Boca Raton, Fl, USA.

Serrouya, R., M.J. Wittmann, B.N. McLellan, H.U. Wittmer, and S. Boutin. 2015. Using predator-prey theory to predict outcomes of broadscale experiments to reduce apparent competition. American Naturalist, 185: 665-679.

Whittington, J., 2015. Caribou captive breeding release scenarios and population projections. Banff National Park Resource Conservation. Banff, Alberta. 16 pages.

APPENDIX B. POPULATION AND MANAGEMENT-SPECIFIC PARAMETER VALUES USED IN PROJECTIONS

In this Appendix, we provide all the parameter values used to model the population projections explored in this study.

Population-specific Parameters

As described in Appendix A, the model is initialized with an initial size ($\boldsymbol{N_0}$) and age class distribution. The number and modeled distributions of age-classified results for the populations projected in this study are shown in Table B-1.

We calculated the following population-specific demographic parameters either from empirical data for each population provided for this study (see Table 1: Main report), of using data and methods described in footnotes to Table A-1.

Finally, we assumed the following management-specific parameters (defined in Appendix A) as shown in Table B-3. These parameters were 'general' and therefore applied to each population during simulations.

Table B-1. Initial estimated population age distribution by sex.
Age distributions are estimated using the overall assumed bull: cow ratio, and a random initial assignment of the animals to each age class. This age distribution was applied to all modelled herds to make them comparable under this assumption.

Pop:	Calendar		Chinchaga		Maxhamish		Parker		Prophet		Snake-Saht		W-S Ft Nel.	
Age	# F	# M	# F	# M	# F	# M	# F	# M	# F	# M	# F	# M	# F	# M
1	22	22	7	7	19	19	1	0	1	1	20	20	2	3
2	16	15	32	30	33	32	4	3	5	4	31	26	7	6
3	16	11	26	17	27	21	3	2	4	3	27	20	6	4
4	15	9	21	10	22	13	3	1	4	2	23	15	6	2
5	14	7	17	6	17	9	2	1	4	1	20	11	6	2
6	13	5	14	3	13	6	2	0	4	1	18	8	5	1
7	12	4	11	2	10	4	1	0	3	1	15	6	4	1
8	11	3	9	1	8	3	1	0	3	1	13	4	4	1
9	10	2	7	0	7	2	1	0	3	1	11	3	3	0
10	9	1	6	0	5	2	0	0	2	0	10	2	3	0
11	8	0	5	0	4	2	0	0	1	0	9	2	3	0
12	8	0	4	0	4	1	0	0	1	0	8	1	3	0
13	8	0	4	0	3	0	0	0	0	0	7	0	2	0
14	8	0	3	0	3	0	0	0	0	0	6	0	1	0
15	7	0	3	0	2	0	0	0	0	0	5	0	1	0
16	6	0	2	0	2	0	0	0	0	0	5	0	1	0
17	6	0	1	0	2	0	0	0	0	0	4	0	0	0
18	6	0	1	0	2	0	0	0	0	0	4	0	0	0

Table B-2. Estimated model parameter values related to estimating survival and recruitment for all ranges.

Parameter Name[1]	Population[2]						
	Calendar	Chinch	Maxham	Parker	Prophet	Snake-Saht	W-S Ft. Nelson
Prop_FemAtBirth	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Current_Propn_Calves_InPopn	0.152	0.06	0.13	0.04	0.037	0.111	0.064
Current_CV_Calf_Birth_Rate[2]	0.216	0.216	0.216	0.216	0.216	0.216	0.216
Current_Annual_Calf_Surv_Wild	0.28	0.1	0.27	0.1	0.06	0.23	0.09
Current_CV_Calves_SR_Wild[2]	0.283	0.283	0.283	0.283	0.283	0.283	0.283
Current_Prop_FemAdults	0.75	0.77	0.62	0.76	0.77	0.57	0.76
Current_WildMortRate_Females	0.04	0.19	0.19	0.13	0.13	0.13	0.13
Current_CV_Female_WildMortRate[2]	0.126	0.126	0.126	0.126	0.126	0.126	0.126
Current_WildMortRate_AdMales	0.3	0.575	0.34	0.35	0.36	0.25	0.34
WildMortPropn_Cougars	-	-	-	-	-	-	-
WildMortPropn_BearsWolverines	0.04	0.04	0.04	0.04	0.04	0.04	0.04
WildMortPropn_Accidental	0.01	0.01	0.01	0.01	0.01	0.01	0.01
WildMortPropn_Unknown	0.2	0.2	0.2	0.2	0.2	0.2	0.2

[1] See Table A-1 (Appendix A) for definitions of these parameters.
[2] Sources for most recent demographic parameter estimates were Table 1 (main report) calculated as described in the Parameter Estimates section.
[3] Annual inventory data from 2008-2015 from the nearby Klinse-Za population of caribou was used to estimate these CV values.

Table B-3. Estimated model parameter values related to estimating survival and recruitment under different management actions. These values are applied to all populations if a scenario invokes the specific management action.

Parameter Name[1]	Parameter Estimate(s)[2]
Months_FemalesPenned	4
Months_CalvesPenned	1.5
Females_SR_UnpennedAug_Ext	0.713/0.9
Females_SR_UnpennedAug_CBF	0.713/0.9
CV_Females_SR_UnpennedAugmented	0
Females_SR_WhilePenned	1
CV_Females_SR_Penned	0
Females_SR_PostPenning_PennedFromLocal	0.9741
CV_Females_SR_PostPenning_PennedFromLocal	0.11
Females_SR_PostPenning_PennedFromAug_Ext	0.713/0.9
Females_SR_PostPenning_PennedFromAug_CBT	0.713/0.9
CV_Females_SR_PostPenning_PennedFromAug	0.11
PTSD_ExtF	0.7
PTSD_CBF	0.7
Calf_SR_WhilePenned	0.95
CV_Calf_SR_WhilePenned	0
Calf_SR_PostPenning	0.79

[1] See Table A-2 (Appendix A) for definitions of these parameters.
[2] Multiple values indicate higher or lower values of post-translocation survival depression (PTSD) modeled as alternate assumptions.

APPENDIX C. DESCRIPTION OF MODELLED BEHAVIOUR OF KEY DEMOGRAPHIC PARAMETERS

In this Appendix, we describe some of the behaviour and interactions between the key demographic parameters and outcome indicators in terms of current conditions and in relation to two types of management actions (no management, management to a wolf density target). The demographic parameters illustrated are:

1. Annual projected calf recruitment, indicated by calf:cow ratio to facilitate comparisons with empirical data;
2. Annual projected weighted mean[8] calf survival rate; and
3. Annual projected weighted mean adult female survival rate.

Primarily, we are interested in verification of the basic model behaviour, through comparisons of the demographic outputs. Note that in this draft Appendix, only two management actions are explored to give an example of their effects on demography.

Parameter behaviour under current conditions (no management)

As described elsewhere (Appendix A), the initial demographic state of each modelled population reflects the most recent demographic data available (e.g., 2014/2015 estimates of adult female survival, calf:cow ratios, bull: cow ratio and population size). See Table 1 (main report) for these estimates, which were used to initialize the population age distribution. For projections, the parameters calf recruitment, annual calf survival, and adult female survival, are coupled with the estimated CVs about each parameter to give rise to a distribution of "realized" values for these parameters across replicates, leading to the distribution of projected growth rates. Under a "no management" assumption, the projected future behaviour of the population in response to these realized parameter values is expected to include recent past behaviour within the distribution of potential outcomes.

We found that the modelled distributions of realized adult female survival rate, and calf:cow ratio appear to align with their respective point estimates as given by recent empirical data (Figure C-1, Figure C-2). In particular, the current empirical estimates for both parameters fall well within the 1.5 interquartile distance from the median projected value for all populations. This suggests that the current behaviour of the populations is captured within the distribution of projected outcomes from the model, at least to the extent represented by these parameters.

We do not have current estimates of the calf survival rates for the modelled populations to compare with the population-specific projected distributions of this rate (Figure C-3). However, the range of median values across all populations (0.05 to 0.32) is broadly consistent with values reported elsewhere.

In addition, the behaviour of the annual female survival rate in relation to predation level (as indicated by the variable levels of target wolf density that we modelled) shows an

[8] These are the geometric means of the individual pools of each age and sex during each portion of the year (e.g. penning period; post-penning; the whole year), weighted by the number of individuals in each pool that are alive at the midpoint of the respective period.

overall decline as wolf density increases (Figure C-4). Although aggregated across many factor levels and sources of uncertainty, this result is consistent with expectations.

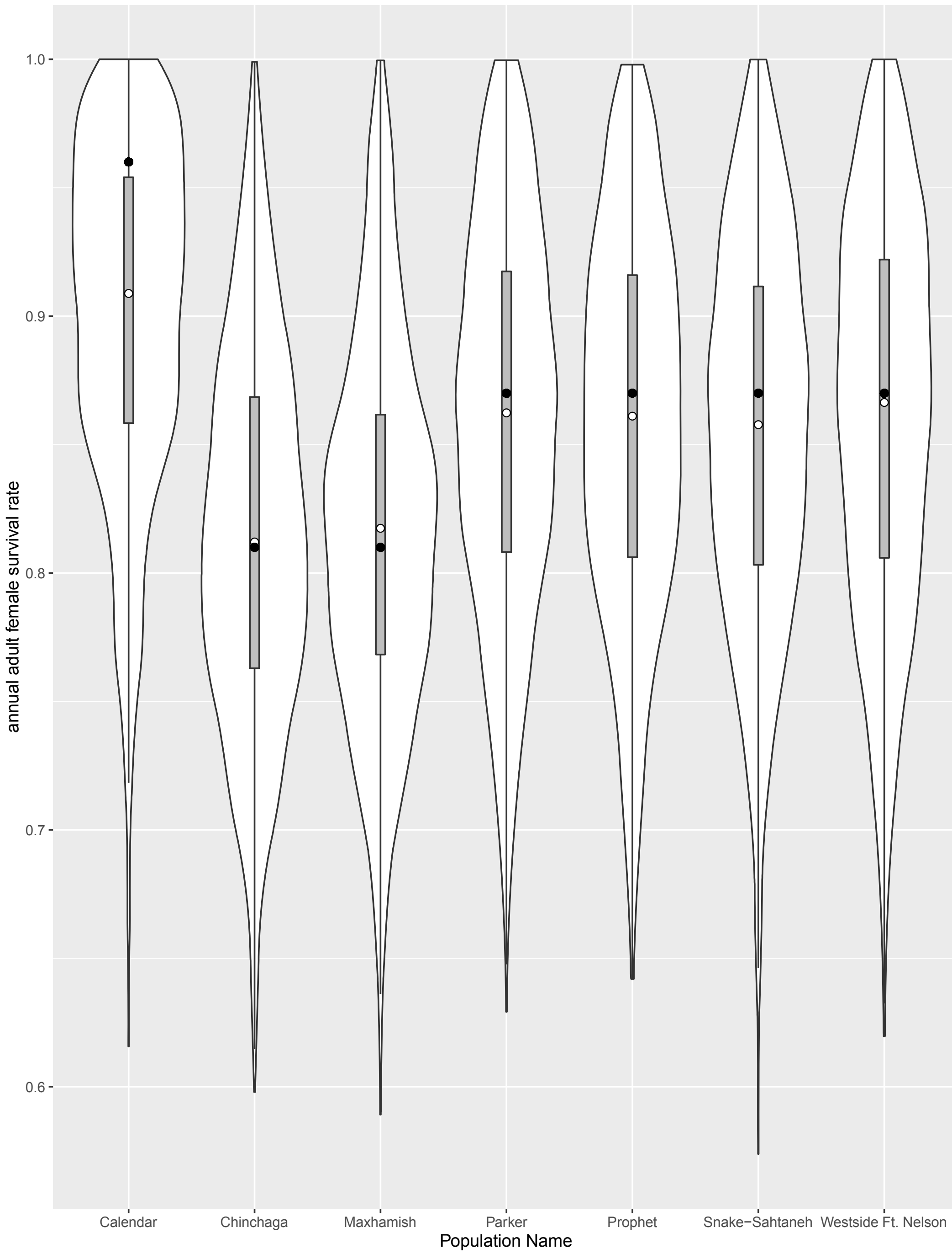

Figure C-1. Kernel density estimates of projected weighted mean adult female survival rate for each modelled population under the “no management” scenario.
Interior box plots, calculated from the replicated data, indicate median (white dot), 25% and 75% quartiles of the distribution (gray bar), and 1.5 times the interquartile range (vertical black lines). The current empirical estimate for the adult female survival rate is shown as a black dot.

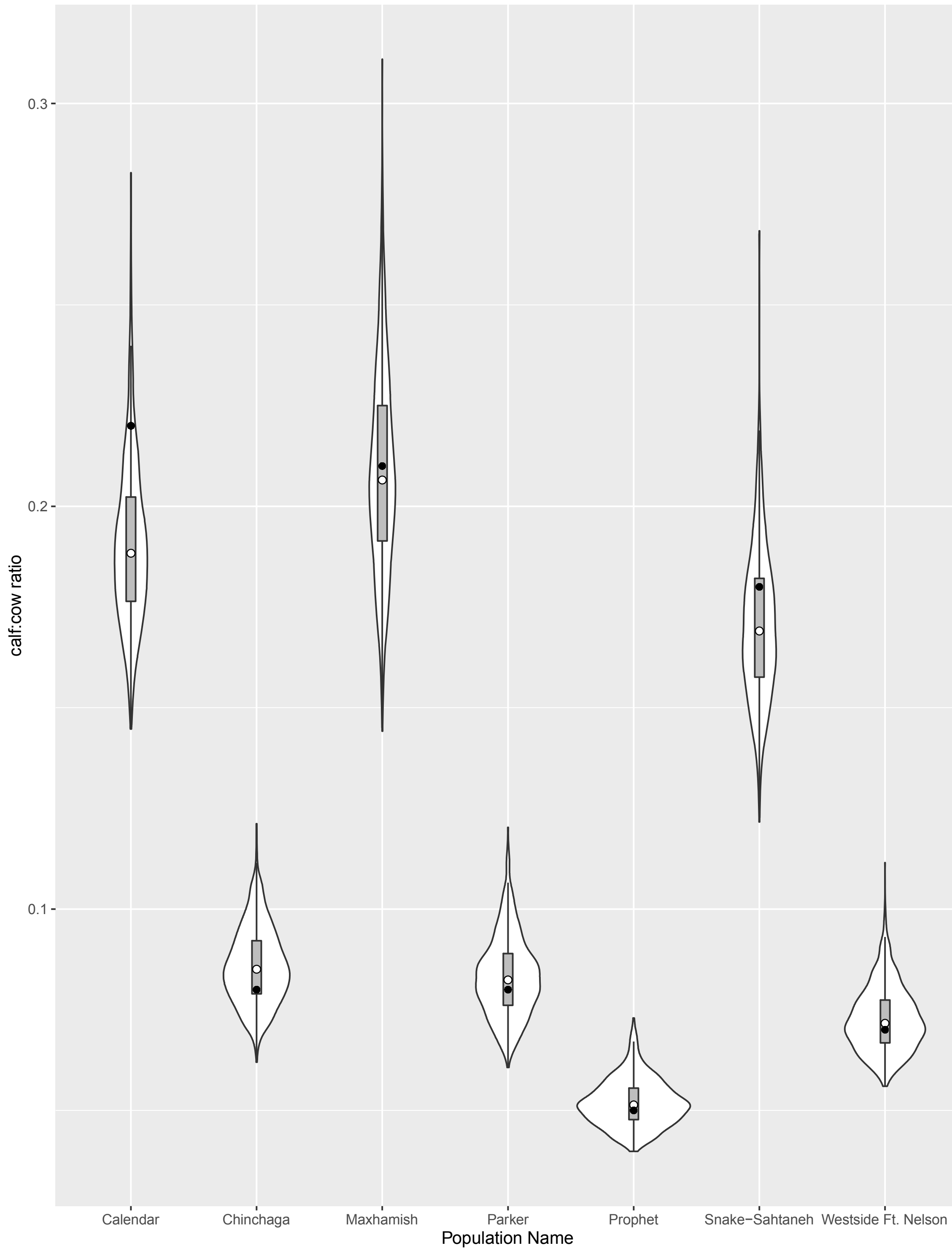

Figure C-2. Kernel density estimates of projected calf:cow ratios for each modelled population under the “no management” scenario.
See Figure C-1 caption for explanation of symbols.

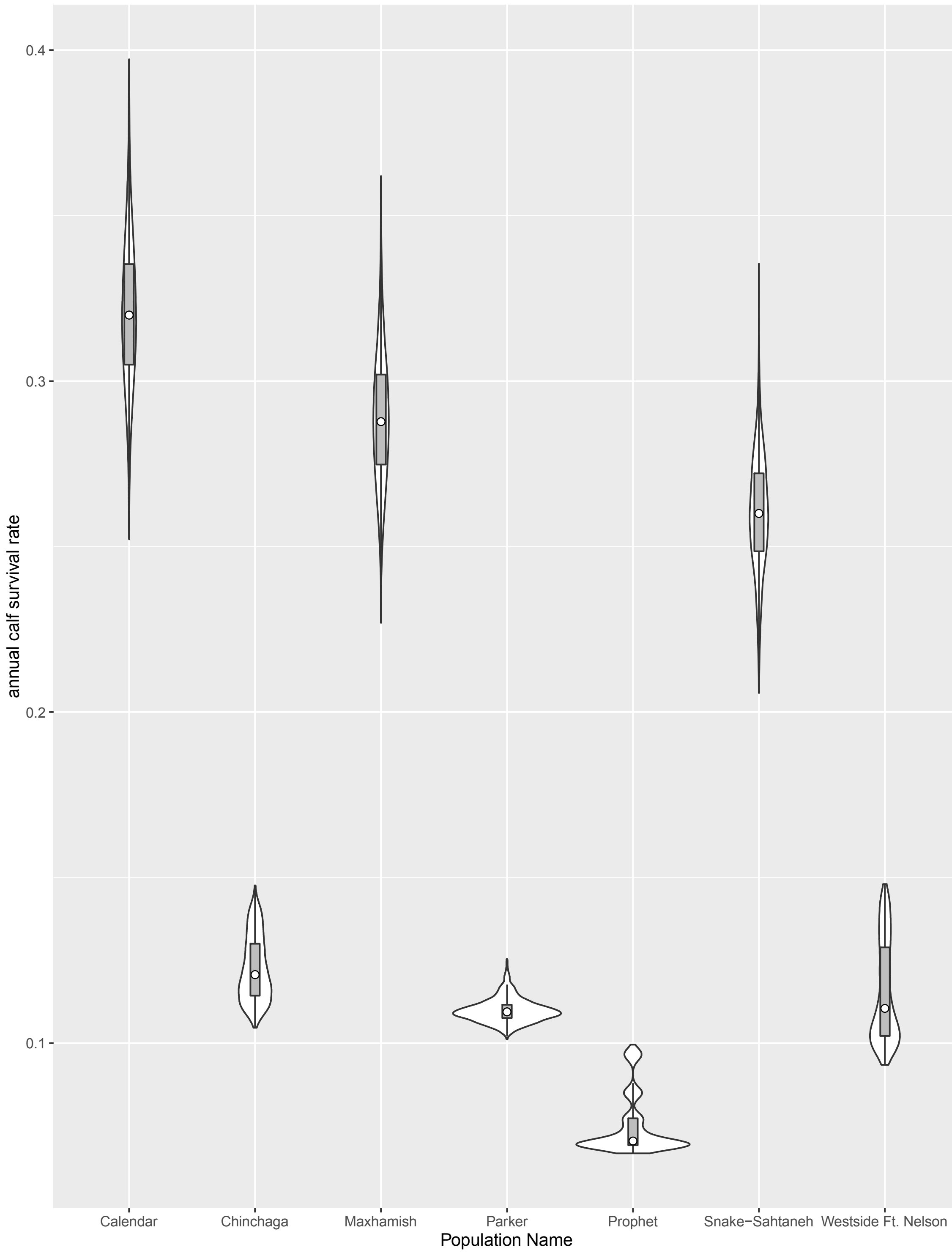

Figure C-3. Kernel density estimates of projected annual calf survival rate for each modelled population under the "no management" scenario.
See Figure C-1 caption for explanation of symbols. Note that no population-specific empirical data was available for comparison with this estimate.

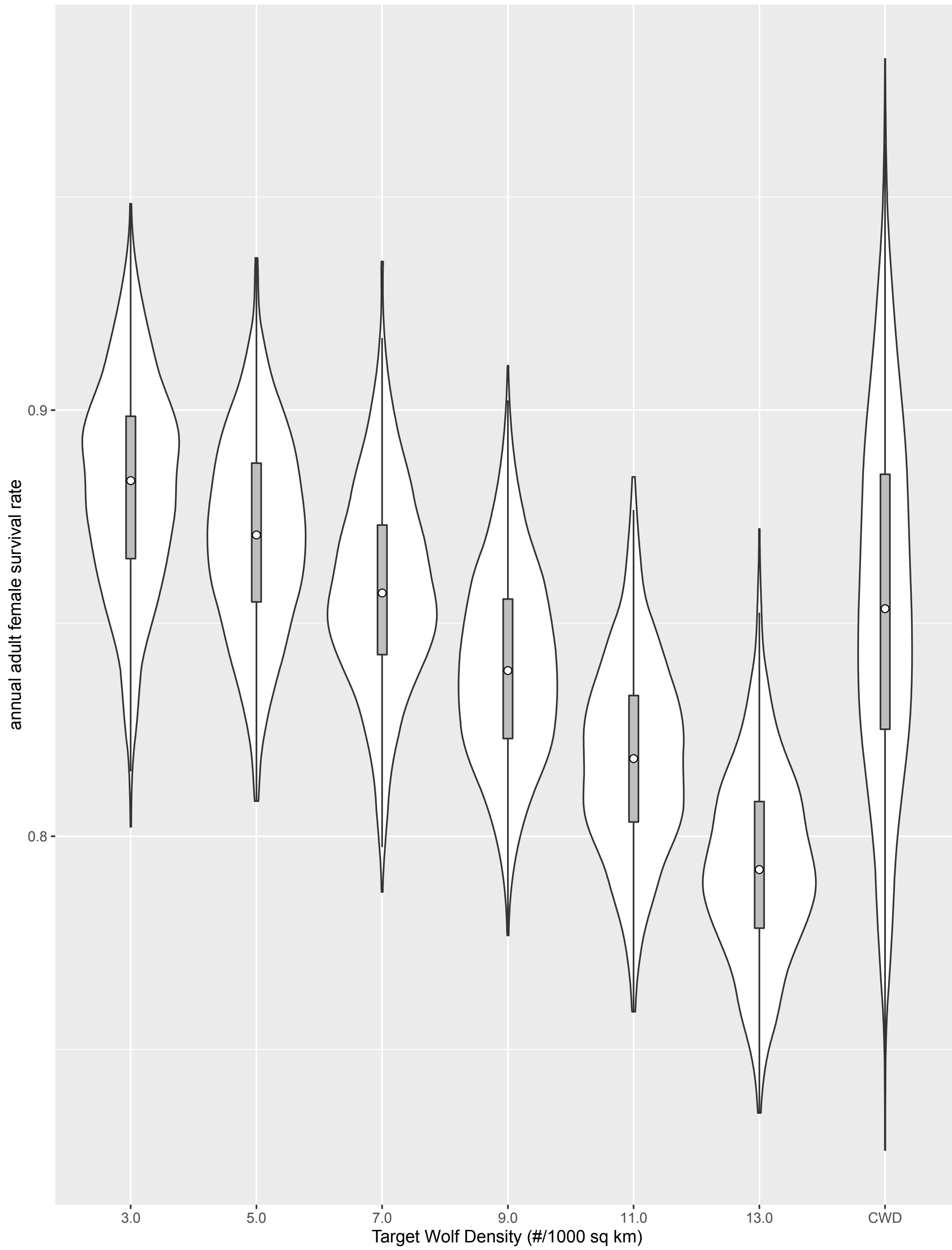

Figure C-4. Kernel density estimates of the median projected annual adult female survival rate projected over different modelled densities of wolves. No other management action (i.e. maternity penning/exclosures, translocations, etc.) is considered here.
See Figure C-1 caption for explanation of symbols. The symbol “CWD” indicates “estimated current wolf density”. The data here summarize results across all populations.