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Triage for conserving populations of threatened species: The case of woodland caribou in Alberta

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ABSTRACT

Prioritization of conservation efforts for threatened and endangered species has tended to focus on factors measuring the risk of extirpation rather than the probability of success and cost. Approaches such as triage are advisable when three main conditions are present: insufficient capacity exists to adequately treat all patients, patients are in a critical state and cannot wait until additional capacity becomes available, and patients differ in their likely outcome and/or the amount of treatment they require. The objective of our study was to document the status of woodland caribou (*Rangifer tarandus*) herds in Alberta, Canada, with respect to these three conditions and to determine whether a triage approach might be warranted. To do this we modeled three types of recovery effort – protection, habitat restoration, and wolf control – and estimated the opportunity cost of recovery for each herd. We also assessed herds with respect to a suite of factors linked to long-term viability. We found that all but three herds will decline to critical levels (<10 animals) within approximately 30 years if current population trends continue. The opportunity cost of protecting all ranges by excluding new development, in terms of the net present value of petroleum and forestry resources, was estimated to be in excess of 100 billion dollars (assuming no substitution of activity outside of the ranges). A habitat restoration program applied to all ranges would cost several hundred million dollars, and a provincial-scale wolf control program would cost tens of millions of dollars. Recovery costs among herds varied by an order of magnitude. Herds also varied substantially in terms of their potential viability. These findings suggest that woodland caribou in Alberta meet the conditions whereby triage should be considered as an appropriate conservation strategy.

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1. Introduction

It has long been recognized that the resources available for conserving the world's biodiversity are grossly inadequate for the task. As a result, strategic approaches have been developed for using the available resources as efficiently as possible. For the most part, these efforts have focused on defining the "best" course of action within an ecological context. Much of the literature on the design and selection of reserve systems using systematic conservation planning falls into this category (e.g., Margules and Pressey, 2000).

More recently, there has been a growing interest in taking into account such things as probability of success and conservation costs when setting conservation priorities and allocating effort (Naidoo et al., 2006; Bottrill et al., 2008; Joseph et al., 2009). There is growing recognition that tradeoffs between economic and conservation objectives are inevitable and that a failure to explicitly consider these tradeoffs can result in the inefficient allocation of conservation resources and unsuccessful outcomes at the point of implementation (Murdoch et al., 2007; Bottrill et al., 2008; Joseph et al., 2009).

Various terms have been used to describe resource allocation approaches that take cost and likelihood of success into account. These include triage (Bottrill et al., 2008), return on investment (Murdoch et al., 2007), conservation efficiency (Wilson et al., 2007), and optimal resource allocation (Joseph et al., 2009). Most of these efforts have focused on the maintenance of regional biodiversity (Rondinini, 2008). In this paper, we explore the application of these concepts to a threatened animal population, using woodland caribou in Alberta, Canada, as an example. We use the triage approach as our conceptual framework because it captures both the concept of optimal resource allocation, particularly cost-effectiveness analysis, and the context of urgency that typifies decision making related to the management of threatened species.

Woodland caribou is listed as a threatened species, both provincially and federally, reflecting declines in population size, contraction of range, and sensitivity to human activities (Alberta

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Woodland Caribou Recovery Team, 2005). At the national scale, Alberta's woodland caribou herds are among the most at risk in Canada (Environment Canada, 2008). The ranges of some herds have experienced the effects of substantial energy sector development and forest harvesting, others are perched on significant oilsands deposits which are about to undergo massive development, and still others, are relatively undisturbed. The provincial caribou recovery plan seeks to maintain the current distribution of all caribou herds in the province despite the economic pressures to develop the oilsands reserves and other resources (Alberta Woodland Caribou Recovery Team, 2005).

There are approximately 3000 caribou in the province, split into 12 main herds (i.e., populations; Table 1). All monitored herds in Alberta have experienced low recruitment and adult female survival in recent years leading to negative population growth (Table 1; McLoughlin et al., 2003). Wolf (*Canis lupus*) predation has been identified as the primary proximate factor in the decline of caribou in Alberta (Stuart-Smith et al., 1997; McLoughlin et al., 2003; Latham 2009) and other jurisdictions as well (Boertje et al., 1996; Hayes et al., 2003; Wittmer et al., 2005). Sport hunting of caribou is not permitted in Alberta, but a low rate of poaching and permitted First Nations harvest may occur.

Caribou have persisted in Alberta for millennia despite the presence of wolves, which suggests that the current unsustainable rate of wolf predation is not the norm and that caribou–wolf dynamics have been modified in some fundamental way over the past few decades. The hypothesis with the greatest support is that increased industrial development, primarily related to petroleum extraction and forest harvesting, has transformed caribou range and surrounding habitat leading to: (1) increased densities of wolves as a result of the increase in density of their primary prey, moose (*Alces alces*) and white-tailed deer (*Odocoileus virginianus*), (2) an increased rate of encounter between wolves and caribou, and (3) increased wolf hunting efficiency (Dyer et al., 2001; Schaefer, 2003; James et al., 2004; Smith, 2004; Sorensen et al., 2008; Latham, 2009). In light of these findings, the reduction and restoration of industrial features and the reduction of wolf predation have become major themes of Alberta's caribou recovery program (Alberta Woodland Caribou Recovery Team, 2005).

Although Alberta's caribou recovery program is mandated to recover all herds in the province, efforts have focused on herds perceived to have a high risk of immediate extirpation (Alberta Woodland Caribou Recovery Team, 2005). But if conservation capacity, including consideration of the opportunity costs of conservation, is limited, efforts allocated to caribou herds with little chance of recovery may deprive more viable herds of the support they need to remain viable. In this situation a triage approach may provide a better overall conservation outcome (Naidoo et al., 2006; Bottrill et al., 2008). Triage in a conservation context involves prioritizing the allocation of limited resources to maximize overall conservation returns (Bottrill et al., 2008). Allocation decisions may involve choices about which management actions to use or about where (or to whom) these actions will be applied, or some combination of both. Here we focus on differentiating potential recipients, similar to the original battlefield antecedents of the triage concept.

The implementation of a triage system is advisable when three main conditions are present. First, insufficient capacity exists to adequately treat all patients (in our case, herds). Second, patients are in a critical state and cannot wait until additional capacity becomes available (i.e., sequential treatment of patients over an extended period of time is not a viable option). Third, patients differ in their likely outcome and/or the amount of treatment they require. The objective of our study is to document the status of caribou herds in Alberta with respect to these three conditions and to determine whether conservation triage might be warranted. The conservation objective that triage is meant to optimize in our case is the long-term viability of woodland caribou at the provincial level.

We also develop a framework for triage-based decision making as a practical application of our findings. The framework is qualitative in nature because the data required for a quantitative population viability analysis linked to individual management actions are lacking in key aspects. Thus we do not perform an optimization analysis with a specific conservation objective and budget (in the strictest sense of conservation triage, Bottrill et al., 2008). Rather, we rank herds in terms of relative risk of extirpation, relative likelihood of conservation success, and opportunity costs of protection and consider how prioritization of herds for conservation changes depending on how some or all of these are taken into consideration. As part of our framework we examine conservation capacity by quantifying management costs and the net present value of new industrial activity (as opportunity costs of conservation activities). Our rankings provide insights into the implications of various conservation strategies (priority and/or number of herds "treated") but we do not have information on benefits and thus cannot conduct a full benefit cost analysis. Also, our analysis was conducted at the caribou range scale and thus our rankings must be interpreted at this scale.

Table 1
Population data for caribou herds in Alberta.

Herd	Code	Range size (km^2)	Number of caribou[a]	Population growth rate[b]
Cold Lake	CL	2679	125	0.82
Richardson	RICH	2689	100	0.96
Little Smoky	LS	2927	80	0.93
Slave Lake	SL	3412	75	0.87
Redrock – Prairie Creek	RRPC	4714	350	0.90
A La Peche	ALP	6254	150	0.96
Bistcho	BIS	12,795	300	0.89
Athabasca River East	ESAR	14,691	175	0.87
Athabasca River West	WSAR	15,010	350	0.95
Red Earth	RE	15,977	300	0.87
Chinchaga	CHIN	17,101	275	0.92
Caribou Mountains	CM	18,642	450	0.88

[a] From Environment Canada, 2008 and West-Central Alberta Caribou Landscape Planning Team, 2008.
[b] Geometric mean of annual population growth rate from 2003 to 2008 based on survival and recruitment data collected by the ACC, except for RICH and BIS which were based on predicted values from the regression equation described in Appendix A.

2. Methods

Polygons defining the range of each caribou herd were obtained from the Alberta Department of Sustainable Resource Development (Fig. 1). Caribou population data were obtained from the Alberta Caribou Committee (ACC) which has collected 114 herd-years of adult female survival and calf recruitment data covering most herds in the province (see McLoughlin et al., 2003 and Sorensen et al., 2008 for a description of monitoring methods).

We compared the relative risk of extirpation among herds on the basis of current estimated population size (Table 1) and current rate of population growth (λ). For this calculation λ was fixed at its mean over the last 5 years (Table 1) and we determined the number of years it would take for herds to decline to less than 10 animals (as a crude measure of relative risk of extirpation). The mean population growth rate was calculated using population data from the AAC and methodology described in Sorensen et al. (2008).

A combination of three recovery actions – habitat restoration, habitat protection, and wolf control – were modeled to explore dif-

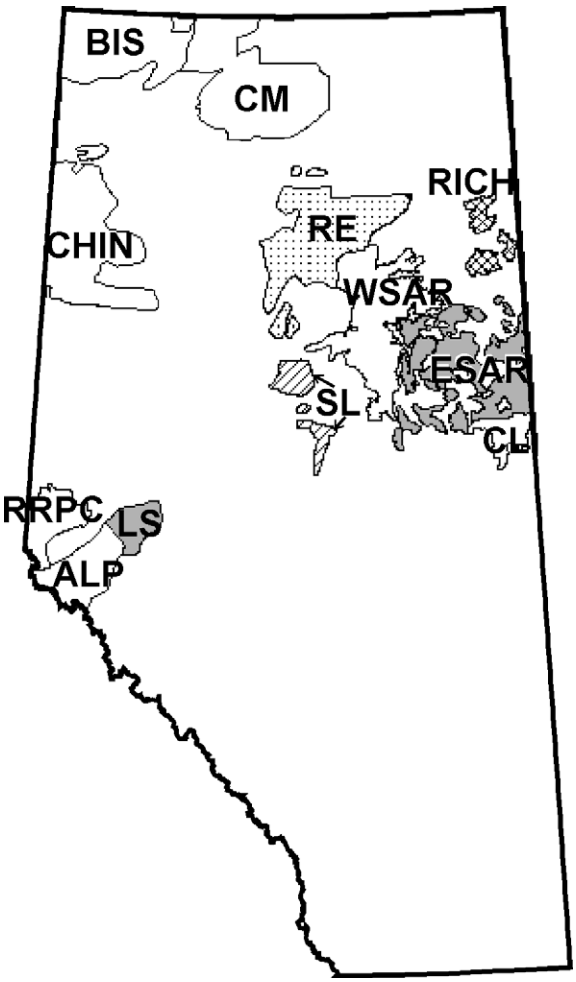

Fig. 1. Location of caribou ranges in Alberta. Some ranges are shaded to clarify boundaries.

ferences among herds with respect to the cost and potential response to recovery efforts. We did not model responses for individual recovery actions because these actions cannot be separated in a realistic management context. For example, habitat restoration efforts would be of limited value in the absence of habitat protection because ongoing industrial activity creates new linear features (which must be the case if resources are to be extracted). Similarly, habitat protection would be ineffective unless current linear features are restored and wolf control is implemented because natural recovery would be too slow for many caribou herds to persist. Finally, public support for wolf control would be unlikely in the absence of habitat-related efforts intended to achieve herds that are self-sustaining.

2.1. Habitat restoration

In our model, restoration was restricted to seismic lines (forest cut-lines, 2–8 m in width). Seismic lines, used in the search and delineation of petroleum resources, are the most common industrial feature on the landscape and the most amenable to restoration because they are generally not needed after their initial use. The density of seismic lines and other linear features within each caribou range was derived from the provincial Base Features Database, updated to 2006 (Alberta Sustainable Resource Development, unpublished data; Table 2).

Seismic lines were divided into two groups: those that require restoration and those that do not because natural regeneration is

Table 2
Linear disturbance density and percentage of young forest, by herd.

Herd	Seismic (km/km^2)	Roads (km/km^2)	Pipelines (km/km^2)	Young Forest[a]
ALP	0.51	0.09	0.02	2.6
RRPC	0.63	0.15	0.06	5.8
CM	0.86	0.00	0.00	36.7
RICH	0.86	0.01	0.01	16.9
CL	0.89	0.02	0.24	30.0
WSAR	1.00	0.05	0.13	3.3
ESAR	1.49	0.05	0.24	24.6
RE	1.98	0.05	0.07	27.1
SL	2.10	0.24	0.41	29.7
CHIN	3.17	0.07	0.12	2.8
LS	3.36	0.21	0.17	10.3
BIS	3.58	0.04	0.07	21.7

[a] Burns and clearcuts less than 30 years old, as a percentage of each herd range.

underway. Estimates of the proportion of lines potentially requiring active restoration range from 15% (B. Coupal, unpublished data) to 90% (Lee and Boutin, 2006). For our base model we assumed that 50% of lines would require restoration, but we also sensitivity tested a high (80%) and low (20%) value because of uncertainty concerning this parameter.

We used early crown closure (reduced suitability for moose and white-tailed deer) as the endpoint of our simulated restoration program. As our base scenario, we assumed that 30 years of growth would be required to reach this endpoint, implying a rate of growth comparable to that observed in forest harvest blocks. However, relative to harvest blocks, seismic lines are subject to soil compaction, low light levels, low soil temperatures, and re-use, all of which reduce the rate of growth (Revel et al., 1984; Lee and Boutin, 2006). Therefore, we also modeled a pessimistic scenario in which 60 years would be required for the restoration endpoint to be achieved. In both cases we added 5 years to the growth period to account for the time required for planning, site preparation, and planting. To achieve these restoration outcomes we assumed that steps would be taken to prevent re-use and motorized access.

On seismic lines not requiring active restoration we assumed an even distribution in the stage of existing regeneration, meaning that a constant percentage of lines would achieve the restoration endpoint each year. The rate of tree growth on these lines was matched to the rate of growth on lines that were actively restored. As with actively restored lines, we assumed that steps would be taken to prevent re-use and motorized access.

We estimated the cost of restoration at $4000/km based on average costs of site preparation and planting in a pilot project undertaken in west-central Alberta (B. Coupal, pers. comm.). No cost was applied to lines that were regenerating naturally or to lines that crossed open peatlands and water bodies.

2.2. Habitat protection

Our habitat protection modeling focused on determining the opportunity costs (foregone revenue associated with alternative use of the land) of prohibiting new industrial development within each caribou range. We determined net present values (NPVs) for each of the four main industrial sectors active in caribou ranges in Alberta – conventional natural gas, conventional oil, bitumen (a tar-like hydrocarbon found in oilsands), and forest products – using models developed by Hauer et al. (2009a). These models projected expected resource flows, revenues and costs over time, and opportunity costs of capital in terms of discount or interest rates. From these projections we determined net resource values for each sector in present value terms (i.e., NPV).

For the oil and gas models the total amount of recoverable oil or gas available per geological layer in each section of land (~278 ha) was derived from spatially explicit data on reserves and ultimate potential housed with the Alberta's Energy and Resources Conservation Board and the National Energy Board (Alberta Energy and Utilities Board, 2007). The flow of resources over time given successful drilling was derived from estimates published by the Alberta Department of Energy (2007a). Seismic, operating costs, and capital costs were also obtained from the Alberta Department of Energy (2007a). Drilling costs were derived from Petroleum Services Association of Canada (2008). For the capital intensive oilsands projects, costs and bitumen outputs per well were derived from the Alberta Department of Energy (2007b,c). For each section of land, flows of oil or gas were multiplied by forecasted oil and gas prices, derived from GLJ Petroleum Consultants (2009a,b). This revenue stream was then discounted using a 4% real rate of return on investment. Discounted operating, drilling, and exploration costs were subtracted from this revenue to obtain expected NPV for each land section.

The raw NPVs represent the value of the resources as if all energy extraction projects were initiated immediately, with resource flows from each project varying in length from 5 to 40 years depending on type and quality of resource. However, the various projects cannot be developed all at once because the industry is subject to various capacity constraints such as drilling capacity, bitumen upgrading capacity and financial constraints. Projects that are scheduled later in time are subject to further discounting (which reduces their present value).

To estimate the effects of scheduling on NPVs we spatially scheduled resource extraction over a time horizon of 50 years using a linear programming model. The linear programming model maximized the NPV of development by choosing the optimal time to develop each section of land under capacity and demand constraints based on expected trajectories of oil, gas, and bitumen development (Alberta Department of Energy, 2007a; Oilsands Review, 2009). For example, the maximum capacity of bitumen extraction used in the model was approximately 4.6 million barrels a day. The linear program did not contain constraints related to caribou habitat or populations.

A heuristic description of the linear programming model is that land-sections are sorted from highest to lowest NPV for each resource type and then scheduled in ten 5-year periods. In each time period, projects are added until the capacity limitation was reached, with priority given to projects with the highest NPV. The final result is a 50-year schedule of land-section developments that generates the maximum sum of NPVs, while fitting within the expected trajectories of bitumen, oil and gas development. Additional detail on the calculation of NPVs and scheduling of projects is provided in Hauer et al. (2009a).

The NPV of land under forest management accounts for less than 1% of total land resource values but was included for completeness. NPVs for forestry were obtained using the methods described in Hauer et al. (2009b). The scheduling of forestry activities was based on maximizing NPV under provincial regulations including sustained yield constraints (Hauer et al., 2009b).

2.3. Wolf control

Wildlife managers faced with declining caribou populations have, in several jurisdictions, implemented wolf control programs as part of their caribou recovery efforts. The results of several large-scale wolf control programs suggest that the best outcome that can be practically achieved is a λ for caribou of 1.16, associated with a recruitment rate of approximately 45 calves per 100 cows (Farnell and Hayes, 1992; Boertje et al., 1996; Hayes et al., 2003). Not all wolf control programs achieve this level of ungulate response (Bergerud and Elliot, 1986; Boutin, 1992; National Research Council, 1997).

Reducing a wolf population by less than 60% is unlikely to achieve a positive outcome for caribou, and reducing a wolf population by greater than 80% is difficult to achieve in practice (National Research Council, 1997). Given these findings we modeled wolf control as a binary process: either maximal control was applied, or none at all. As our base scenario we assumed that wolf control would result in a λ of 1.10 for caribou. We also sensitivity tested λ values from 1.04 to 1.16.

The most common form of large-scale wolf control is aerial shooting using helicopters (National Research Council, 1997). The cost of these programs is largely a function of helicopter time and is proportional to the size of the control area. A pilot wolf control program recently undertaken in the LS range in west-central Alberta cost approximately \$35/km^2 per year to conduct (D. Hervieux, pers. comm.). We used this estimate to calculate the annual cost per herd of our simulated wolf control program, factoring in the size of the range.

2.4. Population model

The habitat restoration and wolf control models were linked together in a simple population model that was run separately for each herd. The objective of this model was to determine how long recovery would take and how long wolf control would be needed for each herd.

In our population model, annual changes in population size were calculated using the linear regression equation: $\lambda = 1.0184 - 0.0234 * \text{linear feature density} - 0.0021 * \text{percent young habitat}$ (see Appendix A). This empirical relationship follows the approach outlined by Sorensen et al. (2008) and was derived by correlating linear features and young forest, calculated at the scale of each herd range, with average lambda for the herds for which data were available (Appendix A). In our population model, the annual status of linear features was derived from the restoration model described earlier and parameters summarized in Tables 1–3. The percentage of each range comprised of young forest was calculated using government records of burns and harvest blocks occurring between 1976 and 2006 (Alberta Sustainable Resource Development, unpublished data; Table 2). As our base scenario we assumed that 1/30th of the original area of young forest would mature each year (i.e., matched to the rate of growth on seismic lines). We also modeled a scenario in which the pool of young forest did not change in order to isolate the effects of seismic line restoration.

Caribou population size at the start of the simulation was set to the current estimated population size (Table 1). The simulation was stopped once the herd was self-sustaining, defined as $\lambda \geqslant 1.0$ and population density above 0.045 caribou/km^2 (the mean estimated density of caribou in Alberta in 1996; Alberta Woodland Caribou Conservation Strategy Committee, 1996). If the herd did not become self-sustaining within 50 years the simulation was terminated.

We applied wolf control on an as-required basis to support caribou populations until habitat conditions permitted herds to be self-sustaining. Wolf control was applied whenever a herd's density was less than 0.045 caribou/km^2 and was stopped whenever the herd's density exceeded 0.06 caribou/km^2. Whenever the wolf control model was active the regression-based λ was replaced by the wolf control λ (Table 3). We assumed that whenever wolf control was stopped, λ would revert to the regression-based value after a lag period and that this decay would be linear over time (Boertje et al., 1996; National Research Council, 1997; Table 3).

Table 3
Model parameter values.

Parameter	Base model	Sensitivity tests
Caribou λ during wolf control	1.10	1.04–1.16
Residual effects of wolf control	3 years	2–4 years
Naturally regenerating seismic lines	50%	20–80%
Time for seismic restoration	35 years	65 years
Young forest regeneration	Yes	No
Target caribou population density	0.045/km^2	
Cost of restoration	\$4000/km	
Cost of wolf control	\$35/km^2	

3. Results

Only three herds – WSAR, ALP, and RICH – will maintain a population size greater than 10 animals for more than 60 years if current population trends continue, with no improvements in habitat condition or changes in industrial activity. The remaining herds will fall below 10 individuals in less than 40 years and two (CL, SL) in less than 20 years (Fig. 2).

In our recovery program simulations using base scenario parameters, half of the herds became self-sustaining ($\lambda \geqslant 1.0$ and density >0.045 caribou/km^2) in year 36, which is when the restored seismic lines reached crown closure. The other herds became self-sustaining in 20–30 years, with the notable exception of ALP which required only 7 years (Table 4). When the base model was run without regeneration of burns and harvest blocks only four of the herds became self-sustaining within the 50-year period of simulation (Table 4). Likewise, when the growth rate of restored seismic lines was reduced by half, only four of the herds became self-sustaining within 50 years (Table 4). Sensitivity testing the parameter for the proportion of seismic lines requiring restoration vs. regenerating naturally produced a maximum 8.5% change in the mean number of years until herds were self-sustaining. This means that as long as a line is regenerating well, the origin of regeneration (natural or human-assisted) matters little.

Table 4
Years until population density = 0.045 caribou/km^2 and $\lambda \geqslant 1.0$, achieved through habitat protection, seismic line restoration, and wolf control.

Herd	Base model	Fixed young forest[a]	Slow seismic restoration[b]
ALP	7	8	7
RRPC	21	36	26
RICH	25	50	28
WSAR	27	36	50
CM	28	50	29
CL	30	50	50
LS	36	50	50
BIS	36	50	50
RE	36	50	50
SL	36	50	50
CHIN	36	36	50
ESAR	36	50	50

[a] Base model parameters but amount of young forest held constant.
[b] Base model parameters but seismic restoration set to 65 years.

In the base model, the number of years that wolf control was applied varied from 0 to 16 among herds. Decreasing the effectiveness of wolf control from the base model value of 1.10 to 1.04 resulted in an 84% increase in the total cost of wolf control. Increasing this parameter to 1.16 resulted in a 24% decrease in total cost. When the parameter for the duration of residual effects of wolf control was sensitivity tested it had minimal influence on model results.

The cost of recovery varied greatly among herds (Table 5). In particular, herds in northeastern Alberta – ESAR, WSAR, CL, and RICH – had a per km^2 cost of recovery an order of magnitude greater than other herds. The total opportunity cost of habitat protection, in terms of potential resource revenue lost (assuming no substitution of energy activity outside herd ranges), was over 400 times greater than the cost of seismic restoration. The total cost of restoration was almost 10 times the cost of wolf control. In many herds, NPV was not uniformly distributed across the range (Fig. 3). RE, RICH, and ESAR, in particular, have very large differences in NPV from one part of the range to another.

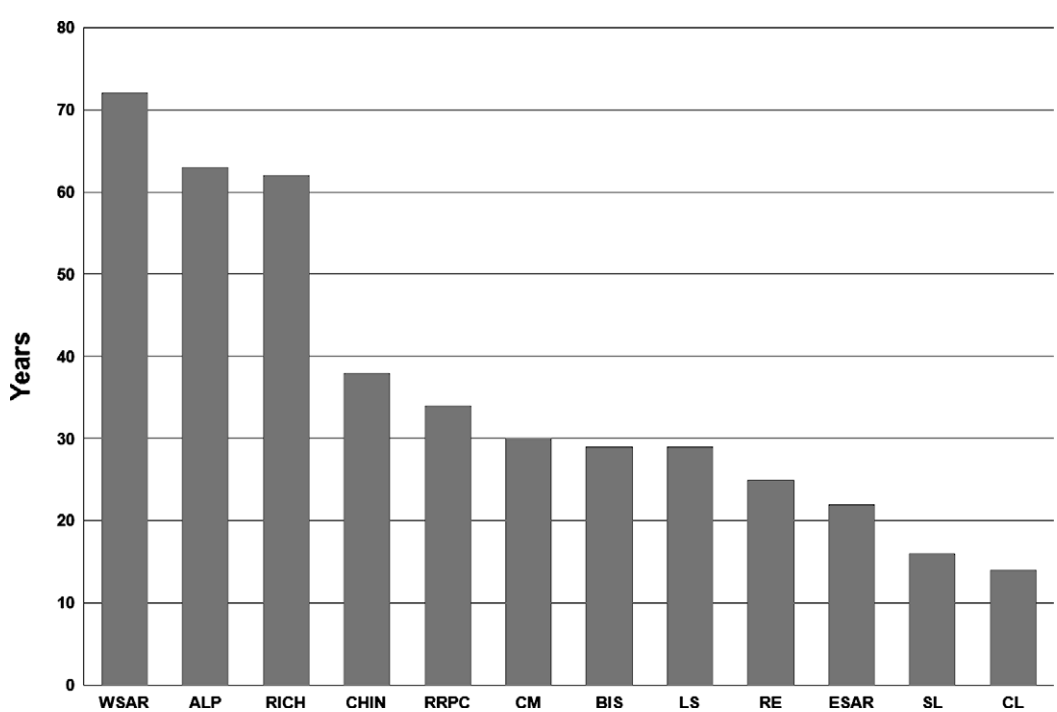

Fig. 2. Number of years until herd size is less than 10 animals given current population size and rate of decline.

Table 5
Recovery costs by herd and type of activity ($ millions CAD). Herds ranked on total cost of recovery per km^2.

Herd	Wolf control	Restoration	Protection[a]	Total per range	Total per km^2
ALP	1.53	4.7	13	19	0.01
BIS	5.37	87.2	739	832	0.06
CM	5.87	26.5	1165	1197	0.06
RRPC	0.00	4.7	514	519	0.11
LS	1.13	18.5	656	676	0.23
RE	6.71	51.4	7334	7392	0.46
CHIN	8.38	90.1	8958	9057	0.53
SL	1.43	12.5	2274	2288	0.67
WSAR	3.68	28.2	44,663	44,695	2.98
ESAR	8.23	36.2	50,117	50,161	3.41
CL	0.38	3.9	15,546	15,550	5.80
RICH	0.38	3.9	30,905	30,909	11.49

[a] Values represent the opportunity cost, in terms of NPV of petroleum and forestry resources, of prohibiting industrial activity in the entire herd range and assuming no substitution of activity outside of herd areas in response to the closure.

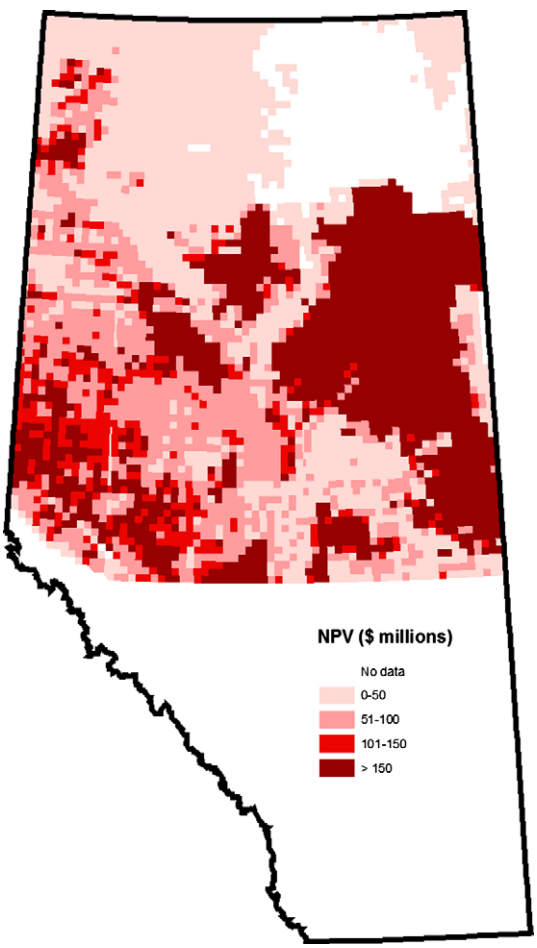

Fig. 3. Relative resource value of townships in northern Alberta based on estimates of unsequenced NPV.

We ranked the best (top 4) and worst (bottom 4) herds according to likelihood of extirpation, cost of recovery and likelihood of conservation success (Table 6). ALP, CM, and RRPC tended to rank highly while SL, ESAR, and CL tended to rank among the worst herds. RICH and WSAR tended to rank highly based on likelihood of success and likelihood of extirpation criteria but were among the most costly herds to recover.

4. Discussion

Our findings suggest that the implementation of a triage approach is warranted for the management of caribou in Alberta. First, the situation is critical and immediate action is required. None of the herds are currently self-sustaining and most will be functionally extirpated within three decades if current population trends continue. Population declines may even accelerate in the face of continued industrial expansion (Sorensen et al., 2008). Second, the projected opportunity cost of a comprehensive recovery program (in excess of $100 billion) exceeds the amount that might realistically be approved for this purpose. Third, herds vary greatly with respect to cost and response to recovery efforts. This variation provides a rational basis for ranking herds and differentially allocating recovery capacity.

In the application of a triage system to caribou in Alberta the conservation goal is (or should be) to maximize the long-term viability of caribou at the provincial-scale given the resources available. That being the case, herds with lower costs are preferred, because more can be recovered, but so are herds with higher intrinsic probability of survival (Hanski, 1991). This potential trade-off between cost of recovery and herd-level viability needs to be explicitly considered when allocating resources.

Decision making is complicated by the fact that the viability of individual herds is influenced by many disparate factors, which all need to be considered. We have attempted to quantify the various factors influencing the viability of herds based on the best available information. However, there is considerable uncertainty in some of the values and important data gaps remain. So rather than attempt a population viability analysis to make predictions about the fate of individual herds, we focused on making qualitative comparisons among herds by ranking herds with respect to several criteria related to viability (Table 6). We used time to decline to 10 individuals (current risk) and range size (long-term risk) as crude measures of relative risk of extirpation. Range size can serve as a proxy for potential population size and all else being equal, herds with small ranges will support smaller populations, with higher risk of extirpation, than herds with large ranges (Diamond, 1976). We used time to self-sufficiency as a relative measure of the likelihood of conservation success assuming herds taking longer to recover are more likely to fail. The current density of linear features and proportion of the range in young forest provides a measure of viability related to the effects of habitat disturbance on caribou λ (Sorensen et al., 2008). The greater the industrial footprint, the more the survival of a herd depends on the success of a habitat protection and restoration program. Finally, we provided the total cost of recovery of each herd expressed on a per km^2 basis.

If high recovery priority is based on higher likelihood of success, lower costs of recovery, and lower risk of extirpation then the ALP (4 of 5 criteria), RRPC (3 of 5 criteria), and CM (3 of 5 criteria) herds rank highly as candidates for recovery efforts, whereas the CL (3 of 5 criteria), ESAR (3 of 5 criteria), and SL (4 of 5 criteria) herds rank as low priority for recovery (Table 6). It is important to note that if prioritization were based on herds most at risk of extirpation, then herds such as CL, SL, and ESAR would receive most attention. Some of these (CL, ESAR) also have some of the highest recovery costs and will take the longest time to reach self-sufficiency. Some herds (RICH and WSAR) provide clear examples of the trade-offs that consideration of both cost of recovery and population viability create. These herds rate highly for viability measures but are among the most costly to recover due to the high opportunity costs of the hydrocarbons beneath their range.

Although we were unable to quantify it, the long-term viability of caribou herds in Alberta is also likely to be influenced, to varying

Table 6
Rank of herds with respect to risk of extirpation (years to reach 10 animals, range size, linear disturbance) likelihood of recovery (years to self-sufficiency), and cost of recovery.

Factor	*Best Herds*[a]				*Worst Herds*[a]			
Recovery cost per km^2	ALP	BIS	CM	RRPC	RICH	CL	ESAR	WSAR
Years to reach 10 animals	WSAR	ALP	RICH	CHIN	CL	SL	ESAR	RE
Range size	CM	CHIN	RE	WSAR	CL	RICH	LS	SL
Linear disturbance	ALP	RRPC	CM	RICH	BIS	LS	CHIN	SL
Years to self-sufficiency[b]	ALP	RRPC	RICH	WSAR	Tie: LS, BIS, CHIN, SL, ESAR, RE			

[a] Rankings reflect the best and worst four herds in each category based on our simulations. Best herds were those that had lower recovery costs, longer times to decline to 10 individuals, larger ranges, lower current linear disturbance, and fewer years to self-sufficiency.
[b] Based on our population modeling, reported in Table 4.

degrees, by climate change. Several ranges (e.g., WSAR, ESAR, RE) are expected to experience the climate of a parkland ecosystem within the next 50 years which may lead to changes in habitat composition and increased densities of white-tailed deer (Schneider et al., 2009). Additional research in this area is warranted.

The question of how many herds to "invest" in is a social decision that must include not only the costs presented here, but consideration of the value of caribou to society and ancillary benefits provided by habitat restoration and protection (whether they can be expressed in dollar terms or not). The role of triage is to make the most efficient use of whatever resources are allocated as a consequence of this societal trade-off decision. A formal optimization process will also ensure that tradeoffs do not remain hidden, compelling decision makers to take a stand and be held accountable for their choices.

Triage is not simply about picking winners and losers – the real objective is to maximize overall conservation gain given conservation capacity constraints. This means that resources that might have been expended on herds with little chance of survival need to be redirected to herds with greater viability, to ensure they survive. Offset programs would be effective for this purpose (Weber and Adamowicz, 2002; Ten Kate et al., 2004). For example, a funding pool obtained from industrial operators across all ranges could be established to undertake restoration programs in ranges selected for recovery. The same principle can be applied to habitat protection. If the economic opportunity costs are too high for the provincial government to protect all caribou ranges, then it would be better to fully protect the habitat of high priority herds than to inadequately protect all herds, including those that are unlikely to survive. These transfers of conservation capacity among herds need to be made transparently if public support for the triage approach is to be achieved. Otherwise the triage approach is likely to be perceived as a contrived justification for abandoning herds.

Our population modeling suggests that, under optimistic assumptions (i.e., full habitat protection and rapid restoration of seismic lines), almost all herds will require several decades to achieve self-sufficiency. This could easily stretch to 50 years or more if the growth rate of trees on restored lines is as slow as it has been on naturally regenerating lines (Lee and Boutin, 2006). The implication is that several decades of wolf control will be required to maintain most herds in Alberta. Also, it will not be possible to add any new industrial features to most caribou ranges for several decades without making matters worse for caribou. The potential for using restoration to offset disturbances from new development is thus effectively thwarted in the short-term by the long lag period for regeneration. It is also likely that, if new development is permitted, demands for continued access and re-use of existing seismic lines would reduce the effectiveness of restoration projects.

Our analysis of recovery costs included most of the main factors influencing cost but was not comprehensive. Nor did we quantify all sources of uncertainty. Consequently, our reported point estimates should only be considered accurate to the nearest order of magnitude, which is sufficient for their intended use in strategic decision making. For example, one can reasonably conclude that a provincial wolf control program could be implemented over the next 50 years for a few tens of millions of dollars, assuming that control efforts are generally as successful as they have been in the Yukon and Alaska. A provincial restoration program could be implemented for a few hundred million dollars, assuming that half of all seismic lines are treated and the cost is not substantially more than in the west-central Alberta pilot program. Finally, the cost of permanently protecting all caribou ranges in the province could exceed a hundred billion dollars, in terms of potential resource revenue that is forfeit.

The estimate of the NPV of resource revenue may seem inordinately high, but it simply reflects the fact that Alberta has 1.7 trillion barrels of bitumen in-place, and most are covered by caribou range (Alberta Department of Energy, 2009). If only 10% of the oil was recovered and the present value of oil was only $1.00 per barrel (accounting for development costs and a high rate of interest), then the NPV of this bitumen alone would be almost $200 billion. An important caveat to this analysis is that the actual economic impact of protection would be spread out over time. In the near term (i.e., the next decade or two) the effect of protection would be to redirect available industrial capacity to oilsands deposits outside of caribou ranges, thereby maintaining projected revenue flows to a significant extent. Also, the opportunity costs of protection would be offset to some degree by ecological benefits (above and beyond the recovery of caribou). Quantifying the net opportunity cost of protection, taking temporal and spatial rescheduling, and economic and ecological benefits into account, is an avenue of future research.

Given that wolf control costs are low (relatively speaking), and the costs of habitat protection are high, it would be desirable from an economic standpoint to maintain caribou through wolf control alone. Technically, this might be feasible, but the implementation of a long-term wolf control program across 117,000 km^2 of Alberta's forests in the absence of a defined endpoint or substantive efforts to make caribou herds self-sustaining is unlikely to achieve public support (Decker et al., 2006). In general, public support for wolf control is becoming increasingly difficult to obtain, even in Alaska and Yukon where it has been used extensively (National Research Council, 1997; Hayes et al., 2003). It is also worth noting that, in contrast to habitat protection and restoration, wolf control does not contribute to broader conservation objectives.

Another potential recovery strategy is to use caribou habitat as the unit of selection for triage, instead of entire caribou ranges as we did. In this way more herds could be protected (albeit only partially) for the same opportunity cost. An additional benefit of this approach is that substantially more caribou habitat could be protected for the same total cost by actively selecting lower cost habitat units (e.g., townships) over higher cost units. A limitation of this approach is that the risks associated with reducing effective

range size have not been quantified and this will hinder implementation. The relationship between range size and viability is likely to be nonlinear because, according to the leading hypothesis, the key to caribou viability is predator avoidance, not simply hectares of suitable forage (Rettie and Messier, 2000; McLoughlin et al., 2005; Bergerud, 2006; Latham, 2009). This implies that large contiguous tracts of land may have to be protected to prevent infiltration by predators from surrounding areas (or conversely, that protecting small portions of ranges may be futile). The determination of minimum patches of habitat required to effectively protect caribou is another important area of future research.

We chose to omit industrial best practices from our analysis of recovery strategies because their effects on caribou population dynamics have not been quantified. What we do know is that after 30 years of caribou management involving industry guidelines, best practices, and various restrictions on activities, Alberta's woodland caribou are now closer to extirpation than they ever have been (Hervieux et al., 1996; Environment Canada, 2008). It seems evident that the path to caribou recovery in Alberta is more likely to be found in aggressive recovery actions and difficult trade-off decisions than continued reliance on best practices and other management half-measures. With prompt action it should still be possible to achieve self-sustaining caribou herds in Alberta, but that window of opportunity is rapidly closing.

To conclude, our findings suggest that woodland caribou in Alberta meet the conditions whereby triage should be considered as an appropriate conservation strategy. Although our analysis was confined to caribou in Alberta, it is not unreasonable to assume that similar analyses for other threatened or endangered species would lead to similar conclusions, at least in some cases. Failure to consider the complicated trade-offs faced by government and society runs the risk that that conservation programs will fail at the point of implementation, as has too often been the case.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2010.04.002.

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