15th North American Caribou Workshop
Whitehorse, Canada
12-16 May, 2014

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Caribou Conservation and Management: What's Working?

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NACW at Thirty: A Work in Progress

The 15th North American Caribou Workshop (NACW) was held from 12-16 May 2014, in the traditional territories of the Kwanlin Dün First Nation and the Ta'an Kwäch'än Council, in Whitehorse, Yukon, Canada. This biennial meeting is the largest technical conference of its kind dealing specifically with caribou biology and management. The first NACW was held in Whitehorse over three decades ago in 1983, and 13 subsequent workshops have been held across North America until now. With nearly 400 delegates from Canada, the United States, Norway, and Greenland attending the 2014 conference, it is evident that this "North American" gathering has truly become an international event. Furthermore, delegates attending this 15th NACW represented federal, provincial, territorial, state and First Nation governments, academia, non-governmental organisations, co-management boards and councils, private consultants, and industry, creating a relatively unique conference setting bringing together a variety of perspectives and concerns. The breadth of the participants in terms of geography, expertise and affiliation resulted in a rich base of human capacity to discuss issues related to caribou conservation and management.

Given that it had been nearly three decades since the inception of this workshop, and with its return to the location of the first NACW, the organising committee felt it was a fitting opportunity to look back and assess what had been achieved with respect to caribou conservation and management. As such, the theme of the 15th NACW was "Caribou Conservation and Management: What's Working?" The opening session of the conference focussed on invited presentations explicitly addressing this question, and included topics on structured decision making, forest management, harvest monitoring, carnivore management, regional land use planning and management, and aboriginal perspectives on a long-term collaborative caribou recovery program in the southwest Yukon. We challenged our speakers to share what was working and why, and the information provided was valuable and timely, prompting many questions and discussion throughout the conference.

The organising committee received a strong response to our call for contributions, resulting in a final selection of over 70 presentations and 50 posters representing a broad range of topics including co-management, caribou habitat, predator-prey dynamics, population dynamics, status assessment, environmental impact assessment, land use planning, and monitoring. Related exchanges were lively, and often spilled into the breaks, which thanks to phenomenal spring weather, were held on the banks of the Yukon River. While no one was complaining about the warmth, it was a poignant reminder of the impacts of climate change on conditions facing caribou in the coming decades.

In addition to the main program, three pre-conference technical workshops were delivered on structured decision making, engaging local indigenous communities, and wildlife photography. As well, an evening event titled "A Celebration of Caribou" was hosted for the public and received very positive feedback. It included videos and stories and was an opportunity for people of all ages to learn more about caribou. These events highlight the increasing importance of social dimensions of caribou management on conservation outcomes.

It is traditional for contributors to NACW to have an opportunity to publish their work in proceedings from the conference. At one time, this captured much of the caribou research taking place. However, caribou have moved into the mainstream, with related theoretical and applied research appearing in a broad array of journals, testimony to the breadth and quality of work being conducted, as well as its sheer volume. The contributions profiled here highlight the diversity of approaches being applied to contemporary caribou conservation, including active habitat management (Stevenson & Coxson, Bentham & Coupal), legislative tools for protection (Ray *et al.*, Poole *et al.*), and application of new, and in some cases controversial, conceptual frameworks (Gonzales *et al.*, Robichaud & Knopff). These speak to the opportunities and challenges that managers face in finding workable solutions to sustaining caribou now and into the future.

On behalf of the 15th NACW organising committee we thank the sponsors, volunteers, and delegates for their shared commitments to the real Rangifer. We look forward to a continued tradition of inclusive and constructive dialogue and demonstration of innovative approaches to caribou conservation and management at future NACW meetings.

Troy Hegel, *Yukon Department of Environment*
Fiona Schmiegelow, *University of Alberta and Yukon College*

Past North American Caribou Workshops

1st – Whitehorse, Yukon (28-29 September 1983)
2nd – Val Morin, Québec (17-20 October 1984)
3rd – Chena Hot Springs, Alaska (4-6 November 1987)
4th – St. John's, Newfoundland and Labrador (31 October – 3 November 1989)
5th – Yellowknife, Northwest Territories (19-21 March 1991)
6th – Prince George, British Columbia (1-4 March 1994)
7th – Thunder Bay, Ontario (19-21 August 1996)
8th – Whitehorse, Yukon (20-24 April 1998)
9th – Kuujjuaq, Québec (23-27 April 2001)
10th – Girdwood, Alaska (4-6 May 2004)
11th – Jasper, Alberta (23-26 April 2006)
12th – Happy Valley-Goose Bay, Newfoundland and Labrador (2-5 November 2008)
13th – Winnipeg, Manitoba (25-28 October 2010)
14th – Fort St. John, British Columbia (24-28 September 2012)

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Can partial-cut harvesting be used to manage terrestrial lichen habitat? A review of recent evidence

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Abstract: Recent research suggests that partial-cut harvesting techniques can be used to alter successional trajectories in pine- and spruce-lichen woodlands, allowing forest managers to extend the period of reindeer lichen growth in mid- to late seral boreal forest stands. In Quebec, a fully replicated partial-cutting trial found that terrestrial lichen abundance remained at least as high in the partial cut as in the clearcuts or unlogged stands, and that the partial cut appeared to be on a trajectory to have even more terrestrial lichen due to sustained higher growth rates. In Alberta, a retrospective study found higher terrestrial lichen abundance in an early horse-logged partial cut than in undisturbed adjacent old forests or in clearcuts. Follow-up studies of partial-cut harvesting trials in British Columbia found that group selection plots 10 years after harvesting had lichen cover equivalent to that of undisturbed forest. In contrast, studies on lichen woodlands that have been defoliated by mountain pine beetle showed a major decline in reindeer lichen cover and a corresponding increase in vascular plant cover, similar to the results of previous studies on clear-cut logging impacts. Taken together these studies provide qualified support for the hypothesis that partial-cut harvesting can be used to enhance, or at least maintain, terrestrial lichen mats used as forage by caribou.

Key words: forest management; lichen woodlands; partial-cut harvesting; terrestrial lichens; woodland caribou.

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Introduction

The changes in stand structure that occur during succession in boreal and sub-boreal lichen woodlands are accompanied by major shifts in the composition of forest floor lichen communities. We review the impacts of three major types of disturbances: fire, partial-cut logging, and canopy mortality due to mountain pine beetle attack. Although data from partial cuts are most relevant to our objectives, they are the most limited, and there are also valuable lessons to be learned from other successional series.

Post-fire succession in the boreal forest

Ahti (1959) described 5 stages in the development of boreal forest terrestrial lichen communities after fire. 1) The first stage was a bare soil or organic substrate stage in the immediate

Figure 1. *Cladonia mitis* mat in late-successional pine forest (130 years old) in the Omineca River watershed, British Columbia, shows progressive infilling and burial of lichen mat by feather moss fronds. The individual clump shown here is ca. 30 cm in diameter.

post-fire environment. Soil microclimate conditions can be extreme during this period, with soil surface maxima reaching 60°C (Rouse, 1976). 2) This was followed by a crustose lichen stage, with species such as *Trapeliopis granulosa* abundant. *Polytrichum* moss mats were common during the crustose lichen stage in the Northwest Territories (Maikawa & Kershaw, 1976), along with crustose species such as *Lecidea granulosa* and *L. uliginosa*. 3) From about years 20 - 60 a cup-lichen (squamulose species in *Cladonia* subgenus *Cladonia*, often referred to as *Cladonia* morphotypes) stage develops, with species such as *Cladonia cornuta* and *C. sulphurina* abundant. Skatter *et al.* (2014) noted that reindeer lichens (mat-forming species in *Cladonia* subgenus *Cladina*, often referred to as *Cladina* morphotypes) were abundant 20 - 30 years after fire in Jack Pine stands from northern Saskatchewan. As the forest canopy ages, both cup lichens and reindeer lichen mats decline in abundance, due to the constraints of growing in more shaded understory environments where there is a high rate of needle litterfall. 4) Between 60 - 120 years after fire reindeer lichens can reach their greatest period of abundance. Species such as *C. arbuscula*, *C. rangiferina*, and *C. uncialis* are common during this stage (Coxson & Marsh, 2001). 5) In some regions, especially in sites with oceanic climates, *C. stellaris* mats develop as the final lichen successional phase. Regional variants occur, however, with *Stereocaulon paschale* woodlands, for instance, replacing *C. stellaris* woodlands in a zone extending from west of Churchill across to Great Slave Lake, immediately north and south of latitude 60° N (Kershaw, 1977).

As lichen woodlands age, progressive canopy closure can occur, with feather-moss mats gradually coming to dominate the forest floor surface (Maikawa & Kershaw, 1976; Coxson

& Marsh, 2001). As the understory becomes cooler and moister feather-moss mat fronds can infiltrate and eventually bury existing reindeer lichen mats (Fig.1). Coxson & Marsh (2001) documented the shift from dominance by Cladina morphotype lichens in mature stands to dominance by pleurocarpous mosses in older stands (Fig. 2) within sub-boreal forests in central-interior BC. The biomass of *Cladina* morphotype lichens, including *C. mitis*, *C. rangiferina*, *C. stellaris*, and *C. uncialis*, in 50 - 100 year old stands exceeded 1700 kg/ha, falling to less than 300 kg/ha after replacement by feather-moss mats had occurred in the stands > 100 years in age (Coxson & Marsh, 2001).

Several regional exceptions to this pattern of canopy closure and domination by feather-moss mats in old-growth lichen woodlands occur. In the clay belt of northwestern Quebec paludification in old stands can lead to reduced vigour and lower tree densities, with the forest floor surface gradually becoming dominated by *Sphagnum* mats (Boudreault *et al.,* 2002; Harper *et al.,* 2003). In cooler oceanic climates, such as in boreal forests in the Grande rivière de la Baleine area of northeastern Quebec, no transition to feather-moss mats occurred even in very old stands, with *C. stellaris* mats dominating the forest floor in stands at least 250 years old (Morneau & Payette, 1989).

Successional changes in forest floor lichen communities reflect both stochastic factors, such as the availability of propagule sources over time (Hilmo & Såstad, 2001), and the response of individual lichen species to gradients of temperature, moisture, and light availability (Tegler & Kershaw, 1980; Kershaw, 1985). Generally, early successional lichen species are thought to be more tolerant of heat extremes associated with post-fire surface microclimate, while late-successional lichen species are more sensitive to extremes of desiccation and heat exposure (Kershaw, 1977; Kershaw, 1985). Terrestrial lichen communities are also sensitive to stand nutrient availability. Lichens can access stand level nutrient reserves indirectly, from exposure to through-flow precipitation in drip-zones under shrubs and trees (Haughian & Burton, 2015), and for some species directly, using rhizines to mobilize soil or bark nutrient reserves (Cornelissen *et al.,* 2007).

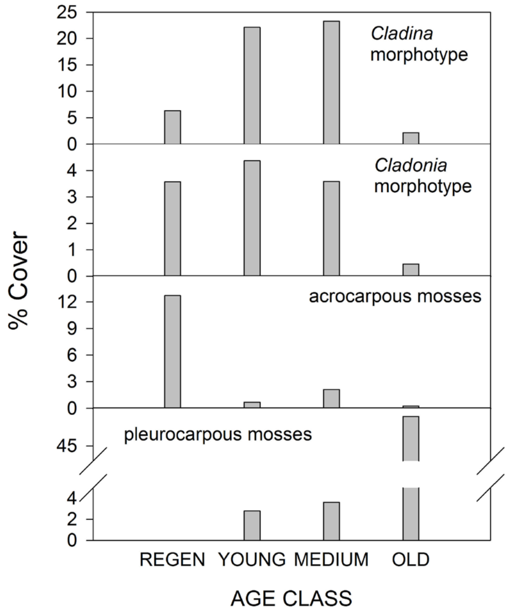

Figure 2. Mean cover (%) by functional group within 50-year age-class intervals (n= 4, 4, 5, and 2 stands respectively) for pine-lichen woodlands in central-interior British Columbia. The *Cladina* morphotype functional group consisted of *Cladonia arbuscula*, *C. rangiferina*, and *C. stellaris* (adapted from Coxson and Marsh, 2001).

Abundance and viability of terrestrial lichens after partial cutting

Partial-cutting provides a means of bringing about changes in stand structure through removal of selected trees. If the objective of forest managers is to prolong the period of reindeer lichen growth in forest stands, the impact of changes in stand structure from partial-cut harvesting should be greatest in late successional stands, especially if canopy closure has already started to occur. Selective removal of trees in late-seral stands should alter forest floor microclimates, creating conditions that resemble more closely those found in earlier stages of stand development. We hypothesize these warmer and drier conditions at the forest floor surface after stand thinning should preferentially favour growth of reindeer lichens over that of feather moss mats. Support for this hypothesis derives both from studies of lichen response to changes in stand structure during post-fire succession, and from observed changes in lichen communities after partial-cut harvesting.

Measurements in undisturbed pine-lichen stands have shown a positive relationship between terrestrial forage lichen abundance and various measurements of solar radiation (e.g., Cichowski *et al.,* 2009; Haughian, 2010), or, conversely, a negative relationship between terrestrial forage lichen abundance and various measurements of the tree canopy (e.g., Coxson & Marsh, 2001; Sulyma & Coxson, 2001). At the microsite level, forage lichens are positively associated with canopy gaps and negatively associated with the areas under tree crowns (Sulyma & Coxson, 2001; Haughian, 2010). A significant preference of *Cladina* morphotypes for microsites with greater canopy exposure within older stands was found by Sulyma & Coxson (2001) who analyzed canopy exposure of lichen mats in 100- to 130-year old pine-lichen woodlands (Table 1). Jonsson Čabrajić *et al.* (2010) similarly showed that in northern Scandinavia the optimum growth of *Cladonia stellaris* and *Cetraria islandica* occurred in forests with <60% canopy cover, corresponding to a basal area of ca 15 m^2 ha^{-1}, and suggested that the majority of forests in that region are now too dense to maintain optimal lichen growth.

Indirect evidence is provided by examining the response of lichen mats to clear-cut logging. Coxson and Marsh (2001) found that in stands where forest cover had been removed by logging 10 years previous on a deep winter snowpack, reindeer lichens were more abundant on shaded north-facing cut block margins than in the adjacent unlogged forest, reflecting the likely influence of greater light availability and higher humidity (with extended duration of wetting episodes) on lichen growth in these sites. They also noted that regeneration of trees and alders in the developing clearcut would soon curtail the development of lichen mats. The sensitivity of feather moss mats to increases in insolation exposure has previously been observed in large clearcuts, especially on drier sites, where feather mosses often show signs of bleaching and dieback (Kershaw *et al.,* 1994).

Evidence for the maintenance or enhancement of reindeer lichens after thinning or par-

Table 1. Mean microplot values (± 1S.E.) for leaf area index in moss- versus lichen-dominated plots in pine lichen woodlands from central-interior British Columbia. Significance values are shown for t-test comparisons between 'Moss' and 'Lichen' plots at each location (from Sulyma & Coxson 2001).

Plot variable	Site	Moss plots	Lichen plots	Significance values
Leaf Area Index ($m^2 \cdot m^2$)				
	West Germansen	1.617 6 (0.016)	1.551 (0.014)	$P < 0.0001$
	Germansen Lake	1.700 6 (0.024)	1.637 (0.0126)	$P = 0.0004$
	Manson	1.961 6 (0.030)	1.787 (0.020)	$P = 0.0003$

tial removal of canopy structure comes from several studies. Snyder and Woodard (1992) did a retrospective study of lichen abundance in 18 different-aged clearcuts, one 20-year-old horse-logged partial cut, and two unlogged stands within the Subalpine Ecoregion of Alberta. Terrestrial lichen cover, in general, was greater in the partial cut than in the unlogged stands. Lichen cover in the unlogged stands was similar to that in the 20- and 30-year-old clearcuts, but greater than that in the 10-year-old clearcuts. Percent cover of *Cladonia* species palatable to caribou, however, was greater in the partial cut than in the unlogged stands, and greater in the unlogged stands than in the clearcuts (Snyder & Woodard, 1992), suggesting that in this case, partial cutting not only retained preferred forage lichens but actually enhanced them compared to clearcutting.

In a replicated study in Québec black spruce forests, Boudreault *et al.* (2013) compared the abundance and growth rates of three *Cladina* morphotype species (*C. stellaris*, *C. mitis*, and *C. rangiferina*) in three treatment types: partial cuts, clearcuts, and controls. Both unlogged stands and partial cuts had higher percent cover of *Cladina* spp. than clearcuts. Growth rates of the *Cladina* morphotype species were higher in both the partial-cut and clearcut sites than in the control sites (Fig. 3). In this forest type, it appeared that partial cuts offered the best combination of retention of preharvest forage lichens, and environmental conditions promoting lichen growth after harvest. Interestingly, measurements for *C. stellaris* and *C. mitis* showed negative growth rates (loss of biomass) in the control stands, perhaps indicative of an already declining status for *Cladina* morphotype species in these late seral stands. As Moser *et al.* (1978) noted, however, negative growth rates may be a periodic feature of even healthy lichen mats, during episodes of unfavourable climatic conditions.

One of the most comprehensive studies to

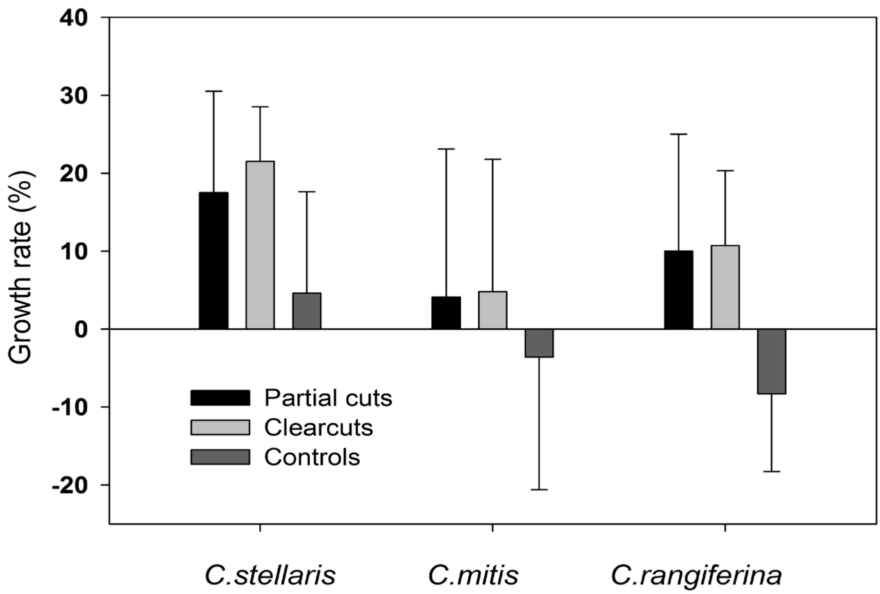

Figure 3. Mean growth rates (+ 1 S.E.) of *Cladonia stellaris*, *C. rangiferina*, and *C. mitis* in black spruce boreal forests of western Québec from spring 2005 to fall 2006. Differences in growth rate that were significant ($P < 0.05$) according to least squares means Tukey's honestly significant difference tests are indicated by different letters (excerpted from Boudreault *et al.,* 2013).

date on the impact of partial-cutting on terrestrial lichen mats was conducted by British Columbia's Ministry of Forests and Range, who began a replicated silvicultural systems study in 1995, described by Miège *et al.* (2001), with follow-up studies described by Waterhouse *et al.* (2011). The study was carried out in the western portion of the range of the Itcha-Ilgachuz caribou herd in British Columbia's central interior plateau. Compared to other forested BC ecosystems, the climate of the Dry, Cold sub-Boreal Pine Spruce biogeoclimatic zone (SBPSxc) and Very Dry Very Cold Montane Spruce biogeoclimatic zone are very dry and cold (Meidinger & Pojar, 1991), with light levels within stands sufficient to allow pine regeneration in the understorey (Waterhouse *et al.,* 2010).

Waterhouse *et al.* (2011) described results from their measurements taken in 2004 (8.5 years after logging), reporting on the following treatments:

- Irregular group shelterwood with stem-only harvesting (IGS-SO)
- IGS with whole tree harvesting (IGS-WT)
- Group selection with stem-only harvesting (GS-SO)
- No harvest

The irregular shelterwood prescription called for 50% removal in openings ranging from 20 - 30 m in diameter, and was designed to provide partial shade for terrestrial lichens in the openings. The group selection prescription called for 33% removal in openings about 15 m in diameter, and was designed to maintain arboreal lichens. In stem-only treatments, debris from topping and delimbing was aggregated and left in the harvested openings. In the whole tree treatment, debris from topping and delimbing was piled and burned at the roadside. Post-harvest surveys showed that the actual area cut was 39% in the irregular group shelterwood and 28% in the group selection, and that the opening sizes were within the targeted range. Although not part of the experimental design, three adjacent clearcuts were also monitored, beginning in 2001.

Cover of forage lichens was significantly lower in the three harvested treatments than in the unharvested control at the first reassessment after harvesting, and subsequently increased. Declines in the harvested units were greater in the openings than in the residual forest stand. In the IGS-SO treatment, healthy forage lichens had declined to 49% of preharvest levels in 1998, and increased to 68% of preharvest levels by 2004. In the IGS-WT treatment, healthy forage lichens had declined to 57% of preharvest levels in 1998, and increased to 71% of preharvest levels by 2004. In the GS-SO treatment, healthy forage lichens had declined to 53% of preharvest levels in 1998, but had nearly reached preharvest levels again by 2004 (Fig. 4). Total moss cover also showed a

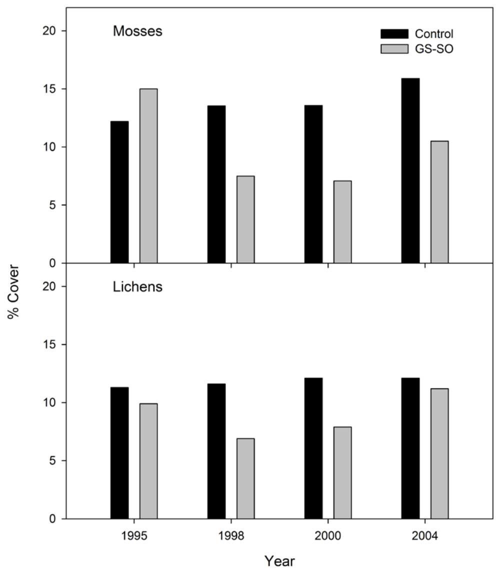

Figure 4. Percent cover of mosses (all moss species) and lichens (preferred forage species) in control versus irregular group shelterwood (stem-only: GS-SO) harvesting plots under pre-harvest (1995) and post-harvest (1998, 2000, 2004) conditions on caribou winter range in west-central British Columbia. Adapted from Waterhouse *et al.,* 2011.

similar pattern of decline and then recovery in the GS-SO treatment, though values remained well below that of the control stands in 2004. This recovery to levels below that of control stands is consistent with a negative response of moss mats to reduced canopy cover, reflecting, among other factors, the loss of stored carbon reserves in feather moss-mats when rewetting follows intense desiccation events (Wilson & Coxson, 1999).

Dwarf shrub cover, initially at 8 - 10%, declined slightly in the partial cuts immediately after harvest, then increased to slightly (2 - 6%) above preharvest levels. Herb cover was only 2 - 3% in the preharvest stand, and increased to 3 - 4% in the partial cuts by 2004. Cover of dwarf shrubs and herbs had a significant negative relationship with cover of healthy forage lichens at all measurement times. Seven to eight years post-harvest in the adjacent clearcuts, forage lichen cover was low (2.8%), dwarf shrub cover was similar (16.9%), and herb cover was high (17.4%), compared to the partial cuts.

Lessons from mountain pine beetle attacks

The mountain pine beetle epidemic has killed at least 726 million m3 of timber in BC's interior forests, affecting more than 17.5 million ha and killing most of the mature lodgepole pine in the Central Plateau region of the province (British Columbia Ministry of Forests, Lands and Natural Resource Operations, 2012). Cichowski (2011) reviewed literature on the potential effects of the epidemic on caribou. In addition to the western portion of the Itcha-Ilgachuz winter range (Waterhouse, 2011), data on the effects of mountain pine beetle mortality on terrestrial lichens are available from three other study areas: the Tweedsmuir-Entiako winter range (Cichowski *et al.,* 2008), the Kennedy Siding winter range (Seip & Jones, 2009), and the northeastern portion of the Itcha-Ilgachuz winter range (Cichowski *et al.,* 2009). The level of pine mortality was moderate to high in all four areas, ranging from 46 - 75% in the Itcha-Ilgachuz to 78 - 96% in the Tweedsmuir-Entiako (Cichowski, 2011).

Cover of terrestrial forage lichens in unlogged stands decreased after the mountain pine beetle attack in all four study areas. In the Tweedsmuir-Entiako, mean lichen cover was 16% in 2001, 13% in 2003, 11% in 2005, and 10% in 2007. The rate of lichen decline had diminished or stopped in most site series by 2007 – about 7 or 8 years after the initial attack (Cichowski *et al.,* 2008). In the northeastern portion of the Itcha-Ilgachuz range, mean lichen cover declined from 20.5% on plots established in 2005 and 2006 to 16.4% in 2008 (Cichowski *et al.,* 2009). In the western portion of the Itcha-Ilgachuz, terrestrial lichen cover in the no-harvest controls for the silvicultural systems study of Waterhouse *et al.* (2011) declined from the 2004 pre-mountain pine beetle cover of 11.3% to 9.7% in 2008, but this change was not statistically significant (Waterhouse, 2011). At Kennedy Siding, terrestrial lichen cover was 24% in 2006, 17% in 2007, and 12% in 2008 and 2009 (Cichowski, 2011).

Concurrently, cover of dwarf shrubs increased in all four study areas. On most sites in the Tweedsmuir-Entiako range, kinnikinnick (*Arctostaphylos uva-ursi*) cover increased from about 30% to about 60% between 2001 and 2007 (Cichowski *et al.,* 2008). In some, but not all, subzones, the rate of increase levelled off after 2005, and dieback of kinnikinnick was observed in some plots. Informal observations in 2010 suggested that the kinnikinnick dieback was continuing and was more widespread (Cichowski, 2011). In the northeastern portion of the Itcha-Ilgachuz, kinnikinnick cover increased significantly from 3.6% in 2005-06 to 6.2% in 2008, and twinflower (*Linnaea borealis*) cover increased significantly from 3.4% to 8.5% (Cichowski *et al.,* 2009). In the western portion of the Itcha-Ilgachuz, mean percent cover of dwarf shrubs increased from 10.1% in

2004 to 17.3% in 2008 (Waterhouse, 2011); species that increased included kinnikinnick, twinflower, crowberry (*Empetrum nigrum*) and grouseberry (*Vaccinium scoparium*) (Cichowski, 2011). At Kennedy Siding, velvetleaf huckleberry (*Vaccinium myrtilloides*) exhibited a 17.5% increase in cover from 2006 to 2008 (Seip & Jones, 2009). Moss cover generally declined (northeastern Itcha-Ilgachuz, Kennedy Siding) or remained the same (Tweedsmuir-Entiako, western Itcha-Ilgachuz) in mountain pine beetle-attacked stands (Cichowski, 2011).

The death of some, but not all, of the trees in the forest canopy, either through mountain pine beetle attack or through partial cutting, may affect terrestrial lichen abundance in several ways. The amount of light reaching the forest floor is increased. There is increased deposition of needles and woody debris on the forest floor, although the timing of deposition differs between insect mortality and logging, and with logging method. Water relationships may also change. Surface drying increases with increased irradiance, but soil moisture may increase as there are fewer canopy trees to transpire moisture into the atmosphere. Significant changes in surface temperature and humidity profiles can occur with loss of canopy cover. Greater insolation exposure in sites with dry continental climates can lead to cooler night-time temperatures and more intense early morning dewfall, triggering wetting events in those lichen species capable of rehydration from dewfall and/or high humidity events alone, similar to those observed by Bidussi *et al.* (2013) for epiphytic lichens in interior BC. However, these same sites with reduced canopy cover will typically experience lower mid-day humidity and thus more intense mid-day drying of forest floor moss and lichen mats. All these changes may affect lichens directly, or may affect them indirectly by altering relationships between the lichens and their competitors.

Although studies in undisturbed lichen woodlands consistently show a positive relationship between terrestrial forage lichen abundance and canopy openness, mountain pine beetle mortality, which increases light transmission, has been associated with declines in terrestrial lichen cover. In the northeastern portion of the Itcha-Ilgachuz range, Cichowski *et al.* (2009) found that the greater the increase in light transmission following the mountain pine beetle epidemic, the greater the decrease in terrestrial lichen abundance. Between 2001 and 2007 in the Tweedsmuir-Entiakio Range, more mountain pine beetle mortality was associated with a greater decrease in lichen cover, but once the effect of mountain pine beetle mortality was taken into account, higher light levels had a positive effect. A separate examination of trends between 2005 and 2007 indicated that changes in light transmission or interception had a significant effect on changes in lichen abundance, suggesting that by seven to eight years after the initial attack, increased light was beginning to create favourable conditions for lichen recovery (Cichowski *et al.,* 2009).

Armleder and Waterhouse (2008) and Waterhouse *et al.* (2011) have asserted that lichen morphotypes that have been growing in subdued light are adversely affected by a sudden increase in exposure to light. Lichens of the same species growing under different light conditions exhibit both morphological and physiological differences. Lichens have the capability of responding to changes in the light regime by increasing the production of phenolic compounds believed to have a protective function; in *Cladina* morphotypes newly exposed to enhanced levels of ultraviolet light, accumulation of phenolics increased and the penetration of ultraviolet radiation into the lichen thallus was more strongly attenuated than in lichens not exposed to enhanced levels of radiation (Buffoni Hall *et al.,* 2002). Gauslaa *et al.* (2012) noted that a rapid transition to high light exposure in desiccated lichens (this in work on

Lobaria pulmonaria) resulted in chlorophyll degradation, an interaction that was exacerbated under low relative humidity conditions. This same high light exposure while the lichens were desiccated gradually, however, triggered changes in the photochemistry of the lichens that improved their tolerance of excess irradiance in the desiccated state (Stepigova *et al.,* 2008).

It is therefore possible that the increase in light levels brought about by beetle mortality of canopy trees reported by Cichowski *et al.* (2009) induced mortality in some lichens, and that those that survived had acclimatized to increased light levels seven to eight years after the initial beetle attack. Differences in increased light exposure could also explain why forage lichens in 15-m wide openings recovered faster than those in 30-m wide openings between 2000 and 2004 (Waterhouse *et al.,* 2011), but were then adversely affected by a further increase in light exposure from mountain pine beetle attack between 2004 and 2008, while those in the 30-m wide openings were not (Waterhouse, 2011). Gauslaa *et al.* (2006) noted that lichen response to changing canopy conditions is a delicate balance between growth potential (more with increasing light) and desiccation damage, which is greater under high light conditions.

Interactions with understory shrubs and woody debris

Mortality of canopy trees from mountain pine beetle attacks is followed by a period of two or three years during which the needles fall from the dead trees, covering ground vegetation and producing a pulse of nutrients. Although lichens do not generally dominate areas immediately under the crown-radius projection of canopy trees (Haughian, 2010), they may be affected by vegetation changes within those areas. Kinnikinnick, a major competitor with forage lichens, often forms circular colonies (skirts) in the high needle litter deposition zone beneath live lodgepole pine trees. Cichowski *et al.* (2008) observed that the rapid expansion of kinnikinnick in the Tweedsmuir-Entiako study area following mountain pine beetle attack corresponded to the massive needle deposition by canopy trees and to the increased availability of nutrients, light and water. They suggested that the needle-pulse combined with the increase in available nutrients allowed kinnikinnick to expand beyond its 'skirt' at the base of trees, into areas normally too poor in nutrients to support kinnikinnick. Needle deposition throughout their study area has slowed down, and the extra nutrient pulse that may have allowed kinnikinnick to establish in marginal habitats is no longer available, resulting in a slowing of kinnikinnick growth or even die-back.

Over time, woody debris levels will increase after mountain pine beetle outbreaks as branches gradually break off the dead trees and the trees begin to fall, but the mountain pine beetle-affected stands reviewed by Cichowski (2011) had not yet reached this stage. After stem-only partial cutting, there is a pulse of litter that is not confined to pre-existing needlefall zones, and is composed of both logging slash and needlefall. It is therefore more likely to cover lichen mats than the needles that fall from beetle-killed trees, and was identified by Miège *et al.* (2001) as a factor adversely affecting post-harvest lichen abundance. By 2008, however, the difference between the group selection and the two irregular group shelterwood treatments, which differed in level of removal and opening size, was far greater than the difference between stem-only harvesting and whole-tree harvesting (Waterhouse *et al.,* 2011), and the authors observed that woody debris left over from stem-only harvesting may have ameliorated microclimate conditions for the lichens.

Lichen management using partial cuts

Current literature supports the concept that partial cutting can be used to maintain, and in some cases perhaps enhance, the preharvest lichen community. This maintenance strategy of partial-cut harvesting has gained acceptance as a viable management approach in BC, especially in the relatively dry and cold Montane Spruce Zone (Meidinger & Pojar, 1991). Partial cutting with the objective of maintaining lichen forage is a key part of the management approach on more than 181,000 ha in the range of the Itcha-Ilgachuz caribou herd (Armleder & Waterhouse, 2008), and was recommended by McNay (2011) for high-elevation plateaus in the Montane Spruce Zone in B.C. Specifically, McNay (2011) recommended an irregular group shelterwood system with openings not exceeding two tree lengths wide by three or four tree lengths long for the purpose of maintaining a sustainable supply of terrestrial forage lichens. He added the following caveat: "Practitioners are advised that while this specific silvicultural regime has been shown to maintain terrestrial forage lichens, further monitoring may be necessary to prove the regime does not subject caribou to greater spatial overlap with an early-seral predator-prey system" (McNay, 2011: p. 68).

Particular care should be taken with respect to the adverse impacts of logging slash (needles and woody debris) on terrestrial forage lichens, as described by Gough (2010). The exclusion of woody debris that occurs under a branch and stem harvesting system probably has mixed impacts on terrestrial forage lichens. In a comparison of several different clearcut harvesting methods in Alberta, Kranrod (1996) found that stump-side delimbing in combination with winter harvest and without scarification left more terrestrial lichen cover the summer after timber harvest than any other treatment combination. He observed that environmental conditions under piles and at pile edges were moderated, and appeared to provide suitable microenvironments for lichens, whereas lichens present in road-side delimbed sites without cover often appeared to be suffering damage from exposure.

Waterhouse *et al.* (2011) observed that slash on the ground and suspended low above the ground adversely affected lichens, but high suspended slash and areas adjacent to slash piles may provide refugia for lichens. An important point raised by Waterhouse *et al.* (2011) was that whole-tree (WT) and stem-only (SO) harvesting systems have the potential of affecting lichens differently. Whole tree skidding is likely to cause more damage to the lichen mat than skidding of delimbed stems, and results in roadside processing areas that are severely disturbed. The slash generated from on-site processing covers lichens. Percent cover of logging slash had a significant negative relationship with abundance of healthy forage lichens at all measurement times, but forage lichen abundance in the two irregular group shelterwood (IGS) treatments did not differ significantly from one another at any time. The authors noted that slash deposited on the ground crushed lichen, while low suspended slash prevented light and precipitation from reaching lichen mats. The influence of slash on lichen regrowth may be broadly similar in other disturbance features, such as seismic lines, which at a microsite level can lead to increased canopy openness. However, as discussed below, linear disturbance features have other attributes which can significantly reduce their value as habitat for caribou.

Harvesting with a high level of retention and small patch size, as recommended by Gough (2010) is a key component of any successful partial-cutting lichen retention strategy. Terrestrial forage lichens in the 30% removal treatment of Waterhouse *et al.* (2011) recovered to preharvest levels, whereas lichens in the 50% removal treatment with 30-m openings did not (Waterhouse *et al.,* 2011). Sulyma (2002) provided a summary of the predicted interactions between harvesting methods, harvesting season, site preparation, and regeneration method (Table 2). Sulyma recommended that harvesting occur during the winter season to minimize disturbance of lichen mats during harvesting with whole tree removal preferred to minimize the amount of residual debris left after harvesting.

The broader context of caribou habitat management

Caribou populations are affected by many factors other than the abundance of terrestrial forage lichens, and forest harvesting can affect caribou in ways other than through its impact on lichens. Here we discuss mainly the impacts of forest harvesting on terrestrial forage lichens, but a landscape-level management plan should take into consideration the entire array of human-mediated influences on caribou populations.

A cornerstone of present-day caribou management in North America has been to consider interactions between habitat modifications and predator response. Current findings suggest that factors which increase predator populations, for instance, creation of early seral habitats which favor growth of moose or deer populations, or those which allow more ready predator dispersal, such as the creation of road networks, can have major detrimental impacts on caribou (Apps *et al.,* 2013; Whittington *et al.,* 2011). Care must also be taken that partial-cut harvesting intensity does not exceed the threshold at which a flush of early seral vegetation might occur (Frey *et al.,* 2003).

The design of partial-cut logging blocks must also include considerations of changes in road access which would be required and future use of these roads. At a small scale, within blocks, planned skid trails would optimally be winter access only, minimizing disturbance of ground cover and limiting future use. The design of access roads for hauling timber from partial-cutting harvest blocks raises further issues of predator access and changes in landscape patterns. Previous studies suggest that the impacts of linear features such as roads and seismic lines may be greatest in the late-winter period, when woodland caribou were found by Dyer *et al.* (2002) to cross active roads 6 times less frequently than simulated road networks. Dyer *et al.* (2001) also found significant avoidance effects in woodland caribou, up to 500 m, from roads and seismic lines, although these effects should decline with time as vegetation recolonizes deactivated roads and seismic lines. A strategy for road deactivation, road access restrictions, and restoration of road corridors, if required, is therefore a vital part of any partial-cutting harvest design in caribou winter habitat, in common with other linear disturbance features in caribou habitat.

Conclusion

The use of partial-cut logging to reduce canopy closure in mid- to late-seral lichen woodlands provides a short- to mid-term strategy for maintaining or even enhancing forage lichen availability for caribou. This strategy would best be considered in landscapes where there are few mid-seral stands that can replenish caribou habitat in coming decades. Under these circumstances, partial-cut harvesting may be used as a tool to maintain lichen forage availability in locations where it might be lost due to successional change. Eventually, however, these stands will require resetting by stand level disturbance factors such as wildfire to develop future lichen communities. Partial-cutting, therefore, cannot be used as a long-term substitute for natural disturbance dynamics that maintain a broad age-class distribution of stands after disturbances such as fire.

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Decision-support model to explore the feasibility of using translocation to restore a woodland caribou population in Pukaskwa National Park, Canada

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Abstract: The distribution and abundance of woodland caribou (*Rangifer tarandus caribou*) have declined dramatically in the past century. Without intervention the most southern population of caribou in eastern North America is expected to disappear within 20 years. Although translocations have reintroduced and reinforced some populations, approximately half of caribou translocation efforts fail. Translocations are resource intensive and risky, and multiple interrelated factors must be considered to assess their potential for success. Structured decision-making tools, such as Bayesian belief networks, provide objective methods to assess different wildlife management scenarios by identifying the key components and relationships in an ecosystem. They can also catalyze dialogue with stakeholders and provide a record of the complex thought processes used in reaching a decision. We developed a Bayesian belief network for a proposed translocation of woodland caribou into a national park on the northeastern coast of Lake Superior, Ontario, Canada. We tested scenarios with favourable (e.g., good physical condition of adult caribou) and unfavourable (e.g., high predator densities) conditions with low, medium, and high numbers of translocated caribou. Under the current conditions at Pukaskwa National Park, augmenting the caribou population is unlikely to recover the species unless wolf densities remain low (<5.5/1000 km^2) or if more than 300 animals could be translocated.

Key words: Bayesian belief network; decision-support; endangered species; expert opinion; process model; protected areas; reintroduction; species at risk; structured decision-making; threatened species; woodland caribou.

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Introduction

Boreal populations of woodland caribou (*Rangifer tarandus caribou*) (hereafter "woodland caribou") historically occupied the boreal forest across North America but are now extirpated from the southern limits of that range (Bergerud, 1974). Due to the declines in the distribution and abundance of this species, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and the Committee on the Status of Species at Risk in Ontario (COSSARO) assessed woodland caribou as Threatened (2000, 2005 and 2014, respectively). Between 1900 and 1950, boreal caribou retracted northward from Lake Superior (Cringan, 1957). They disappeared from the western shore of Lake Superior between 1905 and 1912 (Riis, 1938a, b, c, d) and were declining and scarce on the Sibley Peninsula by 1914 (Cringan, 1957). Three unconnected populations around northeastern Lake Superior persist as the species' most southern representatives in the eastern half of North America. These populations became disjunct from the northern herds in the 1950s or 1960s (Bergerud, 1988). Today, two populations are located on islands (Slates and Michipicoten); they were the products of translocations and are considered to be persistent with >200 individuals that fluctuate with the availability of vegetation (Environment Canada, 2012). The natural population on the mainland is now restricted to a narrow band along Lake Superior coast that includes Pukaskwa National Park (48°N, 85°W). Biennial surveys in the Park since the late 1970s have revealed a steady decline from 30 individuals to only 4 in 2009 (Bergerud *et al.,* 2007; Patterson *et al.,* 2014). With little to no recruitment for over a decade, Bergerud *et al.* (2007) suggested that extirpation is the likely outcome for this population by 2018. Parks Canada must decide between a costly intervention or risk extirpation of a species from a national park.

Translocation has been proposed to augment the population; however, translocations have mixed success as a management tool to recover caribou. Wildlife translocation is one of the more complex management actions used to restore or reinforce populations of species at risk (Decesare *et al.,* 2011). The long-term success of translocations requires managing the behaviour, habitat, metapopulation, and ecosystem level issues that initially led to the decline of the population (Armstrong & Seddon, 2008). Since 1982, the Ontario Ministry of Natural Resources and Forestry has restored or introduced woodland caribou from the Slate Islands to a number of islands and the shoreline of eastern Lake Superior with little success (G. Eason, personal communication; Gogan & Cochrane, 1994).

Failures of caribou translocation projects have been attributed to disease, predation, anthropogenic disturbance and/or insufficient and fragmented habitats (Bergerud & Mercer, 1989; Gogan & Cochrane, 1994; Compton *et al.,* 1995). In a review of 33 caribou introductions in eastern North America from 1924 to 1985, introductions inevitably failed when animals, released in proximity to white-tailed deer (*Odocoileus virginianus*), contracted meningeal brain worm (*Parelaphostrongylus tenuis*) and died (Bergerud & Mercer, 1989). For example, a herd of 51 caribou, released in Cape Breton Highlands National Park, Nova Scotia in 1968 and 1969, was extinct by 1973 due to meningeal brain worm (Dauphiné, 1975). Similar results occurred on Anticosti Island, Quebec (145 reindeer introduced in 1924), Great Cloche Island, Ontario (12 caribou released in 1970), and southern Wisconsin (14 caribou in an enclosure with white-tailed deer) (Bergerud & Mercer, 1989).

Predation was also a key factor in failed translocations. Wolf (*Canis lupus*), cougar (*Felis concolor*), and occasionally bear (*Ursus americanus*) predation were credited with the loss of translocated caribou in Ontario, Quebec, and British Columbia in Canada and Maine in the United States (Bergerud & Mercer, 1989; Gogan & Cochrane, 1994; Compton *et al.,* 1995). Cougar predation was the primary cause of death for 60 woodland caribou translocated from British Columbia to northern Idaho between 1987 and 1992 (Compton *et al.,* 1995). Wolf predation caused the failure of translocations in the Lake Superior region, Ontario, including the Gargantua Peninsula (39 caribou released in 1989) (Gogan & Cochrane, 1994) and Bowman Island (6 caribou released in 1985) (Bergerud & Mercer, 1989). Predation is also the primary limiting factor for almost all natural woodland caribou populations (McLoughlin *et al.,* 2003; Wittmer *et al.,* 2005; Festa-Bianchet *et al.,* 2011). Wolves and white-tailed deer are absent from Newfoundland, which has the highest rate of successful translocations (Bergerud & Mercer, 1989). From 1961 to 1982, 384 caribou were released at 22 sites and 17 of these releases were successful. The failures in Newfoundland were attributed to illegal hunting and anthropogenic disturbance (Bergerud & Mercer, 1989).

The failure of caribou translocations is consistent with reintroductions in general. An early review of reintroduction projects suggested that the majority failed to establish viable populations due to poor planning and insufficient consideration of the biological and ecological factors needed for success (Griffith *et al.,* 1989; Wolf *et al.,* 1998). A more recent review (1990-2005) of 454 projects found most reintroduction programs to be *ad hoc* rather than an organized attempt to assess risk, advance understanding in the field of reintroduction biology, or to improve reintroduction success (Seddon *et al.,* 2007). The authors described most research in the field of reintroduction biology to be retrospective, that is, opportunistic project evaluations and *post hoc* interpretation of monitoring (Seddon *et al.,* 2007). They recommended an increased role for formally planned projects that identify knowledge gaps and address uncertainty coupled with multidisciplinary teams of resource managers and scientists (Seddon *et al.,* 2007).

The planning, documenting, and decision-support for translocation is well served by structured decision analysis (Pérez *et al.,* 2012; Converse *et al.,* 2013). With such a tool, planners and advisors can explore the factors expected to influence the success of a caribou translocation and examine various combinations of environmental settings and introduction scenarios. Federal programs to recover species at risk also benefit from clear communication with stakeholders and the public. The framing of protection and recovery of species at risk is critical because it alters the way we think, talk, and approach the issue (Nie, 2001). Decision support tools are transparent, repeatable, and help conceptualize the key factors and their relationships – all of which facilitates framing and understanding the issue. It was under this premise that we developed a Bayesian belief network to explore the feasibility of a successful translocation of woodland caribou into Pukaskwa National Park.

Bayesian belief networks (BBNs) are graphical models that represent a set of variables linked by conditional probability relationships (McCann *et al.,* 2006; McNay *et al.,* 2006; Rumpff *et al.,* 2011; Conroy & Peterson, 2012). They facilitate communication at the interface of science, politics and community to enhance the decision making process (Reckhow, 1999). A BBN starts with an influence diagram, which is an intuitive graphical representation of the probabilistic dependence among variables (or nodes). In a BBN, a node leading to another one is a parent node, and the dependent node is a child node; the most external nodes (with no parent nodes) are used as the input to the model. Those diagrams are an effective method of modeling potential causal relationships/conditional dependencies (Reckhow, 1999).

Bayesian belief networks can also incorporate the uncertainty inherent in ecology. For example, experts may be uncertain about their own knowledge, there may be uncertainty inherent in the relationships being modeled (functional uncertainty), or uncertainty about the accuracy and or availability of information (epistemic uncertainty) (Kujala *et al.,* 2013). They are particularly useful for articulating the uncertainty that propagates between management actions (such as translocation) and eventual outcomes (such as species persistence).

Methods

Model development

We developed and quantified a BBN iteratively, with expert contribution and review at each stage, and used the freely downloaded software GeNie 2.0 (http://genie.sis.pitt.edu/). The initial graphical model was based on key variables and processes identified at a workshop with ten experts in caribou management, wolves and genetics, as well as regional biologists, local First Nations and park staff (Parks Canada, 2010). Next, experts crafted the "influence diagram", using as nodes the variables and processes identified at the workshop, and setting as input the parent nodes that describe the local environment as well as the variables that can be manipulated. That provided an intuitive presentation of the ecological relationships and a rapid scoping of the management issue (McCann *et al.,* 2006).

The influence diagram contributions were largely supported by scientific literature. Thresholds for each of the nodes are given in Appendix and include a citation when based on scientific literature. Where knowledge gaps existed, particularly with running scenarios specific to Pukaskwa National Park, we relied on expert opinion and identified predictions that could be tested in the event of a translocation. The influence diagram went through six major and several minor iterations before the team

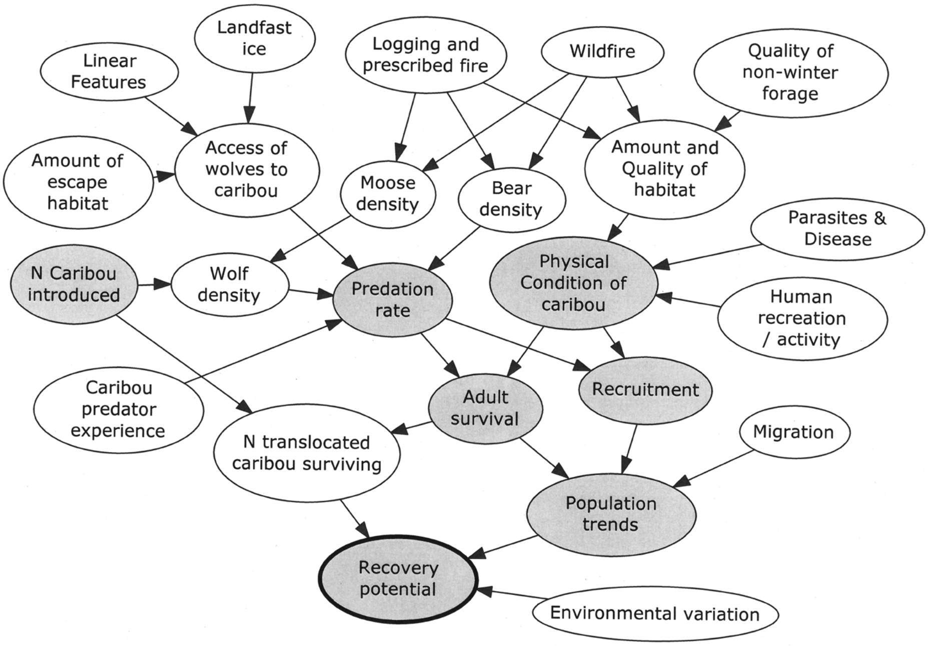

Figure 1. Influence diagram underlying a Bayesian Belief Network for a proposed woodland caribou translocation into Pukaskwa National Park. Grey shaded nodes are those presented in Table 1; the resultant (outcome) node has a thicker border.

reached a consensus (Fig. 1).

Whenever possible we were parsimonious with the model because the conditional probability table (CPT) of a child node becomes difficult to parameterize with increasing numbers of parent nodes. Also, the more links there are among nodes, the less tractable the model becomes (Marcot *et al.,* 2006). Parsimony was also appropriate given the degree of precision available for each node.

<u>Developing the influence diagram (Fig. 1)</u>

The general structure of the BBN is consistent with other efforts to identify key variables for caribou in Ontario (Rodgers *et al.,* 2008). Caribou declines are ultimately caused by habitat alterations and proximately caused by predation. More specific divisions can be traced back to these two broad effects (Festa-Bianchet *et al.,* 2011).

The most external parent nodes of the BBN, also called "input nodes" herein, are the key ecosystem variables and processes that affect caribou persistence and that either are determined by the local conditions or can be modified through management. These include descriptors of the caribou's environment, such as amount of escape habitat, the extent of linear features, and landfast ice, which all influence the access of wolves to caribou (Bergerud *et al.,* 2007). Other parent nodes include "logging and prescribed fire", "wildfire" and "quality of non-winter forage", which all influence moose and bear density and the amount and quality of habitat for caribou (Rodgers *et al.,* 2008; Environment Canada, 2012; Pinard *et al.,* 2012) (Fig. 1). For details on each node's states, thresholds used to separate states, and conditional probability values, see tables in the Appendix.

The child nodes are key variables and processes that influence population dynamics more or less directly, such as rate of predation and adult survival, which in turn are a main determinant of the population recovery potential. The rate of predation was primarily determined by the densities of wolves and bears (Ballard, 1994), and the accessibility of caribou to wolves, which are considered their most significant predator (Bergerud *et al.,* 2007). Predation rate is also likely to be affected by the experience the introduced animals have with predators (Frair *et al.,* 2007). If caribou translocated into Pukaskwa National Park were sourced from nearby predator-free islands, these individuals would be naïve and more susceptible to predators. Given that the experience of translocated animals with predators could affect their persistence, predator-experienced vs. predator-naive caribou was a factor included as a parent (input) node in this model.

Wolf density is in turn affected by the density of their main prey species, which could be moose or caribou depending on their relative availability (Bergerud & Elliott, 1986). For the period 1974–1988, the dynamics at Pukaskwa National Park suggested that wolf predation depended on caribou density (Bergerud, 1996). Caribou recruitment declined and adult mortality increased when wolf numbers increased beyond 20 individuals (Bergerud, 1996). Predation dynamics can partly offset the effect that a larger initial population of caribou would have on recovery potential. This is why the model includes the intermediate child node "number of translocated caribou surviving" between the nodes "number of caribou introduced" and "recovery potential" (Fig. 1). The number of surviving animals (over ~ 5 yrs) is modulated by the survival rate of adults, and therefore links the short-term dynamics to the longer-term projection.

Vors and Boyce (2009) reviewed a variety of potential responses by caribou to climate change, such as indirect, density-independent effects of extreme weather events that cause unpredictable access to forage, or freezing rain events that eliminate access to grazing due to an impenetrable layer of ice. Therefore, we included physical condition (or body mass) as a qualitative biological integrator of key environmental variables: amount and quality of habitat, parasites and diseases, and human disturbances through recreation and resource extraction. Those key environmental variables are determined by the local conditions and/or can be altered through management, so they are set as external parent nodes in the model.

The physical condition of caribou has implications for determining adult survival and recruitment, as relationships between body mass and survival and fertility have shown (Taillon *et al.,* 2012). Caribou may skip reproduction if they are in poor physical condition due to insufficient food resources (Bergerud *et al.,* 2007; NCASI, 2007; Taillon *et al.,* 2012). Caribou are also susceptible to anthropogenic disturbances; they avoid resorts and recreation activities (Nellemann *et al.,* 2000; Carr *et al.,* 2011), active logging (Schaefer & Mahoney, 2007), and are subject to increased bear predation near campsites (Pitt & Jordan, 1996). In Pukaskwa National Park, human recreational activities could include tourists on foot and in boats around islands and coastlines.

Timber volumes harvested in Ontario over the last decade have declined by more than 40%, including from lands adjacent to the park (Ontario Ministry of Natural Resources, 2012). Although wildfire and prescribed fires are permitted in some circumstances in the park, the fire cycle has departed significantly from what it would have been without human influence, and as a result, an older-than-usual forested landscape prevails. Fires are infrequent (Perera & Baldwin, 2000) and typically smaller in size along the coast (C. C. Drake, unpublished data), which is largely believed to be beneficial for caribou (Environment Canada, 2012). Fire improves habitat for moose, which attracts predators. The predators consume moose but also caribou, when they encounter them (Bergerud *et al.,* 2007). These factors were included in the model, incorporating the circumstances more to less favourable for caribou.

Presently, disease is not considered the primary limiting factor in the Lake Superior range mainly because white-tailed deer, the vectors of brain worm, which is lethal to caribou (Anderson & Strelive, 1968), were not historically abundant (Whitlaw & Lankester, 1994). Nonetheless, we included disease as an input node in the model because white-tailed deer are expanding their distribution (Thompson *et al.,* 1998) and have been increasingly detected in Pukaskwa National Park (C. C. D., unpublished data).

The terminal child node of the model is the recovery potential. It is defined as the long-term probability of persistence of the population (i.e., whether a population will be self-sustaining). As such, the node has as parent nodes the population trends, the environmental variation (which drives the random variation in population trends), and the number of translocated caribou surviving. A high recovery potential could be defined as a time to extinction longer than 50 yrs, or a 95% chance of persistence over the next 50 yrs. Although the time scale of the processes included in the model is short-term (~5 yrs), the end result is a projection into the future. When the result of a BBN scenario is a high probability for “high recovery potential”, it suggests that this scenario will produce a successful translocation.

Other factors that might be relevant for other caribou populations, such as predation by felids (Compton *et al.,* 1995), vehicle collisions, or avalanches (Hebblewhite *et al.,* 2007), were not relevant at Pukaskwa National Park. Genetic diversity was not included in the model because, although it is lower in isolated populations, there is no immediate concern for conservation (Courtois *et al.,* 2003; McLoughlin *et al.,* 2004) nor did participants at the 2010 caribou workshop feel this was a significant factor in the success of a translocation (Parks Canada, 2011).

Parameterizing the model

The links among the model's nodes reflect the knowledge we have about the probable influence that a given parent node has on one or more child nodes. These links are assumed to be causal. All the links in this BBN are through CPTs, which we conceived as contingency tables. For example, the probabilities of a population decline were determined by the number of observed cases in which a decline was observed under each combination of two states of adult survival, recruitment, and three states of migration (positive, negligible, negative).

For the node "Population trends", we used data from population surveys and modeling, categorized each case, and compiled a contingency table (Appendix). For all other child nodes, data were less available so we first asked experts to determine what threshold values could be used to tell each state apart. Wherever possible, these thresholds were drawn first from the literature. We then asked the experts to consider how nodes would interact so that we could parameterize the CPTs. For example, we asked, "among all the possible cases where number of caribou introduced were high, in how many cases would the wolf density have remained low?". Experts were asked to consider the breadth of the caribou literature, not specifically caribou in Pukaskwa National Park. Experts were also invited to review each other's assessments. Most often there was consensus or suggestions for additions, fine-tuning of the model, or increased precision in a threshold based on a new literature reference.

Exploring scenarios

To explore the properties of the model and to apply it specifically to caribou translocation at Pukaskwa National Park, we set evidence in all the most external parent nodes according to these 10 scenarios: least favourable vs. most favourable environmental conditions with two levels of translocation effort (4 scenarios), current conditions at Pukaskwa National Park with three levels of translocation effort (3 scenarios), and current conditions at Pukaskwa National Park with low wolf densities with three levels of translocation effort (3 scenarios) (Table 1).

The decline in logging, less frequent wildfire, combined with limited prescribed fire in the park over the last decade (Kuchta, 2012), has created older growth forests adjacent to and within the park that are favourable to caribou. Therefore, the probability of limited logging and prescribed fire and wildfire was set at 100%. Terrestrial lichen, a year-round food source for caribou (NCASI, 2007), is abundant at Pukaskwa National Park but entirely absent on Michipicoten Island (Bergerud *et al.,* 2007) where caribou numbers are high. Therefore the probability of good "quality of non-winter forage" was set at 100%.

Several nodes have high levels of uncertainty or show important variation among years, so virtual evidence was used as input for those nodes. Parasites or disease being transmitted by deer is unlikely to seriously threaten the physical condition of caribou in the near future because of the current low density of deer in the park and surrounding landscape, but the situation could change rapidly. Therefore, the probability of low "parasites and diseases" was set at 90%.

In Alberta, human activities alter caribou behavior and mediate the effects of wolf predation on caribou (Hebblewhite *et al.,* 2005; Wasser *et al.,* 2011). However, Pukaskwa National Park has low human use at sensitive times (calving/rutting), so the probability of low "human recreation/activity" was set to 80%.

In the Lake Superior range, caribou remain vulnerable because escape habitat is limited and, importantly, habitat in their range has been altered by human disturbance (Vors *et al.,* 2007). Near-shore islands may serve as a primary escape habitat from predators (Ferguson *et al.,* 1988; Carr *et al.,* 2011) and limited linear features likely keeps predator access low in the area (Bergerud, 1985). Trends toward warmer winters resulting in less landfast ice may have further limited the access of wolves to caribou in the coastal region (Thompson *et al.,* 1998). To take into account the variation and uncertainty in those factors, the probability of plenty vs. little for the node "amount of escape habitat" was set to 20:80; the probability of limited "linear features" was set to 90% and to 50% for limited "landfast ice". This set of values gives a probability of low "access of wolf to caribou" of about 50% (Appendix).

Based on population size time series, environmental variation (i.e., the long-term yearly random fluctuation in population growth rate due to variation in survival, recruitment, and migration), remains low; therefore, the probability of low "environmental variation" was set at 80%.

Once the values for the input nodes were set, we examined how the probability of recovery potential would increase following the introduction of an increasing number of caribou: less than 50, 50–300, and >300. These values were drawn from a non-spatial population viability analysis which concluded that a population of 300 animals with moderate calf and adult female survival had a 10% probability of quasi-extinction, and that large populations (≥ 300) had a high probability of persistence under favourable demographic conditions (Environment Canada, 2012). It could be argued that introducing such large numbers of animals is unrealistic, but one has to consider the (conceptual) 5 year time frame of the model, which would allow for a lower number of animals to be introduced annually over 5–10 yrs rather than all at once during a one-time translocation event. We also assumed that caribou that were "available" for a translocation into Pukaskwa National Park would originate from islands where caribou are abundant, such as the nearby Slate and Michipicoten Islands and many of those naïve individuals would be lost annually.

Results

The least favourable scenario produced only a 1% probability of population recovery (Table 1). The most favourable scenario resulted in 58% probability of population recovery when fewer than 50 animals were translocated and 90% when more than 50 animals were translocated (Table 1).

Under current conditions in Pukaskwa National Park, the chance of high recovery potential increased with the number of translocated animals to a high of 46% (Table 1). When we set the probability of high "wolf density" to 100%, regardless of its parent nodes, the probability of high predation rate reached 72%. This combination of inputs suggests that even introducing 300 caribou would not increase the probability of population recovery beyond 50% (Table 1). With the same set of evidence, but with probability of low wolf density set at 100%, introducing more than 50 caribou raised the probability of population recovery above 50% (Table 1).

Discussion

Interestingly, the probability of high recovery potential under the current conditions, and with even a large translocation effort, are roughly consistent with the 50% failure rate of caribou translocations in eastern North America (Bergerud & Mercer, 1989; Gogan & Cochrane, 1994) as well as estimates of translocation success in western North America (Decesare *et al.,* 2011). However, the mechanisms leading to that result vary from one application to another, so we cannot claim that our model emulates or explains the more general result of many historic translocations.

Table 1. Probability of recovery potential (%) under different model scenarios and number of caribou introduced. Percent probability of five child nodes are also presented.

Model scenario	**# of caribou introduced**	**Predation rate**	**Physical condition of caribou**	**Adult survival**	**Recruit-ment**	**Population trends**	**Recovery Potential**
		Low/High	Good/Bad	Low/High	Low/High	Decline/Stab-le/Increase	High/Low
Least favourable[1]	<50	0/100	0/100	90/10	100/0	96/2/2	1/99
Most favourable[2]	<50	99/1	100/0	1/99	1/99	0/25/75	58/42
Most favourable[2]	50-300	99/1	100/0	1/99	1/99	0/25/75	90/10
Current[3]	<50	30/70	91/9	38/62	72/28	47/32/21	21/79
Current[3]	50-300	30/70	91/9	38/62	72/28	47/32/21	38/62
Current[3]	>300	30/70	91/9	38/62	72/28	47/32/21	46/54
Low wolf density[4]	<50	87/13	91/9	7/93	17/83	29/28/43	35/65
Low wolf density[4]	50-300	87/13	91/9	7/93	17/83	29/28/43	58/42
Low wolf density[4]	>300	87/13	91/9	7/93	17/83	29/28/43	67/33

[1] Input nodes adjusted to the least favourable environmental conditions or worst case scenario
[2] Input nodes adjusted to the most favourable environmental conditions or best case scenario
[3] Input nodes adjusted to reflect the current conditions at Pukawska National Park, based on best available information. For those scenarios, wolf density node is input as 100:0 high, regardless of the value of its parent nodes.
[4] A hypothetical scenario with same input as current, but in which wolf density node is set at 0:100 low, regardless of the value of its parent nodes.

The differences among the probabilities of high recovery potential for the most favourable scenario (90%) and the current conditions (38%) at Pukaskwa National Park suggest that the translocation of caribou into Pukaskwa National Park would be highly risky unless some of the unfavourable conditions were altered. Although reducing predation would increase the probability of recovery potential by 12–21%, this increase may be insufficient to warrant a potentially unpopular and ecologically harmful management option such as predator control, particularly in a national park. Alternatively, Parks Canada could try managing the predation rate on caribou indirectly. For example, the park could manage habitat to reduce alternate prey (moose and deer) that attract predators, improve escape habitat, limit linear features that facilitate access of wolves to caribou, and provide safe sites for caribou to calve.

Typical of species at risk, elements of uncertainty remain that affect recovery potential. The probability of recovery and persistence of translocated caribou in Pukaskwa National Park hinges on key uncertainties such as the risk of parasites and disease, human disturbance, and the ability of predator-naïve caribou successfully eluding predators. The complexity of the relationships among the nodes of this BBN coupled with knowledge gaps highlights the importance of uncertainty. Complexity and uncertainty are "familiars" in ecology; the advantage of a BBN over *ad hoc* decision-making is that it identifies and prioritizes research needs. The parts of our BBN that are based mainly on expert experience can be used to generate testable hypotheses and can be advanced with iterative testing and updating of the model (Marcot *et al.,* 2006; Martin *et al.,* 2012).

Our BBN is a representation of a collectively agreed upon reality as opposed to a test of causal relationships. We could not formally estimate the predictive accuracy of the model since observation data are unavailable to compare predictions with observations. This may be an unsatisfying outcome for those who value the precision of quantitative models; as data become available, this model can certainly be improved. However, a network of variables with numerical probabilities is not an intuitive way to interpret results for all stakeholders (Renooij & Witteman, 1999). Eliciting expert input for BBNs requires experts to express their beliefs in probabilistic terms that describe dependencies among different factors. It has been argued that inferential reasoning is the mechanism by which people integrate and interpret subjective and incomplete data from various sources (Pearl, 1988). Some of our experts did not feel familiar enough with the concept of probability or they felt it was too difficult to quantify their beliefs. As a result, the probabilities of the outcomes in this BBN are generally described in a relative sense. The model's precision could be improved in the future; presently, it is consistent with available data and the level of uncertainty of the experts.

The translocation of caribou is logistically difficult and expensive to implement. Recovery of caribou requires public funds and so it is important to have local support for caribou translocation programs (Schneider *et al.,* 2010). In this area, the majority of regional residents support conservation actions for caribou in Pukaskwa National Park, however, only 51% would support translocation (Parks Canada, 2011). The lack of strong support may be driven, in part, by local hunters. Over the past century, caribou have declined and moose have increased and local hunters in this region have shifted their harvest to moose. Hunters are aware that managing for caribou habitat does not favour moose habitat, which could result in lower moose densities and fewer moose tags (C. C. Drake, personal observation). The social challenges of translocations can be even more daunting than the biological ones (Reading & Clark, 1996), and successful programs benefit from approaches that integrate the social and biological sciences (Miller *et al.,* 1999). BBNs are well-suited to incorporating social and economic analyses by including model nodes for costs and utilities (Levontin *et al.,* 2011; Haines-Young, 2011). A future application for this caribou BBN could include the addition of socio-economic factors.

Conclusion

Species at risk of extirpation or extinction present unique challenges to land managers given their paucity coupled with political scrutiny and economic realities (Armstrong & McCarthy, 2007). It is often necessary to make decisions for species at risk under considerable uncertainty (i.e., limited demographic data and lack of information on dispersal (Beissinger & Westphal, 1998) and failing to acknowledge or address uncertainty can lead to poor decisions and outcomes (Regan *et al.,* 2005). Despite the *ad hoc* nature of these projects, programs to recover endangered species are expected to maximize species survival and minimize financial cost, while under the scrutiny of stakeholders and jurisdictions with divergent opinions (Maguire, 1986). We presented a BBN for a potential caribou translocation in Pukaskwa National Park to provide structured decision support for resource managers.

This BBN suggests that any size of translocation is unlikely to help recover the population of caribou in Pukaskwa National Park under the current conditions. Although the long-term recovery and persistence of an augmented population of caribou in Pukaskwa National Park is unknown, most of the short-term scenarios explored in the BBN resulted in low to moderate success, which suggests that long-term recovery and persistence may be unlikely either with or without translocation. Importantly, long-term recovery and survival of caribou may be hampered by the lack of contiguity with more northern populations and habitat conditions beyond the boundaries of Pukaskwa National Park.

Although this BBN was developed for Pukaskwa National Park's proposed translocation, we also made it flexible enough to be applied to other caribou populations. It represents and combines empirical data with experts' understanding of caribou ecology. It graphically expresses complex relationships and challenges for caribou recovery and management. It addresses, in a structured way, uncertainties that plague attempts to solve these problems. It evaluates alternative decisions within a context of risk assessment to help identify options with caribou translocation. It also fosters communication among ecologists, decision-makers, and stakeholders who may lack common training, terminology, or experience (Cain, 2001).

Regardless of whether the caribou population in Pukaskwa National Park is augmented through translocation, it is apparent that the factors driving the decline of caribou and the fate of their recovery in this region will not be easily resolved. On-going development of this BBN based on empirical data, as it becomes available, could be an important tool in facilitating the decision-making process for caribou management in Pukaskwa National Park and more broadly, as many caribou populations in Canada are declining (Environment Canada, 2012).

This model was developed using the freely downloaded software GeNie 2.0 (http://genie.sis.pitt.edu/). We invite readers to explore their own scenarios. Our inputs are available in Appendix. Contact the authors to request the model.

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Appendix. Conditional probability tables (Tables A1-A10).

Table A1. Conditional probability table for node recovery potential.

Parent nodes and their state			Recovery potential[1]	
Number of translocated caribou surviving	Environmental variation	Population trends	High	Low
Large	Low	Decline	0.25	0.75
Large	Low	Stable	0.75	0.25
Large	Low	Increase	1.00	0.00
Large	High	Decline	0.00	1.00
Large	High	Stable	0.50	0.50
Large	High	Increase	1.00	0.00
Medium	Low	Decline	0.00	1.00
Medium	Low	Stable	0.75	0.25
Medium	Low	Increase	1.00	0.00
Medium	High	Decline	0.00	1.00
Medium	High	Stable	0.50	0.50
Medium	High	Decline	0.75	0.25
Small	Low	Decline	0.00	1.00
Small	Low	Stable	0.25	0.75
Small	Low	Increase	0.70	0.30
Small	High	Decline	0.00	1.00
Small	High	Stable	0.10	0.90
Small	High	Increase	0.50	0.50

[1] Thresholds for recovery potential:
Low Probability of extinction >5% over 50 yrs OR: time to extinction <= 20 yrs
High Probability of extinction <5% over 50 yrs OR: time to extinction > 20 yrs

Table A2. Conditional probability table for node population trends.

Parent nodes and their state			Population Trend		
Migration[1]	Survival[1,2]	Recruitment[3]	Decline	Stable	Increase
Negligible	Low	Low	0.875	0.125	0.000
Negligible	Low	High	0.571	0.286	0.143
Negligible	High	Low	0.200	0.600	0.200
Negligible	High	High	0.250	0.250	0.500
Positive	Low	Low	0.000	0.875	0.125
Positive	Low	High	0.200	0.500	0.300
Positive	High	Low	0.100	0.600	0.300
Positive	High	High	0.000	0.250	0.750
Negative	Low	Low	1.00	0.000	0.000
Negative	Low	High	0.800	0.200	0.000
Negative	High	Low	0.600	0.200	0.200
Negative	High	High	0.500	0.250	0.250

[1] States for migration: Positive: immigration accounts for >10% of the population size over 5 years; Negative: emigration accounts for >10% of the population size over 5 years; Negligible: migration is less than or equal to 10% over 5 years.
[2] Survival (annual rate): Low: S < 0.88; High: S >= 0.88.
[3] Recruitment (calf:adult ratio): Low: R < 0.105; High: R >= 0.105.

Table A3. Conditional probability table for nodes adult survival and recruitment.

Parent nodes and their state		Adult Survival		Recruitment	
Predation Rate	Physical Condition	Low	High	Low	High
Low	Good	0.0	1.0	0.0	1.0
Low	Bad	0.1	0.9	0.5	0.5
High	Good	0.9	0.1	1.0	0.0
High	Bad	0.9	0.1	1.0	0.0

Note: Probability values assume that predation affects recruitment much more than it affects survival of adults.

Table A4. Conditional probability table for node translocated caribou surviving.

Parent nodes and their state		Translocated caribou surviving		
Adult Survival	N caribou introduced[1]	Large	Medium	Small
Low	Large	0.0	1.0	0.0
Low	Medium	0.0	0.0	1.0
Low	Small	0.0	0.0	1.0
High	Large	0.9	0.1	0.0
High	Medium	0.0	0.9	0.1
High	Small	0.0	0.0	1.0

[1] N caribou introduced: Small <= 50 animals, Medium 50-300 animals, Large >300 animals.

Table A5. Conditional probability table for node predation rate.

Parent nodes and their state				Predation Rate	
Caribou predator experience	Wolf Density	Access of wolf to caribou	Bear Density	Low	High
Yes	Low	Low	Low	1.0	0.0
Yes	Low	Low	High	0.9	0.1
Yes	Low	High	Low	1.0	0.0
Yes	Low	High	High	0.9	0.1
Yes	High	Low	Low	0.9	0.1
Yes	High	Low	High	0.8	0.2
Yes	High	High	Low	0.4	0.6
Yes	High	High	High	0.3	0.7
No	Low	Low	Low	0.9	0.1
No	Low	Low	High	0.75	0.25
No	Low	High	Low	0.9	0.1
No	Low	High	High	0.2	0.8
No	High	Low	Low	0.5	0.5
No	High	Low	High	0.25	0.75
No	High	High	Low	0.0	1.0
No	High	High	High	0.0	1.0

Table A6. Conditional probability table for node physical condition of caribou.

Parent nodes and their state			Physical Condition	
Parasites & diseases[1]	Amount & quality of habitat	Human recreation/activity	Good	Bad
Low	High	Low	1.0	0.0
Low	High	High	0.9	0.1
Low	Low	Low	0.2	0.8
Low	Low	High	0.1	0.9
High	High	Low	0.3	0.7
High	High	High	0.1	0.9
High	Low	Low	0.0	1.0
High	Low	High	0.0	1.0

[1] Thresholds for parasite and diseases: based on deer density: Low: < 6 deer/km^2; High: > 6 deer/km^2 (Bergerud & Mercer, 1989).

Table A7. Conditional probability table for node wolf density.

Parent nodes and their state		Wolf density[1]	
Moose density	N of caribou intro-duced	Low	High
Low	Large	0.75	0.25
Low	Medium	0.9	0.1
Low	Small	1.0	0.0
High	Large	0.0	1.0
High	Medium	0.25	0.75
High	Small	0.5	0.5

[1]Low: <5.5/1000 km^2; High >=5.5/1000 km^2

Bergerud and Mercer (1989) have suggested that even in the absence of deer (the source for *P. tenuis*) when wolf densities exceed 10/1,000 km^2, caribou re-introductions will fail. Bergerud and Elliot (1986) indicated that generally, in the absence of escape habitat, caribou populations cannot maintain their numbers when wolf densities are >=6.5/1,000 km^2.

Table A8. Conditional probability table for nodes moose density and bear density.

Parent nodes and their state		Moose Density[1]		Bear Density[2]	
Logging & prescribed fire	Wildfire	Low	High	Low	High
Limited	Limited	0.90	0.10	0.90	0.10
Limited	Extensive	0.75	0.25	0.75	0.25
Extensive	Limited	0.40	0.60	0.50	0.50
Extensive	Extensive	0.20	0.80	0.20.	0.80

[1] Thresholds for moose density: Low : <0.3 moose/km^2; High = >0.3 moose/km^2

[2] Thresholds for bear density: Low: <10/100 km^2; High = >10/100 km^2

Table A9. Conditional probability table for node access of wolves (to caribou).

Parent nodes and their state			Access of wolves	
Amount of escape habitat	Linear features	Landfast ice	Good	Bad
Plenty	Limited	Limited	1.0	0.0
Plenty	Limited	Extensive	0.8	0.2
Plenty	Extensive	Limited	0.7	0.3
Plenty	Extensive	Extensive	0.5	0.5
Little	Limited	Limited	0.5	0.5
Little	Limited	Extensive	0.4	0.6
Little	Extensive	Limited	0.3	0.7
Little	Extensive	Extensive	0.0	1.0

Table A10. Conditional probability table for node amount and quality of habitat.

Parent nodes and their state			Amount & quality of habitat	
Quality of non-winter forage	Logging & prescribed fire[1]	Wildfire[1]	High	Low
Good	Limited	Limited	1.0	0.0
Good	Limited	Extensive	0.2	0.8
Good	Extensive	Limited	0.2	0.8
Good	Extensive	Extensive	0.1	0.9
Poor	Limited	Limited	0.5	0.5
Poor	Limited	Extensive	0.0	1.0
Poor	Extensive	Limited	0.0	1.0
Poor	Extensive	Extensive	0.0	1.0

[1] Logging and prescribed fire node, and for Wildfire node, the threshold for limited *vs.* extensive is 40% of the range. When total disturbance exceeds 40% of the range, the probability that a Woodland Caribou population would be stable or increasing drops below 0.5 (Environment Canada, 2012).

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Conservation status of caribou in the western mountains of Canada: Protections under the Species At Risk Act, 2002-2014

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Abstract: In April 2014, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) reviewed the status of caribou in the western mountains of Canada, in keeping with the ten-year reassessment mandate under the Species at Risk Act. Assessed as two 'nationally significant' populations in 2002, COSEWIC revised the conservation units for all caribou in Canada, recognising eleven extant Designatable Units (DUs), three of which -- Northern Mountain, Central Mountain, and Southern Mountain -- are found only in western Canada. The 2014 assessment concluded that the condition of many subpopulations in all three DUs had deteriorated. As a result of small and declining population sizes, the Central Mountain and Southern Mountain DUs are now recognised as endangered. Recent declines in a number of Northern Mountain DU subpopulations did not meet thresholds for endangered or threatened, and were assessed as of special concern. Since the passage of the federal Species at Risk Act in 2002, considerable areas of habitat have been managed or conserved for caribou, although disturbance from cumulative human development activities has increased during the same period. Government agencies and local First Nations are attempting to arrest the steep decline of some subpopulations by using predator control, maternal penning, population augmentation, and captive breeding. Based on declines, future developments and current recovery effects, we offer the following recommendations: 1) where recovery actions are necessary, commit to simultaneously reducing human intrusion into caribou ranges, restoring habitat over the long term, and conducting short-term predator control, 2) carefully consider COSEWIC's new DU structure for management and recovery actions, especially regarding translocations, 3) carry out regular surveys to monitor the condition of Northern Mountain caribou subpopulations and immediately implement preventative measures where necessary, and 4) undertake a proactive, planned approach coordinated across jurisdictions to conserve landscape processes important to caribou conservation.

Key words: Central Mountain; COSEWIC; Designatable Units; Northern Mountain; *Rangifer tarandus*; Southern Mountain; Species At Risk Act.

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Introduction

When Canada's Species At Risk Act (SARA) came into force in 2003, the legal list (SARA, 2002) comprised 233 wildlife species (as defined under the Act) in Schedule 1. Among these were Woodland Caribou (*Rangifer tarandus caribou*) residing in the western mountains of Canada, which the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) considered as two "nationally significant" Southern Mountain and Northern Mountain populations (COSEWIC, 2002). Ranging from southern British Columbia and Alberta to Yukon and the Northwest Territories, caribou historically had a relatively widespread distribution and occurred in large (>1,000 individuals) subpopulations (Spalding, 2000). By 2000, about 30% of their early 1900s range was no longer occupied (Figure 1; Spalding, 2000; Dzus, 2001). In 2002, COSEWIC assessed the Southern Mountain population as threatened and the Northern Mountain population as of special concern (COSEWIC, 2002) and they were listed on the SARA registry the next year (Government of Canada, 2014). Subpopulations comprising the Southern Mountain population were generally small in size, increasingly isolated from one another, and subject to threats, with the majority in decline (COSEWIC, 2002). Although numbers of Northern Mountain caribou appeared to be stable, forestry, roads, gas, and other developments were beginning to affect some subpopulations through habitat modification and increasing human access (COSEWIC, 2002).

Various recovery planning and actions directed at these populations since listing under SARA have been undertaken by provinces and territories. For example, both Alberta and British Columbia have released strategic recovery documents that suggest a variety of different actions aimed at recovering subpopulations in southern and central portions of the provinces (e.g., MCTAC, 2002; Alberta Woodland Caribou Recovery Team, 2005; ASRD & ACA, 2010; Parks Canada, 2011a; Mountain Caribou Recovery Implementation Plan Progress Board, 2012). In the past decade, management plans or recommendations have also been developed for individual subpopulations or subpopulation groups (e.g., Chisana Caribou Herd Working Group, 2012; BC Ministry of Environment, 2013). Under SARA, a Management Plan for caribou in the Northern Mountain population (Environment Canada, 2012), and a Recovery Strategy for the Southern Mountain population (Environment Canada, 2014) were both released. Targeted measures, including habitat and population management and protection, have also been implemented under the authority of various provincial legislation and policies (COSEWIC, 2014a).

First created in 1977, COSEWIC was formally established under SARA (SARA, 2002, s. 14), with the functions of conducting assessments, reassessments, and classifications of species at risk "on the basis of the best available information on the biological status of a species, including scientific knowledge, community knowledge and aboriginal traditional knowledge" (SARA, 2002, s. 15). For each species, relevant information is assembled in a status report, which is subjected to an extensive expert review process (COSEWIC, 2011b). Each species is assessed according to criteria based on the IUCN Red List system to measure the likelihood of species going extinct under prevailing circumstances (Mace *et al.,* 2008). Under SARA, the government of Canada considers COSEWIC's designations within designated timeframes when establishing the legal list of wildlife species at risk (COSEWIC, 2014b).

In April 2014, COSEWIC reviewed the conservation status of caribou in the western mountains of Canada (COSEWIC, 2014a), in keeping with the 10-year reassessment mandate under SARA (SARA, 2002, s. 24). This reassessment benefited from an acceleration

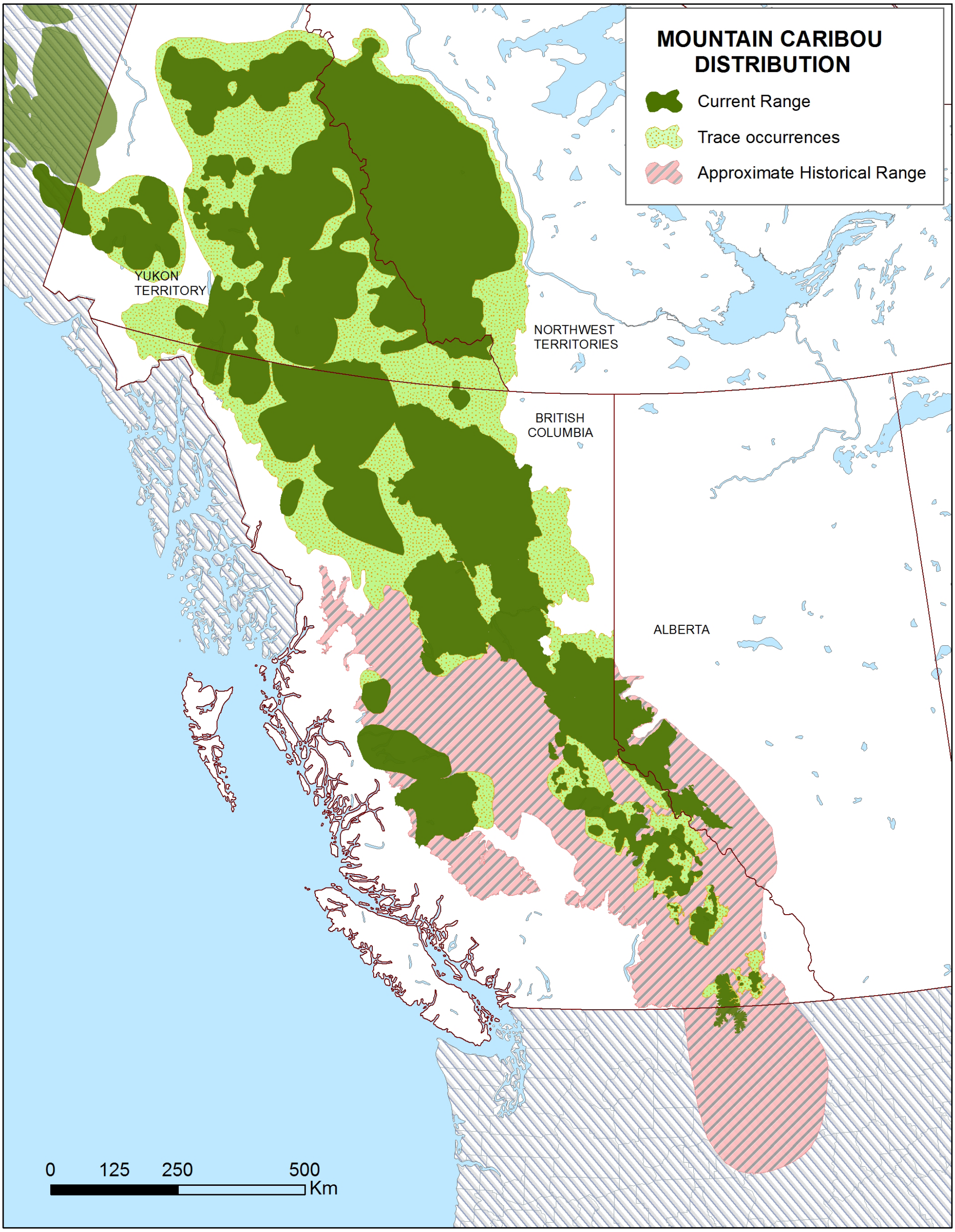

Figure 1. Approximate historic and current ranges of caribou in the mountain DUs of western Canada (from COSEWIC 2014a). Map created by Bonnie Fournier (Environment and Natural Resources, Government of the Northwest Territories, 2013).

of research and monitoring over the past decade that yielded new information on population trends and further insights into threats. Moreover, it took advantage of the recognition of new conservation units for *Rangifer* found across Canada, a special project undertaken by COSEWIC to define discrete and evolutionarily unique "Designatable Units" (COSEWIC, 2011a) for caribou throughout the country. This work used available information to derive conservation units of the species to orient future COSEWIC status assessments and reassessments, thereby addressing widely accepted deficiencies in the current taxonomy (see the 'Classification of Caribou' below). Aboriginal knowledge was also collected and summarized from First Nations and Métis sources by the COSEWIC Aboriginal Traditional Knowledge (ATK) Subcommittee (COSEWIC, 2014a).

The recent reassessment of caribou in the western mountains of Canada provided an opportunity to evaluate how subpopulations comprising these newly recognised Designatable Units have fared since the implementation of SARA just over a decade ago. Our objectives here are to review: 1) the Designatable Unit structure for western mountain caribou in Canada, 2) the 2014 COSEWIC assessments of these units, including population numbers and trends that served as their basis, and 3) the recovery and management actions planned and implemented to date. We conclude with a forward-looking perspective on the conservation outlook for these populations.

Taxonomy and conservation units of western mountain caribou

Prevailing taxonomy (Banfield, 1961) recognizes four native extant and one extinct caribou subspecies in North America, based primarily on skull measurements and pelage, but also antler shape and hoof shape. It is widely considered to be outdated and insufficient for capturing the variability of caribou across their range in Canada (Geist, 2007; Gunn, 2009; Couturier *et al.,* 2009; COSEWIC, 2011a), but is still the most commonly used taxonomy because some aspects do appear to have validity and no alternative has been identified in a systematic manner (COSEWIC, 2011a). Previous COSEWIC evaluations used Banfield's (1961) subspecies as the basis for assessment. Caribou in western mountain regions of North America were included in woodland subspecies, but the nationally significant populations (Northern Mountain and Southern Mountain) were further divided into two western mountain caribou ecotypes based on COSEWIC's National Ecological Areas with the same names (COSEWIC, 2002).

The widely-recognized shortcomings of caribou taxonomy have triggered a reliance on ecotypes, based on behaviour and ecology, for conservation and management purposes. In a broad sense, woodland caribou in North America are informally recognised as 'mountain' or 'boreal' with the designation distinguishing between those subpopulations that exhibit seasonal or annual use of mountainous terrain vs. lowland boreal habitats (Festa-Bianchet *et al.,* 2011). In western Canada, this nomenclature largely coincides with the COSEWIC Southern, Central, and Northern Mountain DUs (mountain caribou) considered here, and the Boreal DU (boreal caribou). Caribou subpopulations in BC are classified by the Province into three formally-designated ecotypes according to behaviour and habitat use, with mountain subpopulations belonging to 'Northern' or 'Mountain', and the remainder as 'Boreal' (Government of British Columbia, 2014). The BC Northern ecotype corresponds with the Northern and Central Mountain DUs and the Mountain with the Southern Mountain DU (Stevenson & Hatler, 1985; Heard & Vagt, 1998). Similarly, 'mountain' caribou in Alberta are distinguished from their 'boreal' counterparts by feeding primarily on terrestrial lichens and spend at least part of their annual cycle in the mountains (ASRD & ACA, 2010).

COSEWIC Designatable Units

SARA recognizes that entities below the species level require conservation, and provides COSEWIC with the mandate to assess them (SARA, 2002, s. 15). Accordingly, COSEWIC's DU concept (formalized in 2009) acknowledges that there are spatially, ecologically, or genetically discrete and evolutionarily significant units that are irreplaceable components of biodiversity (COSEWIC, 2011c). Discreteness may refer to distinctiveness in genetic characteristics or inherited traits, habitat discontinuity, or ecological isolation. Significance is also included in the definition of DU as a reflection of the opinion that isolation alone is insufficient for designation. Evolutionary significance may apply when there is: 1) deep phylogenetic divergence (e.g., glacial races), 2) evidence that the population persists in a unique ecological setting that has likely given rise to local adaptations, especially those related to fitness, or 3) where there is only one natural surviving occurrence in a particular ecological setting.

In previous COSEWIC assessments (COSEWIC, 2002; 2004) prior to the passage of SARA and use of Designatable Units, caribou in Canada were organized into eight "Nationally Significant Populations", not including the barren-ground subpopulations, which have not been assessed (Festa-Bianchet *et al.,* 2011; COSEWIC, 2011a). In preparation for national-scale assessments and reassessments of this wildlife species initiated in 2012, COSEWIC undertook a 2-year exercise to evaluate DUs for caribou in Canada using the new DU guidelines (COSEWIC, 2011a). The process considered established taxonomy, phylogenetics, genetics, morphology, life history, ecology, and behaviour of the species, as well as biogeographical information such as range disjunction and the eco-geography in which the species is found. Using COSEWIC DU criteria for discreteness and significance (COSEWIC, 2011c), western mountain caribou were separated into three units: Northern Mountain caribou of Yukon, Northwest Territories and northern and central British Columbia (DU7), Central Mountain caribou of east-central British Columbia and west-central Alberta (DU8), and Southern Mountain caribou of southeastern British Columbia (DU9) (COSEWIC, 2011a).

Individual subpopulations that comprise each of the three DUs are generally discrete from one another, including those recognized as members of other DUs (see COSEWIC, 2011a). The Southern Mountain DU and Central Mountain DUs are discrete from other neighbouring DUs in that phylogenetically, these caribou have both northern (Beringian-Eurasian) and southern (North American) lineages (Dueck, 1998; McDevitt *et al.,* 2009, Yannic *et al.,* 2014). Caribou sampled in the Northern Mountain DU all come from the Beringian-Eurasian lineage (Dueck, 1998; Zittlau 2004).

The new Southern Mountain DU, restricted to southeastern British Columbia and northern Idaho (Figure 2), is now comprised of 15 extant subpopulations, all of which belonged to the previous Southern Mountain population. Caribou from this DU have a distinct behaviour related to their use of habitats found in steep mountains with deep snowfall (accumulated snowpack of 2-5 m). These extreme snow conditions have led to a foraging strategy that is unique among cervids, that is, the exclusive reliance on arboreal lichens for 3-4 months of the year (Rominger *et al.,* 1991; Terry *et al.,* 2000). Caribou of the Southern Mountain DU differ from Central and Northern Mountain DU caribou based on inherited traits for behavioural strategies and habitat selection that have resulted from the steep terrain and deep snow (COSEWIC, 2011a). Hence, this group of caribou differs markedly from all other cari-

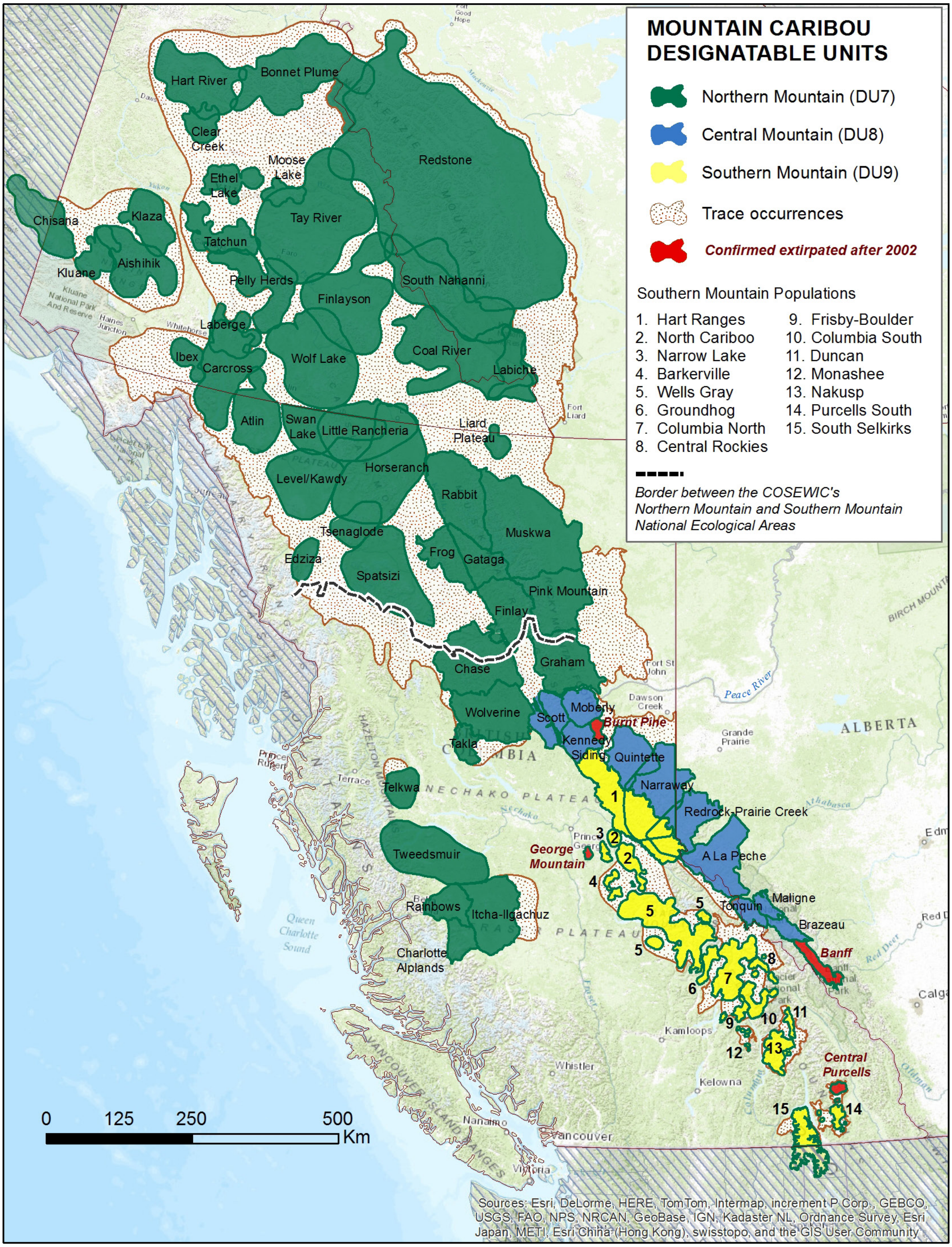

Figure 2. Caribou subpopulations in the Northern Mountain DU, Central Mountain DU, and Southern Mountain DU. The border between COSEWIC's Northern and Southern Mountain National Ecological Areas depicts the COSEWIC (2002) Northern and Southern Mountain Population boundaries (from COSEWIC 2014a). Map created by Bonnie Fournier (Environment and Natural Resources, Government of the Northwest Territories, 2014).

bou, as they have persisted in an ecological setting unique to the species that has given rise to local adaptations.

The Central Mountain DU includes ten extant subpopulations of caribou in east-central British Columbia and west-central Alberta located in and near to the northern Rocky Mountains. There are 45 Northern Mountain DU subpopulations ranging from west-central and northern British Columbia to the northern mountains of Yukon and southern Northwest Territories (Figure 2; Environment Canada, 2014). Subpopulations in the southern part of the Northern Mountain DU have relatively discrete ranges, while range overlap is more pronounced farther north. Animals from these two DUs share similar winter feeding behaviours and seasonal movement patterns, but they differ phylogenetically and are isolated by the Peace River (see COSEWIC, 2014a). Evidence from McDevitt *et al.*(2009) was suggestive of a 'hybrid swarm' of two caribou lineages within the ice free corridor that appeared along the eastern front of the Rockies producing a unique, mixed gene pool at the end of the Wisconsin glaciations ca. 14 000 years ago (Central Mountain DU). Although some evidence indicates genetic relatedness between Northern Mountain DU subpopulations in west-central British Columbia and those in the Central Mountain DU, the majority of sampled subpopulations of Northern Mountain DU caribou differ genetically (Serrouya *et al.,* 2012). All caribou in nine sampled subpopulations the Northern Mountain DU belong to the northern clade (Dueck, 1998; Zittlau, 2004; Weckworth *et al.,* 2012), but only two of 25 subpopulations in northern British Columbia have been sampled, leaving a large gap in phylogenetic information. Further work needs to be conducted to assess phylogenetics and genetic population structure of the Northern Mountain DU in particular.

There are two major differences between this new DU structure and that of the previous assessment (COSEWIC 2002). One change resulted from the reclassification of terrestrial lichen feeding/shallow snow caribou that were previously part of the Southern Mountain population. The new Southern Mountain DU is restricted to central and southeastern BC (Figure 2) and includes only the deep snow/arboreal lichen feeding ecotype. In contrast, all shallow snow/terrestrial lichen feeding caribou were reassigned to either the Central Mountain or Northern Mountain DUs. The second major difference is that the new Northern Mountain DU includes nine subpopulations in central British Columbia from the former Southern Mountain population of Woodland Caribou (COSEWIC 2002), which is currently listed under SARA as threatened and the subject of a recently-released federal recovery strategy (Environment Canada, 2014).

Population abundance and trends

Survey - methods and data availability

The IUCN/COSEWIC criteria most relevant for this assessment (for A and C; Mace *et al.,* 2008; COSEWIC, 2011b) rely on population estimates and trends over time. The 2002 and 2014 status assessments (COSEWIC, 2004; 2014a) and supporting literature explain the methods, including survey frequency, used to estimate the minimum or estimated number of caribou in each subpopulation as well as trends in absolute or relative abundance. In summary, population estimates are challenging to obtain for these animals as they reside in remote areas, occupy large ranges at low densities, and vegetation overstory across forested habitats makes observation difficult. Estimates for some subpopulations may be based on information derived from expert opinion or on sightings of caribou during surveys conducted for other species (e.g., Thiessen, 2009). For subpopulations where late-winter distribution occurs in high-elevation alpine/subalpine habitat (mostly in the Southern and Central Mountain DUs), relatively unbiased minimum counts are reported (e.g., Seip & Jones, 2014). In other cases, however, population estimates are imprecise or do not include a measure of sampling or process variance (Tables 1-3; COSEWIC, 2014a). As with all COSEWIC assessments of wildlife species with appropriate data (COSEWIC, 2011b), the number of mature individuals, either estimated or counted, was used as an approximate estimate of population size or percentage change in population size over two or three generations -- the IUCN timeframes over which declines are measured (Mace *et al.,* 2008). It is important to note that the quantitative criteria used in the COSEWIC status assessments (COSEWIC, 2011b) are dependent on thresholds in total number or percentage change in mature individuals. From this particular perspective, precision and uncertainty becomes most important to consider when estimates approach a set threshold for designations (endangered, threatened, and of special concern).

Survey frequency has varied among the subpopulations for all DUs (Tables 1-3). In the Southern Mountain DU, the earliest available surveys date back to the late 1980s for some subpopulations or portions of those subpopulations (e.g., Barkerville, Wells Gray [south], Groundhog, Quesnel Highlands portion of the Wells Gray [north] subpopulation) (Seip, 1990; Hatter, 2006; McLellan *et al.,* 2006; Freeman, 2012). During the 1990s, at least two surveys were conducted for most subpopulations (Hatter, 2006) and surveys were carried out in most years for Barkerville, Wells Gray (north), Central Purcells, South Purcells, and South Selkirk (Wakkinen, 2003; Kinley, 2007; Freeman, 2012). Since 2002, most subpopulations in the Southern Mountain DU have been surveyed approximately every 2 years.

In the Central Mountain DU, surveys for most British Columbia subpopulations have been conducted only since the mid-2000s (Seip & Jones, 2014). The Jasper National Park subpopulations (Tonquin, Maligne, Brazeau) are surveyed annually during the fall. In addition, population trend, mortality rates of radio-collared caribou and late-winter calf recruitment rates have been tracked for all subpopulations other than Scott (BC) (ASRD & ACA 2010; Seip & Jones, 2014; Alberta Environment and Sustainable Resource Development, unpublished data). By comparison, surveys are incomplete or infrequent for the majority of the subpopulations of the Northern Mountain DU. Twenty-nine of the 45 estimates are older than 5 years, or were based solely on expert opinion, and may not reflect the current population size. Several other population estimates are based on caribou counted during surveys for other species. For 18 of the 45 subpopulations, only one estimate is available and some early surveys did not always include all of the range and so are not comparable to more recent estimates. Only nine of 45 subpopulations have been surveyed more than three times in the past 27 years.

Population trends

Tables 1-3 summarize available subpopulation size and trend data for the Southern Mountain, Central Mountain, and Northern Mountain DUs over the approximate three-generation (27 year) span used for the 2014 COSEWIC assessment (COSEWIC 2014a). Where more than one survey estimate within three generations was available for a subpopulation, we calculated a measure of population change. Few subpopulations had surveys as early as 1987. For those that did not, we used the most recent survey estimate and the highest earliest survey estimate to represent three-generation change, and did not extrapolate further. For subpopulations with one or no survey estimates, when available, population change was inferred from mortality rates of radio-collared caribou and late winter calf recruitment (e.g., Hervieux *et* *al.,* 2013). For subpopulations characterised by few and/or unreliable survey estimates, or where the most recent survey took place five or more years ago, trends could not be determined. We calculated the estimated population trend for each DU since the last COSEWIC assessment by comparing total number of mature individuals in 2014 to those reported by COSEWIC (2002), taking into account changes in DU boundaries.

Southern Mountain caribou DU

The 2014 estimate for the Southern Mountain DU population was 1,354 mature individuals (Table 1). The three-generation decline rate for the overall population was at least 46%. Only two subpopulations had more than 250 mature individuals, nine numbered fewer than 50, six of these fewer than 15. Some former larger subpopulations had split into several due to lack of dispersal within ranges (Wittmer *et al.,* 2005). Two additional subpopulations were recently extirpated: the George Mountain subpopulation in 2003 and the Central Purcells subpopulation in 2005 (Table 1).

All subpopulations in the revised Southern Mountain DU belonged to the former Southern Mountain population of Woodland Caribou (Environment Canada, 2014). The corresponding subpopulations were estimated at 1,850 mature individuals in 2002 (COSEWIC, 2002), indicating a 27% decline. The only increasing subpopulation (Barkerville) has likely benefitted from a recent wolf sterilization and removal program (Roorda & Wright, 2012), although there are <100 mature individuals. Some subpopulations have been subjected to intensive management measures since 2002 (see below).

Because IUCN criteria also take into account projected declines into the future (Mace *et al.,* 2008; COSEWIC, 2011a), recent population viability analyses were informative. Wittmer *et al.* (2010) developed a population viability analysis (PVA) for ten subpopulations of Southern Mountain DU caribou. All ten were predicted to decline to extinction within <200 years and all but two subpopulations had a cumulative probability of extinction of >20% (24-100%) within 45 years (5 generations). Increases in the amount of young forest have resulted in more rapid predicted extinction rates in all populations. Hatter (2006) conducted a PVA for all extant subpopulations in this DU and showed that time to quasi-extinction (N<20 animals) was < 50 years for 10 of 15 subpopulations. The probability of quasi-extinction in 20 years was >20% for 12 of 15 subpopulations and >50% for 13, but Hatter (2006) cautioned that confidence limits indicated a low level of certainty for predictions for five of the subpopulations with a high probability of extinction. By contrast, the largest subpopulations, North Cariboo Mountains and Hart Ranges, were identified in both studies as having a very low probability of extinction in this time period. However, since 2006, both subpopulations have declined, with the Hart Ranges population declining 35% (COSEWIC 2014a).

Central Mountain caribou DU

The 2014 estimate for the Central Mountain DU was 470 mature individuals (Table 2). Nine of ten extant subpopulations each contain fewer than 100 mature individuals, four among them fewer than 50. The long-term trend of the Scott subpopulation in BC, however, is unknown. In addition, the Banff subpopulation was extirpated in 2009 (Hebblewhite *et al.,* 2010), and the Burnt Pine subpopulation was confirmed functionally extirpated in 2014 (Seip & Jones, 2014). The estimated overall decline in the Central Mountain DU population was at least 64% during the last three generations. All subpopulations in the Central Mountain DU belonged to the former Southern Mountain population of Woodland Caribou (Environment Canada, 2014). The corresponding subpopulations were estimated at 1,293 mature individuals in 2002 (COSEWIC, 2002). The decrease in numbers observed during surveys is supported by consistently high adult mortality and low calf recruitment (ASRD & ACA, 2010; Hervieux *et al.,* 2013; Seip & Jones, 2014).

Northern Mountain caribou DU

About 50,000 to 55,000 caribou occurred in the Northern Mountain DU in 2014, of which 43,187 to 47,496 were estimated to be mature individuals (Table 3). These animals accounted for about 95% of western mountain caribou in Canada. Over half (26 of 45) the subpopulations contained more than 500 mature individuals, while 13 subpopulations had fewer than 250. Nine of the 15 subpopulations that consisted of >1,000 mature individuals are located in Yukon and Northwest Territories. Combined, the Bonnet Plume and Redstone subpopulations, the two largest in the DU, comprised >15,000 animals, or 26-29% of the Northern Mountain DU (Table 3).

The four subpopulations that comprised < 50 mature individuals are located in the southern part of the DU in west-central British Columbia (Charlotte Alplands, Rainbows, Telkwa) and north-eastern British Columbia (Finlay). Trend data were limited for subpopulations in this DU, with long-term (three-generation) trend known for only 16 of 45 subpopulations (Table 3). Recent surveys indicate that all five subpopulations in west-central British Columbia (Telkwa, Tweedsmuir, Itcha-Ilgachuz, Rainbows, Charlotte Alplands) are currently declining (COSEWIC, 2014a).

The 2002 COSEWIC assessment estimated the number of mature individuals in the former Northern Mountain population as 43,950 (COSEWIC, 2002), suggesting an overall stable situation for those 36 subpopulations, albeit with considerable uncertainty because of limited survey data (Environment Canada, 2012; COSEWIC, 2014a). In contrast, the nine subpopulations at the southern part of the DU, all of which belong to the former Southern Mountain population of Woodland Caribou (Environment Canada, 2014) have experienced an overall decline of 34% since 2002, from 4,030 to 2,673 mature individuals. (Table 3; COSEWIC 2014a).

2014 COSEWIC assessments of western mountain caribou

In April 2014, the Central and Southern Mountain Caribou DUs were assessed by COSEWIC as endangered (COSEWIC, 2014a). In both cases, the IUCN Red List criteria (Mace *et al.,* 2008) for high decline rate (A) and small and declining populations (C) were invoked because these DUs have experienced pronounced population reductions within the last three generations and most subpopulations are currently small in size.

Criterion A is measured as a percentage of loss of mature individuals over time windows in the past, future, or a combination of the past and future (Mace *et al.,* 2008). The decline of 64% over the past three generations in the Central Mountain population exceeds the criterion of 50% decline for endangered, in cases where the causes of the declines have not ceased and may not be reversible (COSEWIC 2011b). Although the calculated >45% decline for the Southern Mountain population did not exceed the IUCN threshold (50%) for past declines, it qualified as endangered under this criterion based on inferred reduction of >50% within the next three generations based on PVA (Hatter, 2006; Wittmer *et al.,* 2010).

The focus of IUCN Criterion C is on populations that are numerically small and in continuing decline (Mace *et al.,* 2008; COSEWIC, 2011b). Both Central and Southern Mountain caribou are endangered under this criterion, as each population numbered fewer than 2,500 and has experienced an estimated continuing two-generation decline that exceeded the 20%

threshold (at least 62% for Central and 40% for Southern). Furthermore, in the case of Central Mountain caribou there was an apparent continuing decline in number of mature individuals, while no subpopulation was estimated to contain more than 250 individuals (COSEWIC, 2014a).

Northern Mountain Caribou did not meet quantitative thresholds for endangered or threatened when considering overall population size or decline, but were assessed as of special concern due to the deteriorating status of a number of subpopulations and increasing magnitude and scope of threats throughout the DU (COSEWIC, 2014a). All known stable or increasing subpopulations are located in the northern part of the range, whereas nine in the southern part of the range had declined by 34% since the last assessment. However, most subpopulations in this DU receive little to no monitoring attention, and many 2014 estimates were based on survey data older than 5 years. The status of northern subpopulations may be compromised in the future because of increasing threats, particularly land-use change resulting from industrial development, and extent and frequency of forest fires and insect outbreaks related to climate change (e.g., mountain pine beetle) (Environment Canada, 2012; COSEWIC 2014a). Habitat loss and increased predation levels can be expected to influence the distribution and abundance of subpopulations in a similar fashion to that which has taken place in the Central and Southern Mountain DUs (Apps & McLellan, 2006, Wittmer *et al.,* 2007; DeCesare *et al.,* 2011; Hervieux *et al.,* 2013).

Prevailing and future threats

Threats to woodland caribou in Canada, including western mountain caribou, have been well documented (Festa-Bianchet *et al.,* 2011; COSEWIC, 2014a). Recent studies have demonstrated that linear features resulting from roads, trails, geophysical exploration lines, pipelines, and utility rights-of-way can exacerbate susceptibility to predation, and therefore alter the movements, distributions, and population dynamics of caribou. These features facilitate increased predator mobility, hunting, vehicle collisions, disturbance, and directly or indirectly result in habitat reduction and fragmentation (Dyer *et al.,* 2002; Seip *et al.* 2007, van Oort *et al.,* 2010; Williamson-Ehlers, 2012; Apps *et al.,* 2013). Predation is often the primary reason for caribou declines, directly related to increased prey populations that show a numerical and distributional response to early seral forest and linear features that result from cumulative development activities (Serrouya *et al.,* 2011; Apps *et al.,* 2013; Ehlers *et al.,* 2014). Human developments associated with timber harvest, oil and natural gas extraction, wind energy, and mining have a large cumulative footprint, reducing the amount of habitat for caribou and increasing the area of early-successional forests favoured by other ungulate species and the predators of caribou (Nielsen *et al.,* 2005; Nitschke, 2008; Williamson-Ehlers, 2012). Although forest harvesting and mineral and hydrocarbon exploration and development do not generally result in substantial direct mortality of mountain caribou, habitat changes arising from these activities and associated infrastructure affect the abundance, habitat use, and movements of both predators and alternate prey (Festa-Bianchet *et al.,* 2011; Serrouya *et al.,* 2011). Recent large natural disturbances by fire and forest insects may render already limited habitat unavailable for decades, thereby reducing already fragmented ranges. For example, after over 50 years of relatively little fire activity on the Tweedsmuir-Entiako caribou range, a wildfire in 2014 affected over 130,000 ha of winter and spring migration range (R. Krause, BC Ministry of Forests, Lands and Natural Resource Operations, pers. comm.).

In the Northern Mountain DU, human disturbances and habitat loss (including functional habitat loss) have resulted from the cumulative effects of forest harvesting, mineral exploration and development and associated access, motorized and non-motorized recreational activities, changes in forest structure due to mountain pine beetle (*Dendroctonus ponderosae*) infestations and/or associated salvage logging, and impacts from climate change (Environment Canada, 2012; COSEWIC, 2014a). Direct impacts to southern subpopulations in the DU are already evident, whereas those in the northern part of the DU may be affected similarly if the multiple proposed mineral and hydrocarbon exploration and development projects, windfarms, and associated infrastructure are developed in north-central and northeastern BC (COSEWIC, 2014a). For example, in north-western BC, there are known large mineral deposits stimulating exploration activities and mine development in the Skeena region. The 344-km Northwest Transmission line was completed in 2014 to supply power to planned industrial developments and remote communities in the area (BC Hydro, 2015). The new power supply is likely to increase the feasibility of potential projects in and adjacent to caribou ranges in north-western BC.

The primary threats to caribou in the Central Mountain DU include altered predator-prey dynamics due to habitat loss and disturbances from multiple industrial activities including forest harvesting, mining of coal, and the exploration and development of oil and gas reserves. Additional factors include deaths from vehicle collisions, disturbance from motorized recreation (e.g., all-terrain vehicles, snowmobiling), facilitated access to caribou winter range for predators resulting from increased linear corridors and packed trails or ploughed roads in winter, impacts from climate change, and stochastic environmental events associated with small population sizes (DeCesare *et al.,* 2011; Hervieux *et al.,* 2013; Williamson-Ehlers *et al.,* 2012; Johnson *et al.,* 2015). Caribou in the Southern Mountain DU are subject to altered predator-prey dynamics due to habitat change resulting from forest harvesting in adjacent valley bottoms, snowmobiling, heli-skiing, impacts from climate change, and Allee effects that have led to a high likelihood of extirpation due to random environmental and demographic events (Apps & McLellan, 2006; Wittmer *et al.,* 2007; 2013).

Management and recovery actions

Efforts aimed at recovering or managing declining western mountain caribou since the 1980s have focused on habitat protection, population management, and mitigation of individual development projects as the industrial footprint continues to increase across the distribution of all three DUs. In 2007, the Government of British Columbia announced a series of habitat protection measures as part of a Mountain Caribou Recovery Implementation Plan (BC Ministry of Environment, 2015). Specifically, 2.2 million ha of forested lands in the Southern Mountain DU were included in protected areas or designated as Ungulate Winter Ranges or Wildlife Habitat Areas under the provincial Forest and Range Practices Act, whereby mountain caribou habitat requirements receive special consideration when planning and implementing forest harvesting and other industrial (e.g., road building) activities (Environment Canada, 2014; BC Ministry of Environment, 2015). Approximately 1 million ha were closed to motorized vehicles (primarily to restrict snowmobiling; Seip *et al.,* 2007). Ungulate Winter Ranges and Wildlife Habitat Areas generally provide for no or modified forest harvesting and include primarily high elevation habitat in the Central and Southern Mountains, but also low elevation areas in the Northern Mountains. They also provide some restrictions on mineral exploration and guided adventure tourism activities during the calving season. General Wildlife Measures for those areas vary with respect to the proportion of area excluded from forest harvesting, and the levels and methods of forest harvesting in modified harvest areas (COSEWIC, 2014a).

The South Peace Northern Caribou Implementation Plan (BC Ministry of Environment, 2013) provided for protection of ≥90% of identified high-elevation winter ranges across the Central Mountain and a portion of the Northern Mountain DUs. This includes the Graham, Moberly, Scott, Burnt Pine, and Narraway subpopulations in British Columbia. It also specifies protection of ≥80% of identified high-elevation winter ranges on the Quintette range, but provides no indication of how the protected portions of any of the range will be distributed geographically. In the Southern Mountain DU, caribou primarily use high-elevation ranges, and recovery efforts have focussed on protecting most of those ranges from forest harvesting. However, forest harvesting has continued outside of those ranges in adjacent valley bottoms, resulting in increased predation risk for caribou (Apps *et al.,* 2013). Similarly, for caribou in both the Central Mountain DU and the southern part of the Northern Mountain DU, continuing declines in caribou numbers is highly correlated to loss of high-quality habitat and industrial disturbances at low elevations (Johnson *et al.,* 2015).

Intensive management of caribou subpopulations including translocations, predator control, prey control, and captive breeding and rearing initiatives, have been deployed since the mid-1980s (e.g., Compton *et al.,* 1995; Young *et al.,* 2001; Zittlau, 2004; Cichowski, 2014; COSEWIC, 2014a). Initial results can appear promising but then often are not sustained. For example, the Telkwa subpopulation in west-central British Columbia increased after the transplants of 32 caribou from 1997-1999 to at least 144 total caribou in 2006 before declining to the current estimate of 19 animals (Cichowski, 2014). From 1984 to 1991, 52 caribou from the Itcha-Ilgachuz subpopulation were transplanted to the unoccupied Charlotte Alplands range (Young *et al.,* 2001). That subpopulation appeared to remain stable until about 1999, but then declined (Youds *et al.,* 2011). The only transplant of western mountain caribou over the past decade occurred in March 2012, when 19 caribou were brought from the Level-Kawdy subpopulation in the Northern Mountain DU to the Purcells South and Purcells Central ranges in the Southern Mountain DU. Seventeen died within 13 months due to predation by wolves or cougars (n=8), accidents (n=3), malnutrition (n=1), or unknown causes (n=5); the fate of the remaining two is unknown due to GPS-collar malfunction (L. de Groot, BC Ministry of Forests, Lands and Natural Resource Operations, pers. comm.).

Although wolf reduction and/or sterilization programs often enjoy initial success, as measured by enhanced caribou survival or recruitment (e.g., Farnell & McDonald, 1988; Bergerud & Elliott, 1998; Hegel & Russell, 2010), the relatively rare opportunities for longer-term monitoring have demonstrated that such interventions, once ended, do not always have sustained long-term benefits for prey species affected by apparent competition (Wittmer *et al.*, 2013). Over the past decade, predator control efforts have continued, albeit constrained by social acceptability (Serrouya *et al.*, 2011). As part of the Mountain Caribou Recovery Implementation Plan in the Southern Mountain DU, trapping and hunting seasons for wolves and cougars were adjusted in 2007 to encourage removal of those predators near caribou habitat (Mountain Caribou Recovery Implementation Plan Progress Board, 2012). Until 2014, the only wolf removal or sterilization program In the Southern Mountain DU was on the Barkerville and Wells Gray (north) subpopulation ranges, where wolves were removed and sterilized leading to densities of 3.2-3.4 wolves/1000 km^2 across about 60% of the study area; the Barkerville caribou subpopulation increased and the Wells Gray (north) subpopulation remained stable, but calf recruitment remained variable (Roorda & Wright, 2012).

In the Central Mountain DU, a 7-year wolf control effort targeting the Little Smokey range, a boreal caribou subpopulation (Hervieux *et al.,* 2014), likely affected the A La Peche Central Mountain caribou subpopulation as well because it shares the same winter range. In January, 2015, the BC Ministry of Forests, Lands and Natural Resource Operations announced two targeted wolf removal efforts "to save caribou herds under threat from wolf predation" in the South Selkirk subpopulation range (Southern Mountain DU) and the Quintette, Moberly, Scott and Kennedy-Siding) ranges (Central Mountain DU) (BC MFLNRO, 2015). A provincial management plan for grey wolf released by the Government of BC in April 2014 (BC MFLNRO, 2014:17), states that wolf control "to reduce predation risk on endangered caribou" has been a "provincial priority" since 2001. Bag limits for wolf hunting have been removed in specified management units in an effort to reduce predation on caribou.

Two moose population reductions have recently been conducted in the Southern Mountain DU. Liberalized hunting resulted in a 71% reduction in moose numbers and about a 50% reduction in wolf numbers on three ranges in the southern portion of the Southern Mountain DU; the Columbia North population experienced a modest increase while the two small populations (Columbia South, Frisby-Boulder) decreased regardless (Serrouya *et al.,* 2011). In the northern portion of the Southern Mountain DU (Parsnip portion of the Hart Ranges), moose numbers declined, possibly as a result of increased hunting, but over six years, neither wolf nor caribou numbers responded measurably (Steenweg, 2011; D. Heard, British Columbia Ministry of Forests, Lands and Natural Resource Operations, pers. comm.).

Captive breeding has the strong endorsement from the Mountain Caribou Recovery Implementation Plan Progress Board as a means to quickly increase mountain caribou numbers in some key core areas, and there is continued interest by the BC government to augment imperiled populations (C. Ritchie, BC Ministry of Forests, Lands and Natural Resource Operations, pers. comm.). A captive-rearing program was conducted for the Chisana subpopulation in Yukon in the Northern Mountain DU during 2003-2006 (Chisana Caribou Recovery Team, 2010). In that program, between 20 and 50 adult female caribou were captured annually in March and held in large enclosures (pens) until mid-June to increase early calf survival. During the 4-year period, calf survival until mid-June (time of release) averaged 93% for captive-reared calves vs. 33% for calves born in the wild (Chisana Caribou Recovery Team, 2010). Survival of calves after release until mid-October was greater for calves born in the pen (70%) than for calves born in the wild (52%). These results suggested that captive rearing could be an effective tool for small populations that are limited by poor calf recruitment (Chisana Caribou Recovery Team, 2010). Captive-rearing projects are currently being conducted (2014) for the Moberly subpopulation in the Central Mountain DU (10 females captured), and for the Columbia North subpopulation (10 females captured) in the Southern Mountain DU (S. McNay, Wildlife Infometrics Inc., pers. comm.; R. Serrouya, Columbia Mountains Caribou Project, British Columbia, pers. comm.). In 2011, a partnership between Parks Canada, the British Columbia Government, and the Calgary Zoo was created to implement a captive-breeding program that would take breeding stock from British Columbia, and augment or reintroduce animals in the four national parks and in BC

(Parks Canada, 2011b). No further details have been publicly released since then but in late 2014, Calgary Zoo made a decision not to proceed (Ellis, 2014).

The latest Alberta status report (ASRD & ACA, 2010) described various provincial recovery planning efforts for both mountain and boreal ecotypes since 1986. Not until 2005 was a recovery plan (Alberta Woodland Caribou Recovery Team, 2005) approved by the Alberta government, although this was "qualified" in that the recommendation for a moratorium on the allocation of new resource extraction rights until range-specific management plans were in place was not accepted by the government of Alberta (ASRD & ACA, 2010). No habitat has been protected on Alberta provincial lands in the Central Mountain DU for the purposes of caribou protection over the past decade; ongoing industrial development activities are managed through an inconsistently-applied patchwork of caribou-related operating guidelines focused on minimizing the size and duration of individual projects (ASRD & ACA, 2010). Oil leases continue to be sold within Alberta Central Mountain Caribou ranges, as recently as March 2015 (Weber, 2015). Parks Canada has also produced a strategy (Parks Canada, 2011a) to guide conservation efforts, which are primarily focused on measures such as seasonal closures of winter habitat, and management of elk populations, vehicle traffic control measures, and recreation in the four national parks located in the Southern and Central Mountain DUs. Predator-prey relationships in these latter protected areas are heavily influenced by land use practices or human settlements characterized by the surrounding landscapes.

Scientific assessments of Canadian wildlife by COSEWIC represent only the first stage in SARA listing and recovery processes. Assessment is followed by the separate steps of listing decisions and then recovery planning and actions (Mooers *et al.,* 2010). The SARA Recovery Strategy for the Southern Mountain "nationally significant population" assessed by COSEWIC (2002) was finalised at about the same time as the most recent COSEWIC status review (COSEWIC, 2014a). Although COSEWIC (2014a) brought forward changes to both the DU structure and status of many subpopulations (as presented above) that are well-aligned with provincially-recognized ecotypes (Government of British Columbia, 2014), experience demonstrates that it may take some time before legal listing under SARA occurs and these modifications are reflected in the SARA Registry and subject to relevant regulations. The recently completed SARA recovery strategy (Environment Canada, 2014) did, however, seek to clarify this confusing mismatch by acknowledging COSEWIC's new DU structure. That strategy document also partially identified critical habitat specific to the subpopulations of the previously-defined (COSEWIC, 2002) Southern Mountain population.

Conclusions and recommendations

2014 marked the third time COSEWIC has reviewed the status of caribou in the western mountains of Canada (in addition to 1984 and 2002). These status evaluations have documented profound range loss, pronounced and ongoing population declines, unsustainable predation rates, and continuing loss in area and connectivity of functional habitat, resulting in small and isolated subpopulations in southern and central British Columbia and Alberta. At the same time, there are mounting concerns for the welfare of subpopulations in northern British Columbia, Yukon and western Northwest Territories, which face escalating industrial development, even in currently remote regions (Hegel & Russell, 2013; COSEWIC, 2014a). Increased understanding of the distribution, ecology, and genetic variation of these western subpopulations has allowed COSEWIC to apply the Designatable Unit concept to this most recent assessment (COSEWIC, 2011a). This exercise resulted in significant modifications to the boundaries of previously recognized Northern and Southern Mountain "nationally significant populations" (COSEWIC, 2002). COSEWIC also introduced a third unit (Central Mountain), representing subpopulations on the eastern flanks of the Rocky Mountains that were previously considered Southern Mountain caribou (Environment Canada, 2014). Although it may be some time before they are legally recognized, these boundary changes have brought federal recovery units into better alignment with those recognized by provinces and territories, particularly BC.

In spite of considerable management attention to declining populations, available high-quality monitoring data provide a clear indication that recovery actions since the passage of SARA have been generally unsuccessful for caribou in the western mountains of Canada. In some areas, such as the Southern Mountain DU, large areas of important range have been protected from forest harvesting, but herds are still declining, with many reaching very low numbers. Recent actions focused on proximate causes of decline (e.g., predator control or moose reduction) may have helped to stabilise some subpopulations (e.g., Columbia North and Barkerville), but these efforts have not been accompanied by habitat recovery at the scale necessary to enable overall population recovery (Johnson *et al.,* 2015). Alberta, in particular, has relied on mitigation measures to ameliorate site-level impacts of new and past resource development projects. Containment of the human footprint across the range of these mountain caribou DUs, however, is not usually regarded as an option, in light of the economic significance of resource development to provincial economies.

Based on declines, future developments and current recovery effects, we offer the following recommendations:

1) *Commit to reducing human intrusion into caribou ranges, restoring habitat over the long term, and conducting short-term predator control for small and/or declining subpopulations.*
All three components must be conducted simultaneously for successful recovery of western mountain caribou. Implementation of the current recovery and management plans and perhaps more drastic actions will undoubtedly result in trade-offs between the persistence of subpopulations of caribou, economic activity, and societal expectations for conservation. If restraint of the human footprint is not considered, then the prospects for preventing extirpation of declining subpopulation through reliance on mitigation of individual development projects will be increasingly limited.

2) *For management and recovery actions, especially with respect to planned movements of animals to supplement subpopulations, consider carefully COSEWIC's new DU structure for caribou, which explicitly recognizes the evolutionary significance of discrete conservation units of the species in Canada.*
Translocation efforts have involved, on occasion, a transfer of animals from one DU to another, and are being increasingly adopted or considered. Increasingly a component of strategies aimed at maintaining or recovering small subpopulations of caribou in all three DUs (DeCesare *et al.,* 2011; Environment Canada, 2014), have either met with failure, as measured by death, lack of reproduction by introduced individuals, or the results are difficult to disentangle from the effects of other recovery measures applied simultaneously. Translocation projects can also serve to increase threats to caribou subpopulations through 1) the introduction of novel genetic material that could cause outbreeding depression and reduce local adaptations, 2) removal of individuals from source subpopulations that may in some cir-

cumstances exacerbate extinction risk related to small source population sizes, 3) unanticipated disease transfer between environments that characterize caribou ecotypes, or 4) low survival of individuals transplanted from one DU into another if the basis for DU designation is local adaptations to the ecological setting. The ecological and behavioural characteristics that differentiate the three mountain caribou DUs (COSEWIC, 2011a), make the prospects for rescue unlikely through translocation from one DU to another, particularly to the Southern Mountain DU. Experience suggests that the success of most translocations will be compromised if the causes of the original decline are not addressed (St-Laurent & Dussault, 2012).

3) *Carry out regular surveys to monitor the condition of Northern Mountain caribou subpopulations and immediately implement preventative measures on ranges that show signs of population declines or acceleration of threats.*
Although the designation of special concern for the Northern Mountain population confers few obligations under SARA, the current conservation status of the subpopulations in this DU illustrates well the importance of the third stated purpose of the Act “to manage species of special concern to prevent them from becoming endangered or threatened” (SARA, 2002). In light of the worrisome signs already exhibited by southern subpopulations in this DU, intensifying natural resource development and increasing natural disturbance in the region make it necessary to be vigilant and ready to respond.

4) *Undertake a proactive, planned approach coordinated across jurisdictions to address the spatial extent and resource valuation essential to conserving landscape processes.*
Caribou conservation depends on the maintenance of landscape-scale processes expressed across extremely broad areas and a proactive approach to limiting or mitigating land-use changes and cumulative impacts that have demonstrable negative impacts on caribou. Given the limited scope of SARA, recovery and management of western mountain caribou subpopulations will necessitate coordination within and between jurisdictions at appropriate scales, including the effective protection of critical habitat. The prevailing practice of piecemeal project-by-project decision making does not consider how development should proceed at a regional scale and collectively engenders a reactive approach.

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Table 1. Subpopulation and trend estimates for 15 caribou subpopulations in the Southern Mountain Designatable Unit.

Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2001	# full surveys 2002-2014	Survey Estimates[2]							Population Estimates	
				Initial Estimate[3]			2014 Estimate			Long-term % change[4]	2014 Estimate	
				Year	Total	Mature	Year	Total	Mature		Total	Mature
South Selkirks (BC)	SM	9	11	1995[5]	63	53	2014	22	20	-62	22	20
Purcells South (BC)	SM	4	8	1995	69	63	2014	23	22	-65	23	22
Purcells Central (BC)	SM	6	5	1994	22	19	2005	0	0	-100	0	0
Nakusp (BC)	SM	3	9	1996	211	192	2014[6]	64	54	-72	64	54
Duncan (BC)	SM	3	8	1999	31	23[7]	2012	2	2	-91	2	2
Central Rockies (BC)	SM	2	4	1995	30	28	2008	3	2	-93	3	2
Monashee (BC)	SM	1	5	1994	12	8	2011	4	4	-50	4	4
Frisby Boulder (BC)	SM	3	6	1994	43	39	2013	13	12	-69	13	12
Columbia South (BC)	SM	3	8	1994	114	100	2013	7	6	-94	7	6
Columbia North (BC)	SM	3	6	1997	280	247	2013	183	157	-36	183	157
Groundhog (BC)	SM	5	8	1990	109	89	2013	13	11	-88	13	11
Wells Gray (BC)	SM	0	3	1995	631	522	2013	343	298	-43	392	341
Barkerville (BC)	SM	14	6	1988	46	39	2012	88	76	95	90	78
North Cariboo Mountains (BC)	SM	2	4	1999	299	280	2011	222	202	-28	222	202
Narrow Lake (BC)	SM	2	11	1999	81	73	2014	47	45	-38	47	45
George Mountain (BC)	SM	2	7	1993[8]	24	22	2004	0	0	-100	0	0
Hart Ranges (BC)	SM	0	4	2006	716	590	2013	439	381	-35	459	398
Total					**2781**	**2387**		**1473**	**1292**	**-46**	**1544**	**1354**

Footnotes to Table 1.

[1] Nationally significant population (see full explanation in manuscript text).
[2] Censuses of DU9 caribou are conducted using standardized methods and searching predetermined survey areas. Various techniques have been used to estimate number of caribou in the survey area (survey area estimate) and for the whole population (population estimate). For surveys where no radio-collared caribou are present, the survey estimate is usually equivalent to the population estimate. For surveys where radio-collared caribou are available, caribou are sometimes found outside the survey area; these are incorporated into the population estimate but not the survey estimate. In this table, the most recent population estimate is presented for assessing the current population size of caribou in DU9, but % change is based on survey estimates. For surveys where a survey estimate was not provided, the estimate was calculated by applying a standardized sightability correction factor of 0.83 used for DU9 caribou surveys based on Seip (1990) and Young & Roorda (1999) to total caribou seen plus tracks.
[3] This survey/estimate is the oldest reliable survey conducted with the highest count of animals during the last 3 generations (27 years).
[4] Long-term % change is based on the difference between the initial and 2014 survey estimates.
[5] This survey was considered incomplete but it had the highest number of caribou counted in that time period.
[6] This survey includes the Duncan subpopulation but it was not possible to differentiate between the two subpopulations during the survey so all caribou counted are included in the Nakusp subpopulation.
[7] No composition data were available for this year so the number of mature individuals was calculated based on the proportion of adults in all surveys for the subpopulation that included composition data.
[8] No composition data were available for this year; the only year composition was available was 2002 so the number of mature individuals was calculated based on the average proportion of adults in all surveys for the nearby Narrow Lake subpopulation, which included composition data.

Table 2. Subpopulation and trend estimates for 13 caribou subpopulations in the Central Mountain Designatable Unit.

Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2001	# full surveys 2002-2014	Survey Estimates: Initial Estimate[2]			Survey Estimates: 2014 Estimate			Long-term % change[3]	Population Estimates: 2014 Estimate	
				Year	Total	Mature	Year	Total	Mature		Total	Mature
Scott[4] (BC)	SM	0	0	2007	48	37	2014/2007	43	35	Unk	43	35
Moberly (BC)	SM	3	4	1995	189	163	2014	22	18	-89	22	18
Kennedy Siding (BC)	SM	0	7	2007	120	103	2014	30	29	-72	30[5]	29
Burnt Pine (BC)	SM	4	5	1996	20	18[6]	2014[7]	1	1	-94	1	1
Quintette (BC)	SM	0	3	2008	173	147	2014	106[8]	87	-41	106[8]	87
Narraway[9] (BC/AB)	SM	0	0	2008	180	164	2014/2012	86	78	-52	86	78
Redrock/Prairie Creek[10] (AB/BC)	SM	0	0	1999	478	401	2012	127	106	-74	127	106
A La Peche[11] (AB)	SM	0	0	1999	123	106	2012	88	75	-29	88	75
Jasper (AB)[12]	SM	6	12	1989	188[13]	145	2013	51	41	-72	51	41
Banff (AB)	SM	0[14]	6	1986	29	26	2009	0	0	-100	0	0
Total					**1548**	**1310**		**554**	**470**	**-64**	**554**	**470**

[1] Nationally significant population (see full explanation in manuscript text).
[2] This survey/estimate is the oldest reliable survey conducted with the highest count of animals during the last 3 generations (27 years).
[3] Long-term % change is based on the difference between the initial and 2014 survey estimates.
[4] No survey has been conducted for the west side of the Scott range; the estimate for the western portion of the range was based on anecdotal information and expert opinion and has not been updated since 2007, so the same estimate was used for the current estimate for the purpose of assessing overall population trend. The number of mature individuals for the west side of the Scott range was derived by applying the proportion of adults during the two surveys in the east side of the Scott range to the west side estimate. The east side of the Scott range was surveyed in 2007 and 2014.
[5] Based on observation on the low-elevation winter range during the fall, the minimum population is 22 with an estimated population ranging from 25-35 (with a mid-point of 30 from Seip & Jones (2014)). The number of adults was estimated by applying the proportion of adults seen (21/22) to the total population estimate of 30.
[6] No composition data were available for this year so the number of mature individuals was calculated based on the average proportion of adults in all surveys for the nearest subpopulation that included composition data.

Footnotes to Table 2 continued.

[7] Although 1 bull caribou was seen during the March 2014 survey, Seip & Jones (2014) concluded that the population was functionally extirpated.

[8] Extrapolated from a partial survey. Total population estimated at 98-113 (midpoint=106) (Seip & Jones 2014). The number of adults was estimated by applying the % of adults (82%) in the partial survey to the total estimate.

[9] Estimate based on minimum count for Bearhole/Redwillow portion (Seip & Jones 2014) and for the remaining portion, based on 2009 population estimate of 100 caribou and mature estimate of 90 caribou (ASRD&ACA 2010) and then extrapolated back to 2008 using annual lambdas from ASRD&ACA (2010) and extrapolated to 2012 using annual lambdas from Alberta Ministry of Environment and Sustainable Resource Development (unpublished data; 2010: λ=0.983; 2011: λ=0.904; 2012: λ=0.811). Cumulative λ (2008-2012) = 0.55.

[10] Population estimates based on 2009 population estimate of 212 caribou and mature estimate of 178 caribou (ASRD&ACA 2010) and then extrapolated back to 1999 using annual lambdas from ASRD&ACA (2010) and extrapolated to 2012 using annual lambdas from Alberta Ministry of Environment and Sustainable Resource Development (unpublished data; 2010: λ=0.866; 2011: λ=0.921; 2012: λ=0.749). Cumulative λ (1999-2012) = 0.27.

[11] Population estimates based on 2009 population estimate of 135 caribou and mature estimate of 116 caribou (ASRD&ACA 2010) and then extrapolated back to 1999 using annual lambdas from ASRD&ACA (2010) and extrapolated to 2012 using annual lambdas from Alberta Ministry of Environment and Sustainable Resource Development (unpublished data; 2010: λ=0.836; 2011: λ=0.880; 2012: λ=0.835). Cumulative λ (1999-2012) = 0.71. The population was estimated to be 175-200 based on a count of 162 in October 1988 (Brown *et al.* 1994).

[12] There are 3 subpopulations in Jasper, but were combined to assess long-term trend.

[13] Data interpreted from Figure 1 in Hebblewhite *et al.* 2010. Data based on annual maximum counts and telemetry studies. The number of mature individuals was derived by applying the proportion of adults in caribou sightings in that year (Parks Canada unpublished data) to the total number.

[14] Although no structured surveys were conducted prior to 2004 (J. Whittington, Parks Canada, pers. comm. 2015), the 1986 estimate is based on a minimum count obtained from annual maximum counts and/or telemetry studies (Hebblewhite *et al.* 2010).

Table 3. Subpopulation and trend estimates for 45 caribou subpopulations in the Northern Mountain Designatable Unit.

				Survey Estimates							Population Estimates	
				Initial Estimate			2014 Estimate				2014 Estimate	
Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2004	# full surveys 2002-2014	Year	Total	Mature	Year	Total	Mature	Long-term % change[2]	Total	Mature
Northern Yukon/Northwest Territories												
Hart River	NM	1[3]	1	-			2006	2200	1853		2200	1853
Clear Creek	NM	1	0	-			2001	900	801		900	801
Bonnet Plume[4]	NM	0	0	-			1982	5000	4200[5]		5000	4200[5]
Redstone[6]	NM	0	0	-			2012	>10000	7300-10000[5]		>10000	7300-10000[5]
South Nahanni	NM	1	1	2001	1432	1337	2009	2105	1886	10[7]	2105	1886
Coal River	NM	1[8]	1	-			2008	450	413		450	413
La Biche	NM	1	0	-			1993	450	388		450	388
Southwest Yukon												
Chisana[9]	NM	0	9	2003	720	607	2013	701	631	4	701	631
Kluane	NM	1	2	2003[10]	235	204	2009	181	163	-20	181	163
Aishihik	NM	4	1	1981	1500	1399	2009	2044	1813	30	2044	1813
Klaza	NM	2[11]		-			2012	1180	1065		1180	1065

Table 3 continued.

				Survey Estimates							Population Estimates	
				Initial Estimate			2014 Estimate				2014 Estimate	
Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2004	# full surveys 2002-2014	Year	Total	Mature	Year	Total	Mature	Long-term % change[2]	Total	Mature
Central Yukon												
Ethel Lake	NM	1	0	-			1993	316	289		316	289
Moose Lake	NM	1	0	-			1991	300	270		300	270
Tay River	NM	1	0	-			1991	3758	2907		3758	2907
Tatchun	NM	1	0	-			2000	521	415		521	415
Pelly Herds	NM	0	1	-			2002	1000	876		1000	876
Finlayson	NM	4	1	1986	3067	2350	2007	3077	2657	13[12]	3077	2657
Wolf Lake	NM	3	0	-			1998	1491	1240	- [13]	1491	1240
Southern Lakes Yukon												
Laberge	NM	0	1	-			2003	200	176		200	176
Ibex	NM	1	2	1998	424	329	2008	850	748	127	850	748
Carcross[14]	NM	1	2	1997	403	312	2007	775	674	116	775	674
Atlin[14]	NM	1	1	1999	809	679	2007	600-1000	514-857	-2	600-1000	514-857

Table 3 continued.

				Survey Estimates							Population Estimates	
				Initial Estimate			2014 Estimate				2014 Estimate	
Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2004	# full surveys 2002-2014	Year	Total	Mature	Year	Total	Mature	Long-term % change[2]	Total	Mature
Northwest BC												
Swan Lake[14]	NM	0	1	-			2007	600-800	515-686		600-800	515-686
Little Rancheria[14]	NM	2[15]	0	-			1999				800-1600	672-1342
Horseranch[14]	NM	1	0	-			2000	800-1000	680-850		800-1000	680-850
Level Kawdy	NM	1	0	-			1998	1538	1239		1538	1239
Edziza	NM	0	1	-			2006	151	140		151	140
Tsenaglode[16]	NM	0	0	-			2008				100-400	85-340
Spatsizi	NM	1	0	-			1994	2861	2258		2861	2258
Northeast BC												
Liard Plateau[14]	NM	0	3	2005	141	122	2011	151	140	15	151	140
Rabbit	NM	2[17]	1	-			2007	1133	954		1300	1095
Muskwa	NM	1	1	2001	658	559	2007	738	611	9	1000	838
Gataga	NM	1	0	-			2000[18]	265	220		265	220
Frog	NM	1	0	-			2001[18]	237	199		237	199
Finlay	NM	1	1	1994	193	170	2002	26	19	-89	26	19
Pink Mountain	NM	1	0	-			1993	1275	1145		1275	1145

Table 3 continued.

				Survey Estimates							Population Estimates	
				Initial Estimate			2014 Estimate				2014 Estimate	
Subpopulation	2002 COSEWIC Assessment NSP[1]	# full surveys 1987-2004	# full surveys 2002-2014	Year	Total	Mature	Year	Total	Mature	Long-term % change[2]	Total	Mature
North-central BC												
Graham	SM	1	2	-			2009	708	637	- [19]	708	637
Chase	SM	1	4	-			2009	475	404	- [20]	475	404
Wolverine	SM	1	6	1996	361	324	2010	341	298	-9	341	298
Takla	SM	1	1	-			2004	122	98		122	98
West-central BC												
Telkwa	SM	10	13	1987	40	33	2013	16	12	-64	25	19
Tweedsmuir	SM	2	1	1987	471	433	2002	300	248	-47[21]	300	248
Itcha-Ilgachuz	SM	14	7	1987	933	675	2014	1350	905	34[22]	1685	1220
Rainbows	SM	7	1	1987	103	92	2008	50	43	-53	50	43
Charlotte Alplands	SM	6	0	1993	53	38	2001	23	19	-50	23	19

[1] Nationally significant population (see full explanation in manuscript text).
[2] Long-term % change is based on the difference between the initial and 2014 survey estimates.
[3] Although a survey was conducted in 1978, the population estimate was not comparable to the most recent survey because the survey area was not exactly the same.
[4] Estimate based on expert opinion.
[5] The number of mature individuals was derived by applying the average % adults from hunter observations from 1991 to 2010 (Larter 2012).
[6] This estimate is based on an opportunistic ground survey and therefore was not included as a conducted survey.
[7] % change in population size based on 1337 mature individuals in 2001 and 1465 mature individuals in 2009 for an area comparable to the 2001 survey area.
[8] Although a survey was conducted in 1997, the population estimate was not comparable to the most recent survey because the survey area was not exactly the same.
[9] Fall composition surveys have been conducted annually from 1987 to 2011, except 1989 and 2004 (Chisana Working Group 2012). Data prior to 2003 are not considered because population estimates were based on an interpolation of composition data. Since 2003, population estimates are based on formal estimates of the subpopulation's size and are not directly comparable to pre-2003 estimates.

Footnotes to Table 3 continued.

[10] The 2003 survey was a fall composition survey so it was not a formal population estimate.

[11] Although surveys were conducted in 1989 and 2000, the population estimates were not comparable to the most recent survey because the survey areas were not exactly the same.

[12] Although the change from 1986 to 2007 was a net increase of 13%, the population increased from 2350 mature individuals in 1986 to 4474 in 1990, and then decreased to 2657 by 2007 (a 41% decline from its peak in 1990 to 2007).

[13] No recent population estimate so the 3 generation % change was not calculated.

[14] The ranges of the Carcross, Atlin, Swan Lake, Little Rancheria, Horseranch and Liard Plateau subpopulations straddle the Yukon/BC border

[15] Although surveys were conducted in 1988 and 1999, the population estimate were not comparable because the survey areas were not exactly the same.

[16] The population estimate is based on expert opinion; the number of mature individuals was derived by assuming 85% adults.

[17] Although surveys were conducted in 1996 and 2000, the population estimates were not comparable to the most recent survey because the survey areas were not exactly the same.

[18] There is insufficient information available to determine whether this was a full count or a partial count so this estimate should be considered a minimum number present until a full survey can be conducted, and should not be used to assess trend when a full survey is conducted.

[19] Absolute % change not possible to assess since the total survey area varied between surveys.

[20] Although surveys were conducted in 1993, 2002, 2007 and 2008, the population estimates were not comparable to the most recent survey because the survey area/survey method were not exactly the same.

[21] The decline is supported by an average λ of 0.947 for 12 years where data were available during the period 1985/86 to 2008/09 (Cichowski & MacLean 2005, Cichowski 2010).

[22] Although the change from 1987 to 2012 was a net increase of 34%, the population increased from 675 mature individuals in 1987 to 2161 in 2003, and then decreased to 905 by 2014 (a 58% decline from its peak in 2003 to 2014).

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Peary caribou distribution within the Bathurst Island Complex relative to the boundary proposed for Qausuittuq National Park, Nunavut

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Abstract: How caribou (*Rangifer tarandus*), including Peary caribou (*R. t. pearyi*), use their annual ranges varies with changes in abundance. While fidelity to some seasonal ranges is persistent, use of other areas changes. Consequently, understanding changes in seasonal distribution is useful for designing boundaries of protected areas for caribou conservation. A case in point is the proposed Qausuittuq (Northern Bathurst Island) National Park for Bathurst Island and its satellite islands in the High Arctic of Canada. Since 1961, Peary caribou have been through three periods of high and low abundance. We examined caribou distribution and composition mapped during nine systematic aerial surveys (1961–2013), unsystematic helicopter surveys (1989–98), and limited radio-collaring from 1994–97 and 2003–06. While migration patterns changed and use of southern Bathurst Island decreased during lows in abundance, use of satellite islands, especially Cameron Island for winter range, persisted during both highs and lows in abundance. The northeast coast of Bathurst Island was used to a greater extent during the rut and during summer at low abundance. We suggest that Park boundaries which include Cameron Island and the northeast coast of Bathurst Island will be more effective in contributing to the persistence of Peary caribou on the Bathurst Island Complex.

Key words: boundaries; distribution; Peary caribou; protected area; *Rangifer tarandus pearyi*.

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Introduction

Peary caribou *Rangifer tarandus pearyi* regularly occur only on Canada's Arctic islands (Fig. 1) and have been nationally recognized as Endangered since 1999 based on declines and fluctuations in sub-populations (COSEWIC 2004). On Bathurst Island and its satellite islands, Peary caribou abundance has been through three periods of high (early 1960s, early 1990s and 2013) and low abundance (early to mid-1970s and mid- to late 1990s) (Miller & Barry, 2009:Table 1; Jenkins *et al.,* 2011; Anderson, 2014). The die-offs in the early 1970s and mid-1990s coincided with winters characterised by early and unusually high snowfall, freezing rain, and warmer temperatures (Miller & Barry, 2009).

Parks Canada selected the proposed Qausuittuq National Park on northern Bathurst Island and its satellite islands (Fig. 1) in 1996 to be representative of the Western High Arctic Natural Region (Parks Canada, 2012). National parks have a goal to maintain ecological integrity over the long term, which requires that parks encompass the habitat and connectivity needs for viable wildlife populations. To maintain ecological integrity, national parks need to accommodate the natural range of variability (Landres *et al.,* 1999) which, for Peary caribou, is marked by pronounced changes in numbers driven by abundance of forage or sporadic, unpredictable abiotic variables (Species at Risk, 2012). Those changes in abundance are reflected in shifts in distribution and migratory strategies at least on Bathurst and its satellite islands. Thus, for migratory species such as caribou and their predators, park boundaries need to capture sufficient migratory corridors within natural ranges of variability. Elsewhere in North America, the role of national parks for migratory mammals is well-recognized (Berger, 2004).

Habitat requirements of Peary caribou (summarized in Species at Risk, 2012) vary among areas and seasons, but a key habitat requirement is terrain and vegetation features that offer choices as caribou adjust their foraging to changing snow conditions. Upland habitats with shallow snow cover are selected during winter in many areas. Calving areas generally provide snow-free or shallow snow-covered sites. Habitat requirements during the snow-free season relate to maximizing protein intake and the most nutritious forage. Specific fall and rutting areas occur, but these habitat requirements are poorly known. Annual migrations and range sizes vary among years, but can range up to 500 km and 4,000 km^2, respectively (COSEWIC, 2004; Jenkins *et al.,* 2011).

Seasonal distribution and migration strategies vary with changes in abundance and over the longer-term, with climate (Species at Risk, 2012). In 2012 Parks Canada requested we assess Peary caribou distribution and movements relative to the proposed Qausuittuq National Park boundaries (Gunn *et al.,* 2012). Updated with data from 2013 (Anderson, 2014), we examined the ability of the proposed boundaries to protect caribou throughout their population cycles – especially during the more important calving, post-calving, rutting and winter seasons – and how information on distribution can contribute to decisions about the proposed Park boundary. We assumed that if a seasonal range was used by a substantial or disproportionate proportion of the population during any period of cyclic high or low abundance, then to be effective a national park should encompass that range. Our objectives were to 1) determine the relative distribution of Peary caribou based on aerial surveys for Bathurst Island and the Governor General Islands, 2) determine the distribution of Peary caribou relative to the boundaries of the proposed Qausuittuq National Park, 3) determine at the individual caribou scale, seasonal ranges and movements relative to the proposed boundaries of the Park, and 4) summarize the adequacy and effective-

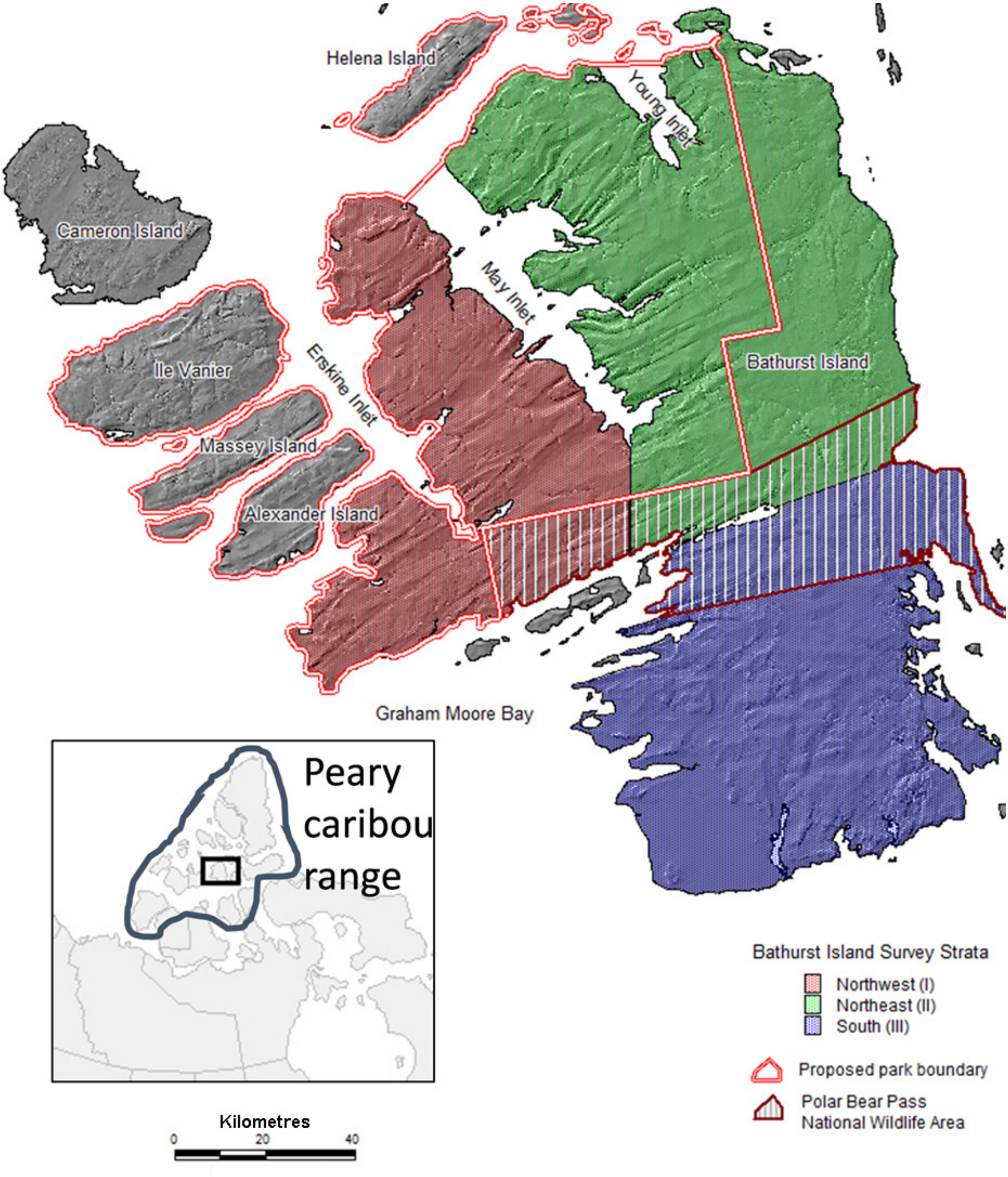

Figure 1. Study area for Bathurst Island, satellite islands and water bodies, NU.

ness of the proposed Park boundary relative to known Peary caribou distribution and habitat requirements (COSEWIC, 2004).

Methods

Study area

Bathurst Island is a relatively large island (16 080 km^2) cut by deep inlets and bays into several large peninsulas; 77% of the land mass is within 10 km of the coast line. Miller (1998) described the 'Bathurst Island Complex' (28 000 km^2) as the approximately 30 islands clustered around and including Bathurst Island. In addition to Bathurst Island, we examined Peary caribou use of the five large islands lying along the northwest coast (Governor General group of islands: Cameron 1,059 km^{2,} Vanier 1,126 km^2, Massey 432 km^2, Marc 56 km^2, Alexander 484 km^2) and island groupings off the north coast (Helena 220 km^2 and the surrounding six small islands; Fig. 1).

The vegetation in this area is mostly High Arctic semi-desert (Gould *et al.,* 2003; Miller & Barry, 2009) with a sparse to moderate cover of cushion forbs, prostrate dwarf shrubs, sedges and grasses. The climate is a short plant growing season marked by variability in the dates in June and August when green-up starts and ends, respectively.

Proposed Qausuittuq National Park boundaries

The current 2002 federal boundary proposal covers Bathurst Island north of the Polar Bear Pass National Wildlife Area (PBPNWA) except for the northeast coast, and includes all the Governor General Islands except Cameron Island, and Helena Island and surrounding small islands (except Seymour Island) (Fig. 1; Parks Canada, 2012). Polar Bear Pass National Wildlife Area is an east-west oriented wetland designated in 1990 (surface and subsurface rights to exploration and development are withdrawn). The Park proposal reflects the recommendations of the Senior Mineral Energy & Resource Assessment Committee, which rated high potential for lead zinc mineralization on the northeast coast of Bathurst Island, and petroleum potential on southwest Cameron Island (Parks Canada, 2012). Southern Cameron Island is the southern extension of the Sverdrup Basin which has petroleum potential, where Bent Horn was a single producing well abandoned due to falling pressure in 1996. The highly fractured field limited production but was included in a 2012–13 call for exploration bids in anticipation that new techniques might be applicable to further development (AANDC, 2012).

Data

Most data on caribou were collected before the lands were withdrawn and, therefore, studies were not specifically designed to examine boundaries. The historic data had variable spatial and temporal resolution, and most were not available digitally.

<u>1. Aerial surveys of Bathurst and satellite islands</u>

Nine complete systematic aerial strip transect surveys were conducted between 1961 and 2013 (Tener, 1963; Miler *et al.,* 1977; Miller, 1989; Ferguson, 1991; Gunn & Dragon, 2002; Jenkins *et al.,* 2011; Anderson, 2014). Data from systematic surveys –mostly text descriptions and the raw data (observation sheets, maps) – were unavailable for all surveys except 1988, 1997, 1998 and 2013. S. Barry and F.L. Miller (Canadian Wildlife Service (CWS), retired) provided scanned images of the original maps for Tener (1963), which included transects, caribou group locations and numbers. The data in the published reports allowed us to describe the proportional distribution of Peary caribou by satellite island and the western and eastern halves of northern Bathurst Island. To further examine the distribution relative to the proposed Park boundaries, we screen digitized report figures. We obtained digital locations for 1997 and 1998 caribou surveys from GNWT-

WMIS and for the 2001 survey we scanned maps from Jenkins et al. (2011).

Between 1989 and 1998, the CWS conducted unsystematic helicopter surveys to describe the relative distribution and sex and age composition in June, July and August (Miller, 1991; 1992; 1993; 1994; 1995a; 1997; 1998; Miller & Gunn, 2003). Bathurst Island was divided into 12–13 search zones, and a helicopter was used to fly a low-level, unstructured path through each zone. Live animals and carcasses were counted, and composition data were collected. We summarize the proportionate use of the areas as described in Miller's reports as the observations were neither mapped nor georeferenced.

2. Satellite telemetry

Miller (1997; 1998; n.d.) captured and fitted satellite and VHF collars on adult cow and bull caribou in 1993 and 1994 on Bathurst Island and the Governor General Islands. We scanned and digitized Miller's (n.d.) maps to describe collared caribou home ranges relative to the proposed Park boundaries. We depicted home ranges using a minimum convex polygon (MCP) (Mohr, 1947) since collar location sample sizes were variable and digital locations were not available. Miller (2002) and Miller & Barry (2003) described the seasonal movements of five satellite-collared cows and one bull for July 1993 and 1994 which was a favourable winter with dry shallow snow. During early winters of 1994-1996 (1 September to 30 November) snowfall was high (>1.5 SD above the 55 yr mean), suggesting these were unfavourable winters. In July 1994, four of the five cows and the bull were re-collared as well as two more bulls. After inspection of the MCPs revealed the ranges in the unfavorable years were larger, we examined differences using a **t**-test (Two-Sample Assuming Unequal Variances and one-tail test). We acknowledge a possible confounding effect of comparing range polygons from 1 year compared to 2 or 3 years, but this comparison broadly supports our point of larger ranges in unfavourable years.

In 2003, M. Ferguson (Government of Nunavut) fitted seven Peary caribou cows with satellite transmitters, which provided data from 29 April 2003 to 18 May 2006. Jenkins & Lecomte (2012) initially reported the collared caribou use of sea-ice, calving and wintering areas. Using the same collar data, we examined calving, summer and wintering areas relative to the proposed Park boundaries. The satellite collars had corresponding <150 m, <350 m, and <1,000 m error with satellite collar locations of location quality classes 3 (79% of locations), 2 (18%), and 1 (2%), respectively, and a 2-day duty cycle from 10 April to 10 July and a 5-day duty cycle for the remainder of the year.

From the 2003–06 collar dataset we developed annual and seasonal (summer and winter) ranges using 90% fixed kernels (Worton, 1989; Seaman & Powell, 1996) using the Home Range Extension (Rodgers & Carr, 1998) for ArcView, with unit variance standardization, a user-defined smoothing factor of 0.60, and raster resolution set to 120. These polygons depicted annual range use with two caveats: a) mid-April to mid-July is over-represented on an annual basis, and b) the collars have an associated error that may be as much as 1,000 m. For summer range, we used 1 July to 15 September, and for winter range we used 1 November to 15 April.

To estimate the 1993–97 and 2003–06 collared cow use of the proposed Park, we measured the proportion of each polygon or kernel within 1) the current proposed Park boundary, and 2) the current proposed boundary plus the northeast section of Bathurst Island east of the boundary (and north of PBPNWA) and Cameron Island.

For each year (1 May to 30 April) and for the cumulative dataset 2003–06, we calculated the number of days each caribou spent within the proposed Park, northeast Bathurst Island outside of the Park boundaries, Cameron Island, and south Bathurst Island. Days were calculated from sequential collar locations; movements that crossed boundaries between areas were weighted by the length of segment in each area. We examined the 2003–06 collar locations to describe the timing and direction of movement by individual cows across the current proposed Park boundaries: the area east of the northeast boundary and movements to or from Cameron Island. We generated sequential line segments by caribou collar number and date, extracted those segments that crossed either the Cameron Island coast or the northeast boundary, calculated segment lengths within and outside both areas, and assigned the crossing date based on the length proportions.

We determined the annual locations of calving sites from the 2003–06 collar movement rates and examination in GIS (Fancy & Whitten, 1991; Kelleyhouse, 2001; Gunn *et al.,* 2008; comparable to average daily displacement in Miller & Barry, 2003). Clusters of locations between late May and early July that demonstrated the lowest daily movement rates and a degree of localization were averaged using the Animal Movement extension to ArcView (Hooge & Eichenlaub, 2000) to determine the approximate annual calving site. We then compared the distribution of the calving sites to Miller's (2001) generalised map of known calving areas.

3. Rutting areas

We obtained locations for about 160 shed bull antlers and bull carcasses from the 1995–97 die-off (GNWT-WMIS) from a July 1998 survey, where Zittlau et al. (1999; unpubl. data) had collected samples for mtDNA analyses from the shed antler and carcass locations (Gunn & Miller, 2003). Prime bull caribou tend to shed fairly quickly after the rut. Many of the carcasses were prime bulls with fully developed antlers, and we assumed where they died was close to or within a rutting area. We also examined the 2003–06 collar data and assumed that rut occurred during the last 10 days of October. We described these locations relative to the proposed Park boundaries.

4. Carcass locations

In July 1998, an unsystematic aerial survey recorded the distribution of carcasses from the 1995–97 die-off on Bathurst Island and the Governor General Islands (Miller & Gunn, 2003). We acquired the carcass database (locations, sex and age classes) through NWT-WMIS. We plotted the locations of the carcasses and summed their distribution relative to the proposed Park boundaries. In addition, Miller (1998) reported on carcasses found during an unsystematic aerial surveys in July 1996.

Results

Relative distribution of caribou from systematic aerial surveys

Between 1961 and 2013 there were nine systematic surveys for Peary caribou on Bathurst Island and seven of those surveys included the Governor General Islands (these islands were not covered in August 1974 and August 1981). The two winter surveys (March–April 1973 and 1974) had a relatively high proportion of caribou on southern Bathurst Island, and during the two spring (latter half of May) surveys caribou were less concentrated in the northeast stratum and more evenly distributed among strata (Table 1). During May 2013 proportionately more caribou groups were observed on the Governor General Islands (only two groups on Cameron Island).

For the systematic surveys flown in summer 1961, 1974, 1981, 1988 and 1997, more than 35% of the groups of caribou were on the northeast stratum, especially in 1974 and 1981 when over 70% were observed in this stratum (Table 1). The proposed eastern boundary of the Park runs north-south and divides the northeast stratum into two segments (Fig. 1). The percentage of groups that would have been outside the Park varied among surveys (Table 1). In 1981, over 45% of the caribou groups in the northeast stratum would have been outside the proposed Park boundary eastern but in the other years, the percentage excluded from the Park was generally 10–15%.

Describing the relative distribution of Peary caribou among the islands off the Governor General Islands is hindered as very few caribou groups were recorded in several surveys. Overall between 1961 and 2001, about 70% of the caribou groups within the Governor General Islands occurred on the two largest islands (Vanier and Cameron), although in May 2013 only 4% of groups occurred on Cameron Island and the remainder were relatively evenly distributed on the other Governor General Islands.

Relative distribution of caribou from unsystematic surveys

Unsystematic aerial surveys were flown in June–August 1989–95 (Miller 1991; 1993; 1994; 1995a; 1997) and 1998 (Miller & Gunn, 2003). The percentage distribution among the three strata changed during calving and post-calving (χ^2 = 179, 6 *df*, $P < 0.001$) with an increase in use of the northwest stratum in July compared to June (Table 2). The pattern of distribution changed with very high (88%) use of the northeast in July 1998 (after the ~90% decline) and low use of the other areas.

Satellite telemetry

From July 1993 to July 1994, two of the four collared caribou on northern Bathurst Island had home ranges that were within the 2002 Park boundaries. Two of the cows used the east coast and crossed the northeast boundary. The two caribou on the Governor General Islands moved between those islands including Cameron Island and Bathurst Island. However, the pattern changed during the following 2 years with unfavourable winters, as the caribou made wide-ranging movements within and off Bathurst Island. One cow in October 1995 moved 110 km across sea ice to Lougheed Island and then a further 110 km to Borden Island where she died in December 1995. All six caribou made movements across the 2002 Park

Table 1. Proportion of caribou groups relative to survey strata (Fig. 1) for Bathurst Island and the Governor General Islands, and for within and outside the proposed Park boundary in the northeast stratum, derived from figures in the survey reports, NU, 1961–2013.

	Proportion of groups				Proportion of groups in NE II	
Survey	**NW I**	**NE II**	**S III**	**GG Is.**	**Within Park**	**Outside Park**
Jun-Jul 61	24	48	1	27	33	15
Mar-Apr 73	18	22	51	8	6	15
Mar 74	6	23	57	13	15	9
Aug 74	14	71	14	0	57	14
Aug 81	12	77	12	0	29	47
Jul 88	18	36	10	35	--[a]	--[a]
Jul 97	29	43	14	14	29	14
May 01	38	21	29	13	13	8
May 13	32	20	15	25	6	14

Table 2. Proportion of caribou observed corrected for effort (flying time) relative to the three survey strata on Bathurst Island during unsystematic surveys in June, July and August 1989-1998, Nunavut (Miller 1991; 1993; 1994; 1995a; 1997; Miller & Gunn, 2003).

	Proportion of caribou by survey									
Strata	Jun 89	Jul 89	Jun 91	Jul 91	Jun 92	Jul 92	Aug 93	Jun 95	Jul 95	Jul 98
NW I	8	42	10	28	16	39	19	42	36	5
NE II	67	38	60	60	48	51	69	46	56	88
S III	25	20	29	12	37	10	12	13	8	6

boundaries and three of the six used Cameron Island. By 1997, all the satellite-collared caribou were dead.

The size of the ranges (100% MCP) varied (t = 2.1, df = 3, P = 0.06) between the 1993–94 (rated as a favourable winter) and 1994–97 rated as unfavourable winters (Fig. 2). In the favourable year, four females had a home range four times smaller (2,118 ± 210 km^2) than the home range area for five females during the unfavorable years (8,899 ± 3,191 km^2). For the three individual females collared for more than 1 year, home ranges in the favorable year (2,004 km^2) were five times smaller than during the unfavorable years (10 983 km^2). Home ranges were larger for the single male during the favourable year (6,132 km^2) and also for the two males during the unfavourable years (5,604 and 7,780 km^2).

The seven female caribou tracked from 2003–06 (Jenkins & Lecomte, 2012) had a mean annual range (90% fixed kernel) of 3,994 ± 747.4 km^2 (SE). Seasonal ranges did not show any consistent trends in size among 2003–06.

Although none of the 1993–97 caribou home ranges were restricted to within the 2002 boundaries, four of the six caribou in a favourable year used the Park but also the segment of northeast Bathurst Island outside the Park. The other two caribou also used the Park, Cameron Island and outlying area. The mean percentage of home ranges in the Park in a favourable year was 55% (± 7.2% SE) which rose to 83% (± 11.7%) if the northeast coast of Bathurst Island was included. However, in unfavourable years the percent of home ranges within the Park dropped to 31% (± 8.3%) and 42% (± 7.1%) if northeast Bathurst was included. The percentage use of Cameron Island scarcely changed, but the use of outlying areas increased sharply during unfavourable years (from 13 to 51%).

None of seven 2003-06 caribou home ranges was restricted to the proposed Park and only one cow's home range on southern Bathurst Island (summer and winter) was outside the 2002 boundaries, northeast Bathurst and Cameron Island. For the other six cows, the mean percentage of home ranges in the Park was 40% (± 4.0% SE) which only increased to 43% (± 4.6%) if the area of northeast Bathurst Island was included and 54% (± 5.5%) if Cameron Island was included.

The summer ranges show more concentrated use for northern Bathurst Island extending southwest into the PBPNWA (Fig. 3). Of the Governor General Islands, only Vanier Island received use in summer although calving occurred on Alexander Island. The winter ranges for the six cows (Fig. 4) showed a contrasting pattern to summer. Four of the six cows wintered on Cameron Island for two or three winters (Fig. 4).

The seven collared caribou monitored during 2003–06 spent an average of 41% of their time within the proposed Park boundaries, 30% on Cameron Island, 3% on the northeast coast, and 26% on south Bathurst Island. The seven animals made limited use of northeast Bathurst Island. Differences among years were relatively minor (e.g., time within the proposed

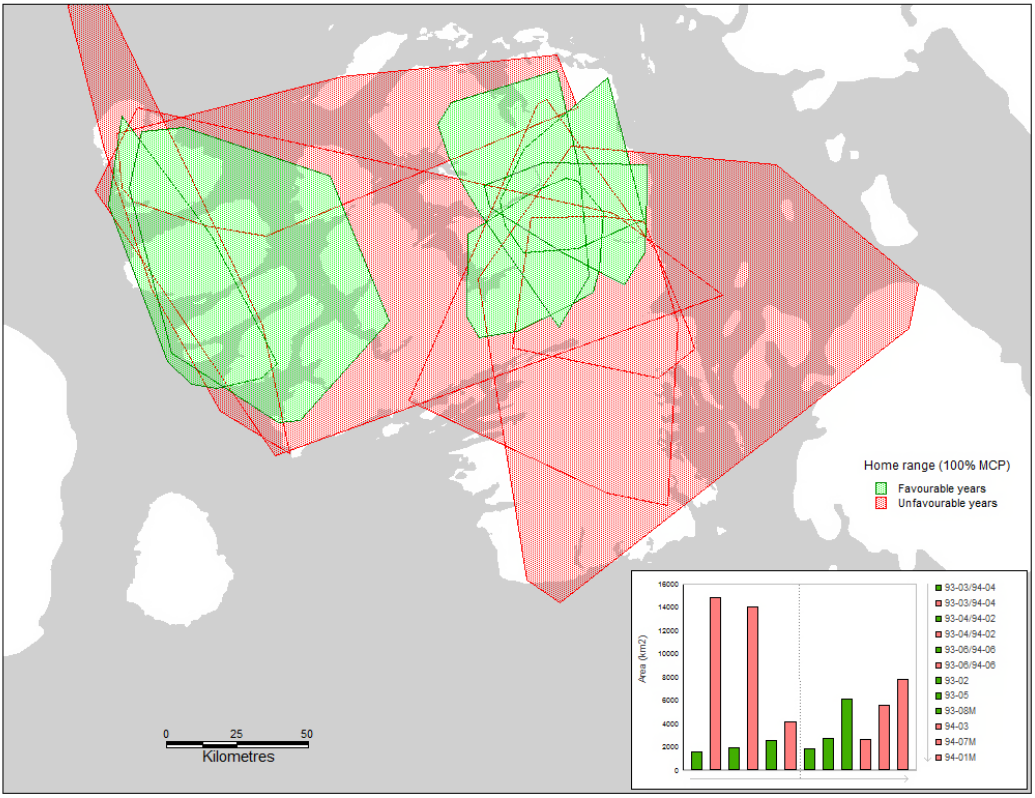

Figure 2. Minimum convex polygons (100%) for 1993–94 and 1994–97 satellite-collared caribou relative to favourable (green polygons) and unfavourable (red polygons) winters (see text); Bathurst Island Complex, NU (derived from F. L. Miller. unpubl.).

Park boundaries for 2003–04, 2004–05, and 2005–06 were 48%, 36%, and 40%, respectively) and non-significant (χ^2 = 7.9, df = 6, P = 0.24).

Two cows from the 2003–06 telemetry dataset crossed the northeast boundary a total of 16 times and five cows crossed a total of 31 times to or from Cameron Island. Most crossings to Cameron Island occurred in September and October (median 7 October, 80% occurred between 15 September and 25 October; n = 15). Most crossings from Cameron Island occurred in April (median 20 April, 81% occurred between 21 March and 20 May; n = 16). Movements to and from northeast Bathurst Island occurred throughout the year, with roughly half occurring from late July to late September.

Three of four collared cows in 1994 calved within the 2002 boundaries and the fourth cow calved on the east coast (Miller & Barry, 2003). The cows collared 2003–06 had a total of 21 suspected calving sites and 80% were within the 2002 boundaries. One calving site was outside the northeast boundary and one cow had three calving sites on south Bathurst Island. One cow calved for 3 years on Alexander Island which is the only one of the Governor General Islands used by the seven cows for calving.

Rutting areas

Based on cast bull antlers and 1993–97 telemetry, five rut areas (Miller, 2001) were mapped which included two within the Park (Fig. 5). The 2003–06 collars indicate rutting occurred primarily on Cameron Island (12 of 21 rut-years) with other locations similar to those identified by Miller (2001). The 2002 Park boundaries would exclude the northeast coast and Cameron Island rutting areas.

Caribou carcass distribution

The carcasses observed represented the cumulative deaths between fall 1994 and summer 1997; the month of death was unknown but most likely occurred during winter. No fresh carcasses were observed in July 1998 to suggest mortality the previous winter or spring. Miller and Gunn (2003) found that bulls occurred at

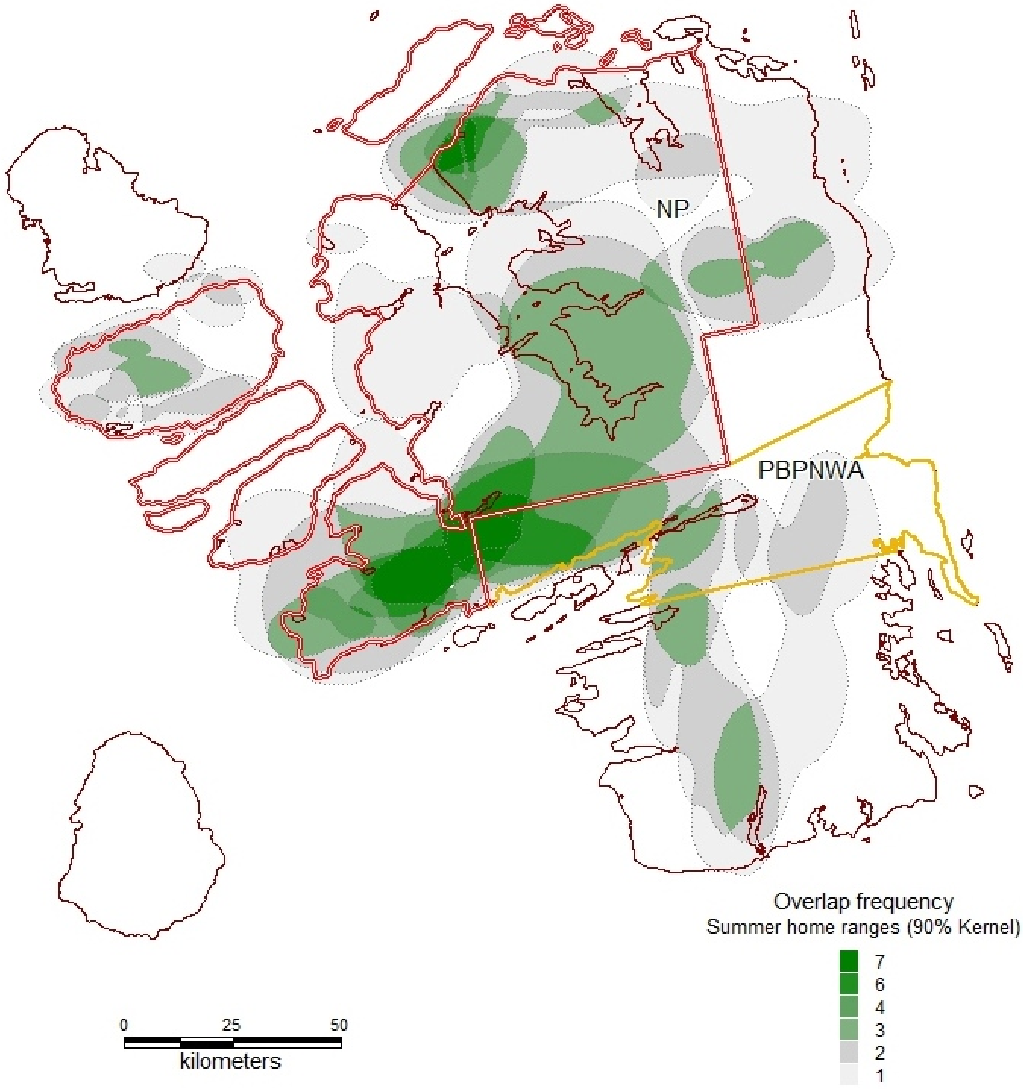

Figure 3. Summer individual female Peary caribou ranges for 2003–06 (unpubl. GN data).

nearly twice their expected rate in the carcass sample, while cows and juvenile/ yearling males and females were underrepresented. Miller (1998) reported 146 dead caribou (estimated 1,143 ± 164) found during unsystematic aerial surveys in July 1996 with a significantly greater than by chance distribution on Cameron and Vanier islands compared to Bathurst Island and fewer carcasses than expected on southern Bathurst Island. In 1998, about 25% and 30% of all carcasses were on Cameron Island and southern Bathurst Island, respectively (Miller & Gunn, 2003).

Discussion

We suggest that based on information collected from 1961 to 2013 the proposed 2002 boundaries for Qausuittuq National Park on the

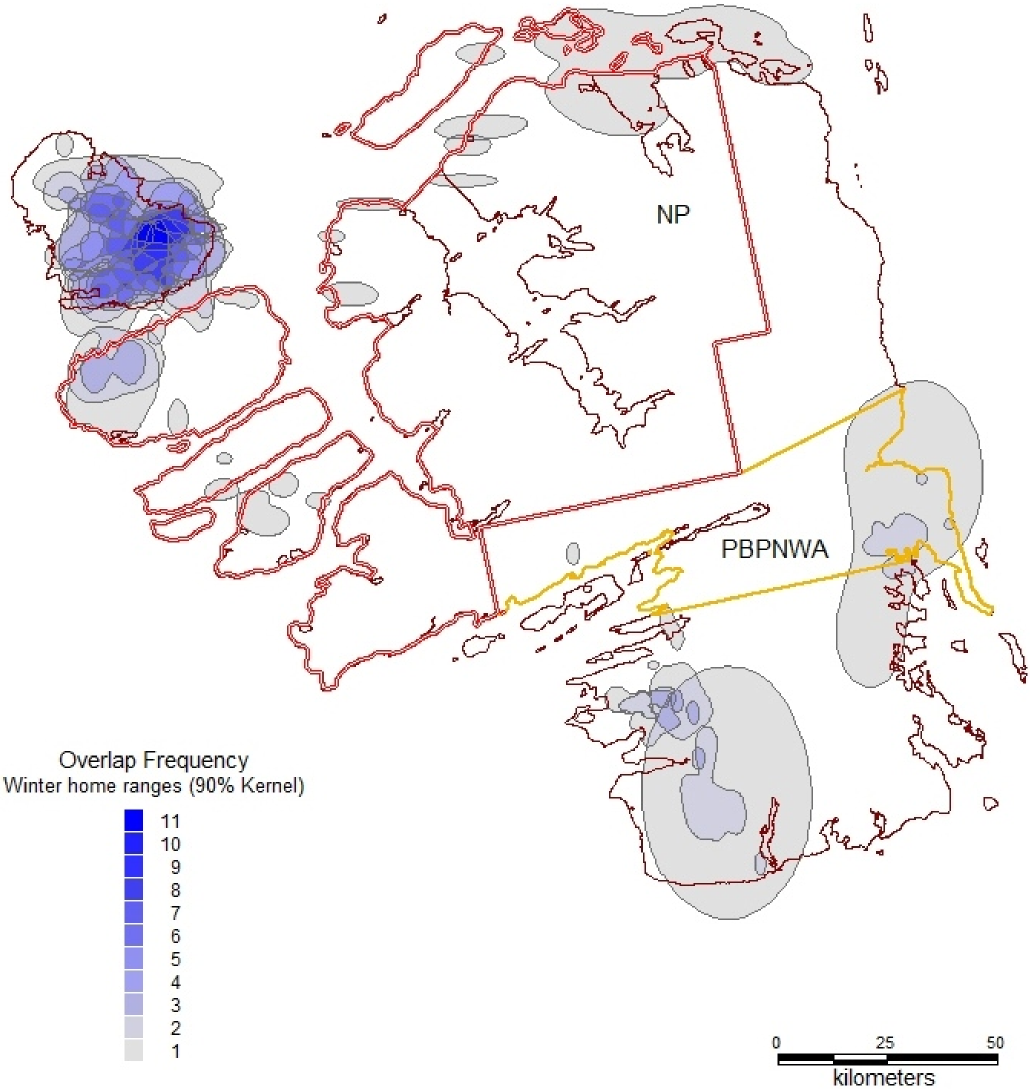

Figure 4. Winter individual female Peary caribou ranges for 2003–06 (unpubl. GN data).

Bathurst Island Complex are relatively ineffective to protect Peary caribou during all seasons and levels of abundance. The available data sampled caribou spatial distribution over a 50 year period including periods when population abundance was both high and low. The aerial surveys had systematic coverage (except in the 1990s) at relatively high coverage. The low sample size for the satellite-collared caribou, while a limitation, was offset by the collars covering a period of high numbers, a decline and a period of low numbers. The sites where

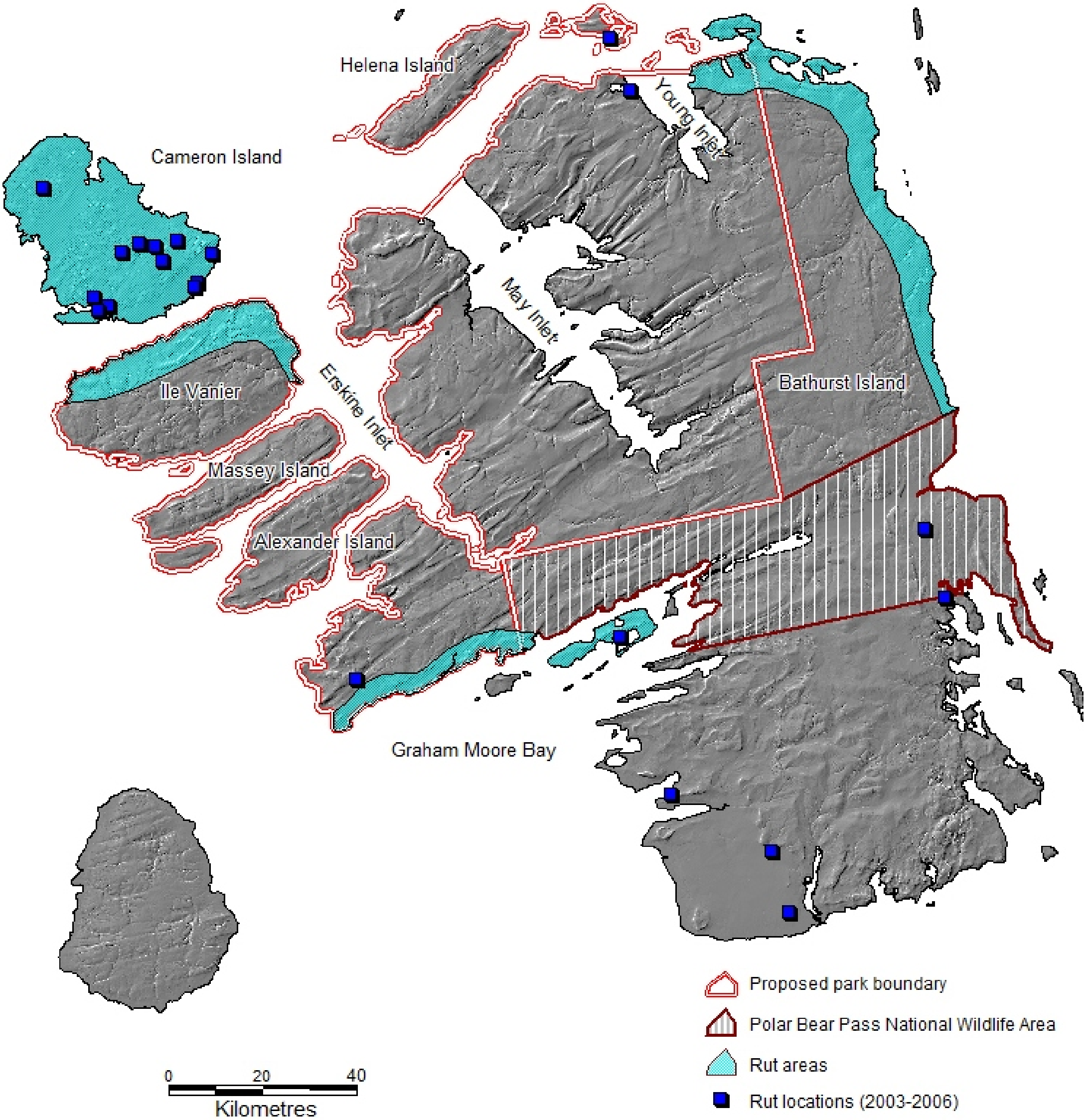

Figure 5. Rutting areas as obtained from 2003–06 collared cow Peary caribou. Shaded "rut areas" are from Miller (2001).

the caribou were collared were well-dispersed across Bathurst Island and included two of the northwestern satellite islands (Miller, 1997, 1998; Jenkins & Lecomte, 2012).

Data indicate the 2002 boundaries of the Park are effective for protection of calving areas. Miller (2002) mapped calving areas which are largely within the 2002 boundaries except the strip of coastal calving along the north coast of Bathurst Island. The 1994 calving locations of the five satellite collared cows were within those areas identified by Miller (n.d.). However, while 44% of calving sites on northern Bathurst for 2003–06 were within the areas mapped by Miller (2002), the other 56% were either on north Bathurst Island (within the 2002 boundaries) or along the Park's south boundary. Those calving sites had high between-year fidelity. It is difficult to assess whether the 2003–06 calving sites are a partial shift in calving distribution following the 97% decline in abundance, individual variability or small sample size.

The summer distribution of Peary caribou on the Bathurst Island Complex based on systematic aerial surveys (1961–1997) revealed that 71–88% of Peary caribou used northern Bathurst Island and the southern islands of the Governor General Islands. Unsystematic surveys also showed high use (80–94% of caribou observed) of northern Bathurst Island during July–August 1989–1998 during a shift from high to low abundance. Based on systematic surveys the proportion of caribou groups in the northeast stratum located outside of the proposed Park boundary varied from 10–45%, with the highest use during a late summer survey in 1981 when caribou densities were low. Use of the area outside of the proposed Park boundary is supported by the 1993–97 and 2003–06 collar data. Four of six home ranges (1993–94) and two of seven home ranges (2003–06) involved movements across the northeast Park boundary.

The information on the rut distribution is limited for the Bathurst Island Complex as aerial surveys were not timed for the pre-rut or rut. Mapping shed antlers and telemetry identified Cameron Island, as well as the north coasts of Vanier Island and Graham Moore Bay and the southern coast of Bathurst Island as rutting areas. The 2002 Park boundaries only include two of Miller's (2002) five areas and notably excludes Cameron Island.

Areas outside of the currently proposed Park boundary are important to the ecology of Peary caribou in the Bathurst Island Complex. During a series of severe, unfavourable winters (1994–97), collared caribou increased the size of their annual ranges with increased movements, and five individual home ranges expanded beyond the Park's boundaries. Mapping from Miller (2001) placed part of calving and rutting areas on northeastern Bathurst Island. Relatively low use of northeast Bathurst was detected in 2001 and 2013 (17–21%), but the surveys occurred in May and possibly caribou were moving from winter ranges.

The likelihood of seasonal movements by caribou across the southern Park boundary is based on late winter (March-April) systematic aerial surveys when about half the caribou groups were on southern Bathurst Island. However, this is based on only two systematic surveys of in the early 1970s after a die-off (Miller *et al.,* 1977). The use of southern Bathurst Island was expected as Inuit observations (Riewe, 1976) led Miller *et al.,* (1977) and Ferguson (1991) to suggest that southeastern Bathurst Island was the wintering range and northern Bathurst was the summer range; more recent local knowledge also showed winter use of southern Bathurst Island (Taylor, 2005) . Further support for this seasonal pattern was the directional movements of large groups moving south through Polar Bear Pass during the first two weeks of September 1970 and 1971 (Gray, 1998). At the individual scale, two of seven home ranges of satellite collared cows were on southern Bathurst Island and while their movements brought them into the PBPNWA, those movements did not include the Park.

The Park boundary should include Cameron Island. There are three lines of evidence that support the importance of Cameron Island as a rut area and winter range. Firstly, Miller (2002) identified Cameron Island as a rutting area based on sightings of cast male antlers; the 2003–06 collar data support this observation. Secondly, at the individual scale, five of 12 satellite-collared caribou during 1993–96 used Cameron Island in their annual range including winter; Fig. 2). Miller (2002) reported that between August 1993 and July 1994, a satellite-collared cow and a bull spent 24–46% of the year (during winter) on Cameron Island. Miller (n.d.) commented that while there were caribou with their annual range among the islands of the Governor General Islands, there were also caribou from Bathurst Island that moved to Cameron Island for the rut and winter. This is supported by both the annual and winter ranges for five of seven 2003–06 satellite collared cows; use of Cameron Island averaged 11% (± 2.3%) by these five individuals. The mapped winter range for the 2003–06 cows shows extensive use of Cameron Island (Fig. 4). Lastly, a high number (25%) of carcasses were recorded on Cameron Island after the 1994–97 severe winters. Considering that Cameron Island comprises only 3.8% of the Bathurst Island Complex land area, these three lines of evidence show disproportionately high use of the island in during the rut and winter, especially in years with extreme severe winters.

The distribution of vegetation complexes (Gould *et al.,* 2003) within the Bathurst Island Complex suggest that adding northeast Bathurst Island and Cameron Island would provide a greater diversity than the current proposed boundaries (Gunn *et al.,* 2012). These two areas contribute a greater proportion of cushion forb barrens – a vegetation type used to a moderate degree by caribou in the past (COSEWIC, 2004) – and Cameron Island has a higher proportion of prostrate dwarf shrub-graminoid tundra.

Currently, there is not enough information to assess the boundaries relative to any population structure and longer-term viability of Peary caribou on the Bathurst Island Complex, although fine-scale spatial and temporal genetic structure is likely (for example, Nussey *et al.,* 2005). Zittlau *et al.* (1999; unpubl. data) did not find differences based on nuclear DNA between cast antlers and carcasses found on Cameron Island, the east central coast and the southwest coast of Bathurst Island. A possible model is female philopatry and male-mediated gene flow as the males moved between the rut aggregations of the females from two neighbouring herds (Roffler *et al.,* 2012).

Over the longer-term, the effectiveness of the Park boundaries in maintaining ecological integrity including the population viability of Peary caribou will depend on many factors including the severity of any future declines and consequent population bottlenecks. Genetic variation for Peary caribou sampled in 1998 across the Bathurst Complex is lower than other caribou, possibly from a genetic bottleneck resulting from the 1973–74 die-off (Zittlau *et al.,* 1999; unpubl. data). Consequences of low genetic variation, such as in-breeding depression, are usually considered less likely than demographic risk, but the relevance of this to Peary caribou is uncertain given the extent of, for example, the 1994–97 declines from an estimated 3,000 caribou to less than 100, which suggests that evolutionary selection is extreme and the survivors may be a particular sub-set of the population (see Sinclair *et al.,* 2003). It is unknown over the longer term how the boundaries of a national park could affect the likelihood of dispersal and the scale necessary to minimize population fragmentation.

A likely significant factor in assessing boundaries for a national park will be climate change. Understanding influences of a changing climate on Peary caribou distribution is complex with many interacting changes. Later formation and earlier break up in the extent of land fast ice is already measurable in the western Queen Elizabeth Islands including the Bathurst Island Complex (Galley *et al.,* 2012) which will have implications for inter-island movements. Although Peary caribou swim between the closer islands (Miller, 1995b), changes in the timing of land fast ice and the greater distances between Bathurst and Cameron islands, for example, may have an impact.

Our assessment from 1961–2013 sampled the known range of natural variability for distribution and abundance on the Bathurst Island Complex. We found that the 2002 Park boundaries are crossed in the northeast, and in the northwest to Cameron Island, by the seasonal movements of a relatively large proportion of Peary caribou, even though Peary caribou have been through three peaks of abundance and two die-offs and associated periods of low abundance. The 2002 boundaries of the Park are more effective for protection of calving but the distribution of caribou suggest that northeast Bathurst Island, including outside of the proposed boundaries, are important during summer, both Cameron Island and northeast Bathurst Island are rutting areas with relatively high use, and Cameron Island is important during winter. Our analyses of the available information indicate that Qausuittuq National Park boundaries which include Cameron Island and the northeast coast of Bathurst Island will be more effective in contributing to the persistence of Peary caribou on the Bathurst Island Complex during most seasons and at differing population levels.

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Biodiversity offsets and caribou conservation in Alberta: opportunities and challenges

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Abstract: The federal recovery strategy for boreal woodland caribou (*Rangifer tarandus caribou*) sets a goal of self-sustaining populations for all caribou ranges across Canada. All caribou herds in Alberta are currently designated as not self-sustaining and the recovery strategy requires an action plan to achieve self-sustaining status. At the same time, continued natural resource extraction in caribou ranges may be worth hundreds of billions of dollars. Some regulatory bodies have recognized an opportunity for biodiversity offsets to help meet the caribou recovery strategy's goals while still permitting economic benefits of development. In this review, we evaluate offset opportunities for caribou in Alberta and practical impediments for implementation. We conclude that a number of actions to offset impacts of development and achieve no net loss or net positive impact for caribou are theoretically feasible (i.e., if implemented they should work), including habitat restoration and manipulations of the large mammal predator-prey system. However, implementation challenges are substantial and include a lack of mechanisms for setting aside some resources for long periods of time, public opposition to predator control, and uncertainty associated with loss-gain calculations. A framework and related policy for offsets are currently lacking in Alberta and their development is urgently needed to guide successful design and implementation of offsets for caribou.

Key words: Alberta; biodiversity offsets; conservation; habitat restoration; no net loss; woodland caribou.

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Introduction

In an effort to halt escalating global biodiversity loss caused by human development activities, new development projects are increasingly required to achieve no net loss (NNL) or net positive impact (NPI) for biodiversity. Such requirements come from a variety of sources, including governments, lending institutions, and even the corporate sector. Examples of government-driven NNL or NPI requirements are numerous and include the United States wetland policy (Environmental Law Institute, 2002), Canada's fish habitat compensation policy (Pearson *et al.,* 2005), and France's no net loss policy for biodiversity (Quétier *et al.,* 2014). The International Finance Corporation's Performance Standard 6 requires that developers demonstrate an approach to achieve NNL for biodiversity in natural habitats and NPI in critical habitat prior to obtaining a loan from the World Bank (IFC, 2012a), and many other large lending institutions have similar requirements. Several resource extraction and manufacturing companies, including large mining companies such as Teck and Rio Tinto, have adopted voluntary corporate NNL or NPI policies with respect to the impacts of their operations on the environment; by 2013 at least 32 companies had made public commitments to NNL or NPI (Rainey *et al.,* 2014).

Biodiversity offsets are a key mechanism for achieving NNL or NPI and addressing some of the world's most pressing conservation problems. Biodiversity offsets are defined by the Business and Biodiversity Offsets Programme (BBOP) as *"measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken"* (BBOP, 2012a). Offsets can be applied to biodiversity as a whole, but are frequently applied to individual biodiversity elements, such as a habitat type or even individual species, depending on what biodiversity elements are significantly impacted (IFC, 2012a; Bull *et al.,* 2013b). Actions to achieve offsets occur along a continuum of compensation for adverse impacts, but compensation must reach a minimum of NNL before a true offset is achieved (BBOP, 2012a).

Offsets should be employed only as the final step of the standard mitigation hierarchy for development projects, which includes avoiding adverse effects where possible, minimizing adverse effects to the extent feasible, restoring biodiversity to the extent practicable through restoration, and finally compensating for any residual effects that could not be otherwise mitigated (Kiesecker *et al.,* 2010; Saenz *et al.,* 2013). Offsets should achieve long-term conservation outcomes (IFC, 2012b), lasting at least as long as the impacts from the project (Bull *et al.,* 2013a). Offsets are typically achieved through three primary types of actions: financial mechanisms, protecting existing biodiversity that might otherwise be lost, and enhancing existing biodiversity through management actions (Poulton, 2014).

Biodiversity offsets have received much attention over the last decade and are increasingly advocated as a conservation tool (Bull *et al.,* 2013a). However, with the exception of compensation for fish habitat, offsets have not been widely applied in Canada. This is now changing with increasing pressure for application of offsets in Alberta (Dyer *et al.,* 2008; ABCOG, 2009), and regulatory requirements to maintain critical habitat as defined under Canada's *Species at Risk Act*.

Conservation of boreal woodland caribou ([*Rangifer tarandus caribou*], hereafter caribou), which are a species listed federally as 'Threatened' on Schedule 1 of the *Species at Risk Act* (Species at Risk Public Registry, 2014) and provincially as "At Risk" (AESRD, 2010), is among the most pressing issues for which offsets are being considered in Alberta (Habib *et al.,* 2013). Most of Alberta's caribou populations are declining rapidly (Hervieux *et al.,* 2013), generating international attention and numerous appeals to limit development in Alberta's caribou ranges (e.g., ALT, 2009; Boutin, 2010; Ethical Consumer, 2010). On the other hand, development in Alberta's caribou ranges contributes significantly to the Canadian economy and opportunity costs associated with protecting all of Alberta's caribou ranges (i.e., avoiding all future impacts) have been estimated to be in excess of 100 billion dollars (Schneider *et al.,* 2010).

In theory, offsets designed to achieve NNL or NPI for caribou could simultaneously support both development and caribou conservation; however, such offsets may prove difficult to achieve in practice. Demonstrations of NNL or NPI for biodiversity as a consequence of large development projects are few, and offset implementation rarely meets the conceptual principles applied during planning (Fox & Nino-Murcia, 2005; Bull *et al.,* 2013a). In Canada, for example, 67% of fish habitat compensation failed to meet objectives and resulted in net losses of habitat area (Quigley & Harper, 2006a). Moreover, government NNL or NPI policies may not be successful because organizations to govern such policy or institutional frameworks to evaluate and enforce them have not been created, and/or because the appropriate legal framework permitting implementation of offset requirements is not in place (Quétier *et al.,* 2014).

The purpose of this review is to better understand the opportunities and challenges associated with developing a caribou offset strategy in Alberta and improve the potential for successful offset applications. We have organized our review into sections presenting: 1) the causes of caribou decline in Alberta, 2) a review of recent regulatory mechanisms and decisions recommending or requiring caribou offsets, 3) a theoretical discussion of offset opportunities for caribou, 4) an investigation of the practical challenges associated with implementing caribou offsets, and finally 5) our conclusions and recommendations for a successful offset application.

Causes of caribou decline in Alberta

In order to identify opportunities to efficiently and effectively offset adverse effects of a development project to a particular biodiversity value, one must first understand the proximate and ultimate drivers of change for that value, even when they may not be immediately linked to activities of a project. In the case of caribou in Alberta, substantial research over the last 3 decades has clearly identified causes of rapid decline (Hervieux *et al.,* 2013). To achieve measurable conservation outcomes of NNL or better for a development project in a cost effective way, offsets must focus efforts on addressing the most important of these causes.

Available evidence overwhelmingly indicates that predation is the primary proximate factor limiting caribou populations (Bergerud, 1988; Stuart-Smith *et al.,* 1997; James & Stuart-Smith, 2000; Boutin *et al.,* 2012; Hervieux *et al.,* 2013; Latham *et al.,* 2013). Increased predation can largely be attributed to a phenomenon known as apparent competition (Holt, 1977) where growing number of predators, such as wolves (*Canis lupus*), increase in tandem with the number of primary ungulate prey, such as white-tailed deer (*Odocoileus virginianus*), to the detriment of secondary prey species, such as caribou. White-tailed deer have increased substantially in northeast Alberta in recent decades (Dawe, 2011; Latham *et al.,* 2011a), resulting in a near doubling of the wolf population in the west side of the Athabasca River caribou range from approximately 6 wolves/1000 km^2 in the mid 1990's to more than 11 wolves/1000 km^2 in 2005-2009 (Latham *et al.,* 2011a), and probably in most other caribou ranges (Hervieux *et al.,* 2013). Other predators such as cougars (*Puma concolor*) and black bears (*Ursus americanus*) may also contribute to caribou declines in some places, and white-tailed deer increases have been linked to cougar population increase and expansion in Alberta, including into caribou range (Knopff *et al.,* 2014).

Although predation is the proximate cause of decline, changing predator-prey dynamics in Alberta's caribou ranges are ultimately driven by broader landscape-level habitat changes caused by agriculture, forestry, oil and gas, and climate change, which are creating increasingly favorable conditions for species like white-tailed deer (Dawe, 2011; Boutin *et al.,* 2012). Specifically, anthropogenic development is creating more early successional or other habitats with higher forage potential for non-caribou ungulates. In addition to increasing the number of predators on the landscape, anthropogenic development in caribou range may facilitate increased wolf movements into caribou habitats along linear features such as roads, pipelines and seismic lines, which can increase encounter rates between wolves and caribou, thereby increasing predation (Latham *et al.,* 2011b; DeCesare, 2012; McKenzie *et al.,* 2012). Wolf use of linear features to make forays into caribou habitat may be especially high during summer when wolves focus hunting efforts on prey such as beavers (*Castor canadensis*) that tend to be found in the same habitats as caribou (Latham *et al.,* 2013).

Less well understood are possible changes in caribou carrying capacity as a result of anthropogenic development. Forage availability is, of course, fundamental to the persistence of caribou populations (Darby *et al.,* 1989). Some studies have found that caribou avoid anthropogenic disturbances such as seismic lines, roads, or forestry cutblocks (Dyer *et al.,* 2001), and cutblocks in particular may reduce available forage over long periods of time (Herbert & Weladji, 2013). Other developments such as oil sands mines completely remove caribou foraging opportunities over large areas. However, forage quantity and quality probably does not limit non-migratory caribou populations (McLellan *et al.,* 2012). Indeed, because caribou may avoid areas of high forage quality that also have high predation risk (Briand *et al.,* 2009), addressing the predation problem could both improve survival and recruitment and provide recovering caribou populations with additional access to high quality forage resources.

Consequently, to achieve measureable conservation benefits for caribou, caribou offsets focused on managing large mammal predator-prey dynamics in caribou range may prove to be most effective. Until the predation problem has been addressed, actions focused on creating new habitats, more forage, or otherwise increasing landscape-level carrying capacity may fail to improve caribou conservation prospects in the short term. Over the long term, creating new habitats through restoration activities will be essential to address the ultimate cause of caribou decline (Hervieux *et al.,* 2013).

Regulatory requirements for caribou offsets

In 2012, the Government of Canada released a federal recovery strategy that aims to achieve "self-sustaining local populations in all caribou ranges throughout their current distribution in Canada, to the extent possible" (Environment Canada, 2012a, p. 19). The status of 51 identified local caribou populations include 26 that are "not self-sustaining", 10 that are "as likely as not self-sustaining", 14 that are "self-sustaining", and 1 is "unknown". Where populations are not self-sustaining, the federal strategy dictates that recovery actions be implemented. All of the populations in Alberta (n=12) are considered "not self-sustaining" (Environment Canada, 2012a).

To achieve acceptable probability of a self-sustaining caribou population (i.e., 60%), the recovery strategy sets a target of at least 65% undisturbed habitat within each caribou range. The proportion of undisturbed habitat within Alberta's 12 caribou ranges varies between 5% and 43% (Environment Canada, 2012a). Because critical habitat for caribou has been identified and a disturbance threshold within critical habitat has been set, developments can theoretically be stopped under Canada's *Species at Risk Act* should the development compromise the ability of a range to maintain or be restored to 65% undisturbed habitat.

The recovery strategy is based on a habitat disturbance threshold because the probability of a population being self-sustaining is linked to the proportion of disturbed habitat contained within its range (Environment Canada, 2012a). Environment Canada (2011) concluded that the percentage of range disturbed, defined as all anthropogenic disturbances plus a 500 m buffer and all areas burned in the last 40 years, best explained the variation in calf recruitment across 24 ranges, which probably also reflects the extent to which the predator-prey system has changed. The 500 m buffer around anthropogenic disturbances not only captured the effects of habitat loss but also those related to fragmentation and spatial configuration of disturbances (Environment Canada, 2011). Hervieux et al.'s (2013) evaluation of caribou demographics in Alberta supported Environment Canada's habitat-based approach linking range condition and population viability. Habitat creation through caribou offsets for each newly approved project is one way to work towards achieving recovery strategy habitat intactness goals (i.e., if NPI is achieved), or at least to prevent further reductions of undisturbed habitat within critical habitat ranges already below 65% undisturbed habitat (i.e., if offsets achieve NNL).

Some regulatory bodies within Canada have begun to request or require caribou offsets as part of review panel recommendations or approval conditions for new development projects in caribou ranges. Recently issued project approvals recommending or requiring offsets for caribou are summarized in Table 1. Guidance from the National Energy Board on offset plan design has evolved with each project decision as more detailed requirements have been included in project decisions over time. Projects reviewed by joint federal and provincial panels have offset considerations limited to recommendations, as opposed to conditions.

Offset opportunities

In theory, actions to achieve a caribou offset can take a variety of forms. Based on our review of the causes of caribou decline, we investigated four types of actions that might achieve a caribou offset: 1) protecting existing caribou habitat that might otherwise be lost, 2) restoring disturbed caribou habitat, 3) manipulating the predator-prey system to reduce predation rates, and 4) in lieu fees.

Regardless of which type of action or combination of actions is used, at least NNL must be demonstrated to achieve an offset. Demonstrating NNL or NPI entails some method of measuring losses as a result of development and gains as a result of conservation actions (Quétier & Lavorel, 2011; BBOP, 2012b). The ideal measure, or currency, to use for offsets focused on a single species, such as caribou, is the number of individuals in the population (Doherty *et al.,* 2010), or a surrogate that accurately reflects this. Gains measured using the selected currency must demonstrate additionality relative to a counterfactual in an amount equal to or greater than the losses incurred from the project. Additionality means that the conservation actions undertaken as part of a development project must be over and above actions planned without the project (BBOP, 2013). A counterfactual is a measurement of what might have occurred without implementation of the conservation action (Bull *et al.,* 2014).

Protecting existing caribou habitat that would otherwise be impacted achieves what is known as an averted loss offset. Averted loss offsets can be problematic because they would still result in a net decline in caribou populations relative to existing conditions (Maron *et al.,* 2012). However, averted loss offsets still produce an advantage compared to a case where all development were to proceed, and such offsets can be acceptable where background rate of habitat loss is high and protective legislation and mandatory compensation policies are not in place (Gibbons & Lindenmayer, 2007; Maron *et al.,* 2012).

The most common method of achieving an averted loss offset is to apply some mechanism of permanent protection to land that is other-

Table 1. Project approvals during 2011-2013 recommending or requiring the implementation of caribou offsets in Alberta.

Project	Project Description	Regulator	Location	Offset Requirements	Source
NOVA Gas Transmission Ltd. Horned River Project	Construction of a new natural gas pipeline consisting of approximately 72 km of 914 mm outside diameter pipe, 2.2 km of 610 mm outside diameter pipe, and related facilities.	National Energy Board	NW Alberta / NE British Columbia	NGTL shall file with the Board, prior to requesting leave to open, a plan which describes measures to offset unavoidable and residual impacts to boreal woodland caribou habitat identified by NGTL within the Footprint. For the purposes of this Project, offset measures for boreal woodland caribou do not include actions that require land acquisition, replacement or substitution of habitat, habitat compensation, terrestrial no-net-loss measures or the regional application of mitigation strategies.	NEB, 2011
NOVA Gas Transmission Ltd. Northwest Mainline Project	Construction of three new natural gas pipeline loops totaling 111.2 km, of which 103.8 km is alongside existing disturbances. Project requires a minimum 32 m wide right of way for its entire length.	National Energy Board	NW Alberta / NE British Columbia	NGTL shall file with the Board for approval a plan to offset all unavoidable and residual Project-related effects to caribou habitat within the Chinchaga caribou range. The plan shall describe measures that would offset all effects identified in the quantitative and qualitative assessment to be conducted as part of Condition 7(b)(vi)[1]. NGTL shall file with the Board for approval […] a plan for monitoring the caribou habitat restoration and offset measures implemented […].	NEB, 2012a
NOVA Gas Transmission Ltd. Leizmer to Kettle River Crossover Project	Construction of 77 km of new natural gas pipeline and associated facilities. Approximately 55 km parallels existing linear disturbances.	National Energy Board	NE Alberta	NGTL shall file with the Board for approval […] a preliminary and final versions of a plan to offset all residual Project-related effects resulting from directly and indirectly disturbed caribou habitat, after taking into account the implementation of the EPP [environmental protection plan] and CHRP [caribou habitat restoration plan] measures. NGTL shall file with the Board for approval […] a program for monitoring and verifying the effectiveness of the caribou habitat restoration and offset measures implemented as part of the CHRP and Offset Measures Plan.	NEB, 2012b

Table 1 continued.

Project	Project Description	Regulator	Location	Offset Requirements	Source
Shell Canada Energy Jackpine Mine Expansion Project	Amendment to expand an existing open pit oil sands extraction mine. Addition to include additional mining areas, and associated processing facilities, utilities and infrastructure. Increase production capacity by 15,900 m^3/day dry bitumen.	Alberta Energy Regulator and Canadian Environmental Assessment Agency (joint review panel)	NE Alberta	The Panel recommends that […] the Governments of Canada and Alberta consider the need for conservation offsets to further mitigate project effects. The potential use of conservation offsets should include a consideration of the need to compensate for project effects to wetland-reliant species at risk and migratory birds that are wetland-reliant or species at risk. […] The Panel further recommends that during development of the biodiversity management framework, consideration be given to principles such as no net loss of caribou habitat, limiting linear disturbances in critical caribou habitat, and restoration of historical and present caribou ranges.	AER & CEAA, 2013
Northern Gateway Pipelines Inc. Northern Gateway Pipeline Project	Construction of a new oil product pipeline spanning 1,178 km and associated facilities. Project application identified a 1 km wide right of way along the entire route.	National Energy Board	Alberta / British Columbia	Northern Gateway must file with the NEB for approval […] an Offset Measures Plan for each affected caribou range to offset all residual Project-related effects resulting from directly- and indirectly-disturbed caribou habitat […] Northern Gateway must file with the NEB for approval […] a program for monitoring and verifying the effectiveness of the caribou habitat restoration and offset measures implemented as part of the CHRP and Offset Measures Plan.	NEB, 2013

Notes: NGTL = NOVA Gas Transmission Ltd.; NEB = National Energy Board; CHRP = caribou habitat restoration plan; NE = north-east; NW = north-west.

[1] Condition 7(b)(vi) refers to a quantitative and qualitative assessment of the area of caribou habitat within the Chinchaga caribou range that was directly and indirectly disturbed as a result of construction of the Project. The assessment shall identify and assess the caribou habitat to be mitigated for as a result of the implementation of the Caribou Protection Plan and Caribou Habitat Restoration Plan, as well as identify the remaining residual effects.

wise under legitimate threat of disturbance. For caribou in Alberta this type of offset could be achieved by protecting land that occurs in caribou range and would otherwise be developed. A theoretical example of an averted loss offset would be an oil sands development identifying and protecting an area of caribou habitat that otherwise would be disturbed by an approved forestry operation and that supports an equal or greater number of caribou than the land disturbed by the oil sands development.

Active management interventions are required to achieve caribou offsets that provide NNL or NPI relative to baseline conditions. Caribou populations are linked to their habitat (Environment Canada, 2012a), much of which has been previously disturbed and has not been reclaimed. Restoration efforts to offset the impacts of a new development project on caribou populations can target historic disturbances. This includes reclaiming historic disturbance on public lands, but may be especially effective in areas that were previously disturbed but now reside in newly created conservation areas protected under provincial land-use plans, such as the Lower Athabasca Regional Plan (Government of Alberta, 2012).

Given that anthropogenic disturbances and areas deforested by wildfires within the last 40 years cover 57 to 95% of caribou ranges in Alberta (Environment Canada, 2012a), we can confidently assume that there exist many opportunities to reclaim forested habitat preferred by caribou in each of Alberta's 12 caribou ranges. Habitat restoration can be implemented using different techniques depending on disturbance type and local site conditions. Treatments such as mounding, tree planting, tree/shrub transplanting and spreading of coarse woody debris can accelerate natural reforestation or encourage reforestation on sites that otherwise might remain a shrubland or grassland (Coupal & Bentham, 2014).

Reclaiming historic linear features such as trails, seismic lines, and abandoned roads is a logical first step given their prominence on Alberta's landscape and potential importance for caribou predator-prey relationships (Latham *et al.,* 2011b), but reclaiming any disturbance in caribou range would likely count towards an offset. To meet the requirement of additionality, restoration activities must target disturbances outside the proponent's approved project footprint that have not recovered either due to environmental conditions (e.g., cold, wet soils) and/or historical clearing and restoration practices, such as mulched seismic lines, admixing of soils during facilities construction, low-blading during access clearing, and seeding of grasses on reclaimed areas.

Offsets achieved by active restoration of previously disturbed areas away from the project can be implemented directly by project proponents, or provided through a conservation bank managed by a third party. A conservation bank is an offset generated by a third party that develops and controls the offset and subsequently sells it, in whole or in part, to developers. Conservation banks provide an opportunity to combine habitat protection (i.e., averted loss), restoration (e.g., reforestation), and enhancement (e.g., reduced white-tailed deer density), and most importantly, would ensure the creation of offset credits before development occurs. Third party offset banking is the preferred offset approach under the US federal wetland compensatory mitigation system (Gardner *et al.,* 2009), and could be applied to caribou.

Most offsets focus on conservation actions that benefit habitat, and habitat restoration is likely the only way to address the ultimate causes of caribou population decline and achieve the federal recovery strategy goal of self-sustaining caribou herds (Hervieux *et al.,* 2013). However, because predation is the central proximate cause of caribou decline, predator management may be required to stop caribou declines in the short term (Boutin *et al.,* 2012; Hervieux *et al.,* 2013; Hervieux *et al.,* 2014), and offsets focusing on actions to reduce predation may have some of the strongest immediate benefits for caribou.

Although not a traditional habitat-based offset, actively managing interactions between caribou and their predators addresses the most immediate threat to caribou. Similar non-traditional offset actions have been proposed elsewhere. For example, impacts of unintentional bycatch of seabirds from the fishing industry may not be best addressed by focusing on the industry itself; instead knowing that a much greater source of seabird mortality results from nest predation from invasive predators provides opportunities to deliver offsets using predator control (Pascoe *et al.,* 2011). Another proposed application of non-traditional offset is to fund anti-poaching efforts to improve conservation prospects for saiga antelope (*Saiga tatarica*) in Uzbekistan and Kazakhstan, where traditional protected area offsets cannot achieve NNL because saiga migrate over large areas and the primary cause of decline is poaching, not habitat loss (Bull *et al.,* 2013b).

Managing interactions between caribou and their predators can take a variety of forms. One option is to take action to directly reduce wolf populations. Wolf control measures, including aerial gunning and poisoning, have been implemented by the Government of Alberta in the Little Smoky Caribou Range since winter 2005-2006 (ASRD and ACA, 2010; Hervieux *et al.,* 2014). These measures appear to have been effective; the Little Smoky caribou population growth rate increased and the population has stabilized (Hervieux *et al.,* 2013; Hervieux *et al.,* 2014).

Environment Canada (2012a) and the Government of Alberta (2011) highlight that maintenance and recovery of caribou is unlikely to succeed without the implementation short-term predator management. By contributing to such efforts, developers can partially offset some of the adverse impacts they have on caribou. This type of management action would need to be implemented by the appropriate government body (i.e., Fish and Wildlife Division) as project proponent and third parties do not have jurisdiction over wildlife management.

Increasing wolf populations in northern Alberta appear to be driven by invading white-tailed deer (Latham *et al.,* 2011a). Consequently, direct control measures also could be applied to white-tailed deer, which should elicit a numerical response in the wolf population and presumably reduce the predation pressure on caribou (Serrouya, 2013; Wittmer *et al.,* 2013; Serrouya *et al.,* 2015). However, we caution against the use of primary prey reductions as an offset mechanism in isolation of other management actions such as wolf control. Without simultaneously controlling wolves, reducing non-caribou ungulate populations could cause wolves to switch to killing more caribou before their numbers fall, exacerbating instead of relieving predation pressure on caribou (Wittmer *et al.,* 2013; Serrouya *et al.,* 2015). Similar to wolf control, reductions in white-tailed deer would require implementation by government; however, hunters could play a key role if the approach involves liberalizing harvest quotas.

Another method of changing predation rates on caribou is the use of predator fencing. Caribou calf survival is typically low, with the highest rate of mortality occurring in the first month after birth (e.g., Stuart-Smith *et al.,* 1997; Mahoney & Virgl, 2003; Gustine *et al.,* 2006). Juvenile recruitment rates are important determinants of population dynamics (Gaillard *et al.,* 1998); in Alberta caribou recruitment is very low due to high predation, ranging from 0.100 to 0.206 calves/cow (Hervieux *et al.,* 2013). Improving calf survival could therefore constitute an offset action. By corralling female caribou into maternity pens, equipped with predator-proof fencing, while they give birth and for the first few months of the newborn calf's life, recruitment may improve. Maternity pens have been previously implemented in Alberta (Smith & Pittaway, 2011), the Yukon (CCRT, 2010), and in British Columbia, where pens have been developed through partnership among industry, First Nations, and the provincial government (Hume, 2014). Industry contribution to such a project may contribute to or meet offset requirements for a new development, depending on the magnitude of the development and of the benefit from the maternity pen. Although not without risk, predator fencing also might be extended beyond maternity pens to encompass larger areas that could support the entire caribou lifecycle, similar to conservation fencing implemented in other parts of the world (Hayward & Kerley, 2009).

The recovery goal for caribou targets self-sustaining populations in each caribou range. Active management of predator-prey interactions by controlling predators and non-caribou ungulates and constructing fences may sustain a caribou population artificially, but when active management ceases, caribou decline towards extirpation may begin anew. In such cases, caribou populations would not meet the self-sustaining requirement of the federal recovery strategy. Predator control can be part of an offset strategy, but is an interim solution to a problem that requires substantial change in habitat such that the carrying capacity for non-caribou ungulates and the predators they feed is significantly reduced. Reclaiming disturbed habitats to mature forests that support caribou but contain minimal forage for other ungulates would be required with predator control.

Lastly, in lieu fees represent payments set by a regulator and made by a developer according to a predetermined fee schedule to finance actions that lead to an offset. Such payments are convenient for developers because costs of the offset are clearly defined up front and the developer is not responsible for designing or implementing the offset. In lieu fees can work to achieve a caribou offset, as defined by BBOP (2012a), only if they fund actions that achieve a NNL or better conservation outcome. Consequently, these financial mechanisms require the regulator, or a third party, to implement actions already described. Although in lieu fees have not been formally identified as an option for caribou in Alberta, the Government of British Columbia has proposed payment of a predetermined fee per hectare of caribou habitat disturbed as an offset mechanism for future development projects (MFLNRO, 2012). The proposed amount paid increases from \$1250/ha to \$10 000/ha as one moves from 'low value' caribou habitat to 'very high value' caribou habitat (MFLNRO, 2012).

Implementation challenges

Achieving NNL or NPI for caribou through careful application of the mitigation hierarchy, including offsets, represents an ambitious and laudable environmental goal and, in theory, there are several actions that might be implemented to achieve this for caribou. However, as good as offset theory may be, implementing offsets in practice has proven challenging. Measurable conservation outcomes that achieve NNL and NPI have rarely been demonstrated (Quigley & Harper, 2005 and 2006b; Burgin, 2010). Indeed, the concept of biodiversity offsets has recently been criticized by academics and non-government organizations (NGOs) for achieving the opposite of what it intends; instead of biodiversity conservation, offsets sometimes create a "license to trash" because developers receive approvals for their developments based on a promise to offset that cannot be realized or for which actions are not appropriately implemented (e.g., ten Kate *et al.,* 2004; Matthews & Endress, 2008; Walker *et al.,* 2009; Burgin, 2010).

Failure to implement appropriate action to achieve an offset can have a variety of causes, and effective implementation of caribou offsets in Alberta requires that these are overcome. Potential problems include inconsistent interpretations of NNL (Gardner *et al.,* 2013); lack of information required to clearly assess and quantify project impacts (Brownlie *et al.,* 2013); failure to identify impacts that cannot be offset under any circumstance (Norton, 2009; Bull *et al.,* 2013a); inappropriate use of metrics or currencies (Quigley & Harper, 2006b; Walker *et al.,* 2009; Doherty *et al.,* 2010; Bull *et al.,* 2013a; Gardner *et al.,* 2013); non-compliance with regulations and lack of enforcement (Quigley & Harper, 2006a; Matthews & Endress, 2008; Norton, 2009); failure to use appropriate offset ratios (Quigley & Harper, 2006b; Moilanen *et al.,* 2009); implementation without prior evidence of technical feasibility or effectiveness (Gibbons & Lindenmayer, 2007); inadequate regulatory framework and government oversight (Quétier *et al.,* 2014); and lack of monitoring and maintenance (Brown & Lant, 1999; Quigley & Harper, 2006b).

Perhaps the most important challenge that applies to all offset opportunities summarized in this paper is that there currently is no clear guidance or framework for offset requirements in Alberta. Provision of key design elements for offsets provided through a comprehensive offset framework and policy would greatly improve the effectiveness of offset implementation (e.g., Quétier *et al.,* 2014). Proponents requiring offsets in Alberta currently have little guidance on basic standards and performance criteria such as (i) offset currency, (ii) loss-gain calculations, (iii) equivalency and 'trading up' (e.g., Bull *et al.,* 2013a; Habib *et al.,* 2013), (iv) uncertainty and time lags, (v) duration, and (vi) monitoring requirements and appropriate indicators.

An offset policy is required to identify what actions constitute permissible offsets and to ensure that offsets are consistently applied across development projects. Recent applications of caribou offsets in Alberta have been either voluntary or individual offset plans required by regulators (Poulton, 2014). Although these one-off project-specific offsets can effectively compensate for impacts to caribou at the local project scale, they are unlikely to contribute to broader landscape conservation strategies and outcomes if they are not coordinated with regional plans or initiatives (Kiesecker *et al.,* 2010). On the other hand, if offset requirements are regulated and standardized, land managers can more readily incorporate offset actions into broader initiatives (Poulton, 2014). Previous consideration of offsets in Canada (DFO, 1986; Government of Canada, 1991; Lynch-Stewart, 1996; Environment Canada, 2012b; DFO, 2013; Poulton, 2014), could serve as useful building blocks for policy makers and environmental practitioners tasked with developing effective and efficient offset plans for caribou.

Even if a regulatory framework were established, implementation of theoretical options for caribou offsets is not straightforward. Financial mechanisms, for example, typically include fees paid to support caribou and wolf monitoring programs, maternal pens, as well as other research and outreach programs (MFLNRO, 2012). Although worthwhile endeavours, monitoring, research, and outreach do not typically deliver measurable conservation outcomes and therefore do not provide offsets. Although the simplest option for developers, the risk associated with financial offsets is that the funds do not deliver the direct conservation outcomes required to achieve NNL or NPI. Adequate means of defining on-the-ground benefits of actions implemented using funds generated from financial offset requirements are needed to demonstrate success of this approach, but guidance for achieving this for caribou is not currently available. The US regulations governing compensatory mitigation for wetlands and other aquatic resources can provide useful guidance for a payment program (Gardner *et al.,* 2009). Achieving specific milestones and performance standards prior to the release of offset credits is a central consideration of the regulation and emphasizes the importance of timely compensatory actions (Gardner *et al.,* 2009).

Averted loss offsets established by protecting caribou habitat that otherwise would be disturbed are even less straightforward, to the point of being nearly impossible under current provincial legislation. Caribou range in Alberta is almost entirely restricted to public lands which cannot be purchased. The exception is a small tract of private land located in the Chinchaga caribou range in northwest Alberta (http://thecarbonfarmer.ca). There is no established conservation banking system in the province, and even if one existed, private lands that could be purchased and protected or improved primarily occur outside of caribou range and would not benefit caribou.

Extinguishing development rights on public land in Alberta also is prohibited and averted loss offsets cannot be achieved by one industry paying for another not to develop (e.g., an oil and gas company cannot purchase development rights from a Forestry Management Agreement holder). The petroleum and natural gas mineral rights leasing system, guided by the *Mines and Minerals Act*, requires all lease holders to 'prove the mineral resource' by drilling, production or technical mapping. Failure to do so can result in loss of the lease. Some oil and gas producers in the province have indicated that provincial regulators do not always consider technical mapping an acceptable means of delineating the resource, which encourages more drilling and therefore more habitat loss (CAPP, 2013). Hence, setting aside or otherwise not developing an oil and gas lease in caribou range as an averted loss offset for impacts elsewhere does not appear to be an option.

Inability to implement averted loss offsets may not be a substantial constraint for caribou conservation because such offsets do not typically provide NNL or NPI relative to existing conditions; hence, they cannot contribute to the net restoration of habitat required to achieve the caribou recovery strategy objective of achieving self-sustaining caribou herds.

Habitat restoration is the best approach to address ultimate causes of caribou decline and is currently being implemented in Alberta (Coupal & Bentham, 2014). Key problems with habitat restoration are the long time-lag before caribou benefit from the action and uncertainty about loss-gain calculations, restoration success, and the potential for future development programs where habitats have been reclaimed. Boreal forests grow slowly and even with extensive restoration, readjustments to predator-prey systems that are driven by landscape change at broad spatial extents (i.e., north-eastern Alberta) probably will take decades. Despite Environment Canada's (2012a) support for habitat restoration, they fail to provide a formal definition of restored caribou habitat, pointing to a need to develop targets and measureable criteria for restoration. Such targets and criteria are required to determine when an offset is realized.

Some guidance for offset currency is provided by the federal recovery strategy, which uses 65% undisturbed habitat within a caribou range as a surrogate for achieving a self-sustaining population (Environment Canada, 2012a). Surrogate currencies can be useful, but may have risks associated with them if they are not closely linked to the desired outcome (e.g., self-sustaining caribou populations). For example, reclaiming linear features may provide the greatest value for a developer's investment in terms of demonstrating an offset using loss-gain calculations derived from the federal caribou recovery strategy habitat models, but this may not translate into an equal benefit to caribou. Consider a 5 m wide seismic line that extends over 1000 m with no other disturbances nearby. Because caribou range disturbance metrics in the federal recovery strategy were calculated by applying a 500 m buffer on either side of that seismic line, successful restoration returns 5000 m^2 (i.e., 5 m x 1000 m) of forest, but 1 010 000 m^2 of caribou habitat (i.e., ([500 m x 1000 m] + [5 m x 1000 m]) x 2). Reclaiming 1000 m of seismic line will likely benefit caribou populations, but perhaps not by a factor of over 200 for every habitat unit reclaimed.

Uncertainty around loss-gain calculations and restoration success, along with any time-lags prior to achieving functional habitat also may require offset multipliers that are unachievable, depending on how loss-gain calculations are implemented. Curran *et al.,* (2014) suggest ratios required to achieve a true NNL offset through habitat restoration may be as high as 100:1, which is much higher than typically applied ratios less than 10:1 and probably cost-prohibitive for most development projects in Alberta. A better option to address time lags is to have an offset policy requiring demonstrated conservation outcomes ahead of development, which would reduce the required ratio (Gardner *et al.,* 2009; Maron *et al.,* 2012).

There is also the matter of where to implement restoration (or any other offset action). In the context of oil and gas development, a developer may choose to implement offsets by conducting on-lease or off-lease habitat restoration. On-lease restoration provides more certainty that restoration efforts will not be disturbed by future development because the developer exerts more control over the land base, albeit not full control owing to overlapping oil and gas tenures issued in stratigraphic layers and forestry management areas. On the other hand, on-lease restoration is likely to have limited benefits for caribou during the project's operational phase because the restored areas will presumably be located in proximity to existing and/or future disturbances. Off-lease restoration provides the opportunity to target areas of core habitat to maximize the benefit to caribou. The Alberta *Public Lands Act* does not include mechanisms for permanent protection of such restoration efforts; therefore, they are at risk of being destroyed or compromised by other land-users. This deficiency must be addressed to ensure that offsets are in place for the duration of project impacts, and preferably for much longer (Gibbons & Lindenmayer, 2007; McKenney & Kiesecker, 2010; Bull *et al.,* 2013a). Protection of restored sites could be achieved by creating a new disposition type for offsets under the *Public Lands Act* and would require the development of a regulatory review process to approve offset site selection.

To be most effective, caribou habitat restoration activities should consider future development footprint and industrial access requirements, relative quality of adjacent caribou habitat, recreational use by the public, Aboriginal use, caribou occurrence, and provincial habitat restoration priorities for caribou. Weighing competing land-use demands is a challenging process and may require complex and lengthy consultation, but ignoring it may result in failure. For example, recreational all-terrain vehicles use can significantly hinder revegetation efforts implemented at high cost (Coupal & Bentham, 2014).

Resolving issues pertaining to habitat based caribou offsets will be challenging, and even if achieved, probably cannot be implemented without simultaneously addressing predation because habitat values for caribou herds in Alberta are already considered below those required to achieve caribou conservation (Environment Canada, 2012a), because caribou habitat takes a long time to reclaim, and because caribou are in such steep decline that any substantial time-lag may result in conservation failure (Hervieux *et al.,* 2013).

Addressing predation in the short-medium term is necessary, but applying them as offsets is extremely challenging. Management actions such as predator and/or prey control through aerial gunning or poison directly address the proximate causes of caribou decline and are much more likely to facilitate caribou population persistence, but still present numerous challenges (Hervieux *et al.,* 2014). Wolf control and prey control generally generate negative public perceptions (NRC, 1997; Martínez-Espiñeira, 2006; Van Ballenberghe, 2006), and developers could contribute only through a program implemented by the province, which has the responsibility for directly managing wildlife. Fencing options, especially large-scale predator exclusion fences may work, but also will probably be viewed negatively (Pickard, 2007; Scofield *et al.,* 2011) and may have unintended consequences (Pople *et al.,* 2000; Norrdahl *et al.,* 2002; Long & Robley, 2004; Hayward & Kerley, 2009).

Changes in hunting regulation to encourage greater harvest rates of moose and deer may be more palatable and have been applied previously (Serrouya *et al.,* 2015), but this is also not an offset that can be actioned by a developer. Developers might contribute to these efforts by creating financial incentives for hunters and trappers to reach prescribed quotas for moose, deer and wolves. Implementation of such incentives would likely necessitate lengthy negotiations with government in order to reach an agreement, they may be viewed negatively as "bounties", and their effectiveness relative to aerial control or poisoning is questionable (Webb *et al.,* 2011).

A final challenge to offset implementation is achieving clarity about where offsets, either habitat-based or focused on addressing predation, might be appropriately undertaken. Standard like-for-like approaches indicate that offset sites should be as close as possible to impact sites to ensure that benefits are realized in the same area (McKenney & Kiesecker, 2010); however, the selection of offset locations that best balance proximity to the impact sites with effectively achieving conservation outcomes is often unclear (Kiesecker *et al.,* 2009). For caribou offsets, our interpretation of the federal recovery strategy's goal to maintain or recover all populations within caribou range in Canada (Environment Canada, 2012a) is that any negative impacts to caribou or caribou habitat should be offset within that same range. This added restriction poses an additional challenge by spatially limiting acceptable offset locations. The requirement to offset within a given caribou range precludes the application of the triage-based approach recommended by Schneider *et al.,* (2010). Given that there are limited resources to implement recovery efforts and that population viability varies among herds, Schneider *et al.,* (2010) argue that caribou conservation efforts should focus on probability of success and cost as opposed to risk of extirpation. Considering the Alberta context where resources have been over-allocated in some caribou ranges, we think it prudent that policy makers consider prioritizing caribou offsets where there is a greater probability of success. This approach would consider caribou offsets at the provincial scale with the trade-off of potentially losing herds in highly impacted areas while increasing the odds of successful conservation of other herds.

Conclusions and recommendations

We have discussed offset opportunities for caribou and identified practical challenges associated with them. The prevalence of implementation challenges is not surprising given that the science of offsets is still in its early development stage and government policy has not yet developed to accommodate it. A common challenge for all caribou offset opportunities in Alberta is the lack of framework and policy to guide consistent and appropriate application of offsets. Successful implementation of caribou offsets will depend in part on the development of comprehensive offset framework and policy to guide project proponents and environmental practitioners. Such framework and policy are urgently needed.

Regulations and policy need to emphasize that offsets are not a panacea. Offsets are the last resort in the mitigation hierarchy and are not a solution to failing to do a good job of avoidance, minimization and rehabilitation/ restoration. The mitigation hierarchy has not always been systematically applied in Alberta's environmental assessment process (Clare *et al.,* 2011; Clare & Krogman, 2013). Project developers and environmental practitioners could benefit from evaluation criteria to help determine when one can defensibly move down the mitigation hierarchy (e.g., move from avoidance to minimization). Clear guidance from government agencies would provide much needed consistency across projects and could prevent bureaucratic slippage (Clare & Krogman, 2013), that is the propensity for broad policies to be changed through successive reinterpretation, such that the ultimate implementation may bear little resemblance to the broad statements of policy intent (Freudenburg & Gramling, 1994).

Habitat restoration is likely the most promising caribou offset strategy for industry given the extensive opportunities for habitat restoration, its technical feasibility, and the fact that maintenance of critical habitat is mandated under the federal recovery strategy. Although habitat restoration has been criticized as an inappropriate offset tool in some cases (e.g., Curran *et al.,* 2014), we argue that this approach is applicable to achieve offsets specifically for caribou because habitats can be restored to a form that is less likely to support alternate prey. Under the current public lands tenure system, no mechanism is in place to secure restoration efforts on caribou ranges and this deficiency must be addressed before restoration can serve as a viable offset strategy.

Time-lags between restoration actions and the provision of measurable offsets means that restoration cannot be the sole solution to caribou recovery. Current levels of population decline dictate that restoration should be used in conjunction with immediate management actions (i.e., manipulations of the large mammal predator-prey system) addressing proximate causes of caribou declines to ensure their persistence over the short and medium-term. These kinds of programs are more difficult for industry to contribute to and may be less commonly used as a caribou offset. Where industry cannot contribute, such programs must remain the responsibility of government (Environment Canada, 2012a; Hervieux *et al.,* 2014).

Given the many challenges of implementing caribou offsets, we think there is much value in considering Schneider et al.'s (2010) triage perspective further. Using a provincial scale for caribou offset site selection would facilitate this approach. Although a triage approach where offsets focus on the least affected herds may mean accepting the loss of some of Alberta's caribou herds, it may also mean that some can be saved while simultaneously developing some of the province's most valuable resources (Schneider *et al.,* 2010). The current policy of exploration and development everywhere all the time and conservation everywhere all the time may result in both conservation failure and higher costs and increased uncertainty for developers.

We recommend that future research focus on evaluating the efficacy of proposed offset strategies for caribou. Further empirical evidence is required to reduce uncertainty and to help policy makers, regulators, project proponents, and environmental practitioners make informed decision on offset design and implementation. Specifically, a caribou offset framework would benefit greatly from understanding the time interval until restored habitat benefits caribou by adjusting predator-prey dynamics, or the scale at which restoration of historic disturbance must occur to achieve a measurable benefit to caribou populations given the scale at which large mammal predator prey systems operate. Similarly, changes in predation rates or caribou populations due to a project are rarely quantified in environmental assessments, making it difficult to estimate what kind of changes to the predator-prey system may be required to offset impacts of a project (e.g., using wolf control, large scale predator fencing, or maternity penning). A better understanding of how to apply multiple offset currencies, including both habitat and predation rates (*sensu* Bull *et al.,* 2013a), would be helpful for loss-gain calculations.

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Habitat restoration as a key conservation lever for woodland caribou: A review of restoration programs and key learnings from Alberta

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Abstract: The Recovery Strategy for the Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population in Canada (EC, 2012), identifies coordinated actions to reclaim woodland caribou habitat as a key step to meeting current and future caribou population objectives. Actions include restoring industrial landscape features such as roads, seismic lines, pipelines, cut-lines, and cleared areas in an effort to reduce landscape fragmentation and the changes in caribou population dynamics associated with changing predator-prey dynamics in highly fragmented landscapes. Reliance on habitat restoration as a recovery action within the federal recovery strategy is high, considering all Alberta populations have less than 65% undisturbed habitat, which is identified in the recovery strategy as a threshold providing a 60% chance that a local population will be self-sustaining. Alberta's Provincial Woodland Caribou Policy also identifies habitat restoration as a critical component of long-term caribou habitat management. We review and discuss the history of caribou habitat restoration programs in Alberta and present outcomes and highlights of a caribou habitat restoration workshop attended by over 80 representatives from oil and gas, forestry, provincial and federal regulators, academia and consulting who have worked on restoration programs. Restoration initiatives in Alberta began in 2001 and have generally focused on construction methods, revegetation treatments, access control programs, and limiting plant species favourable to alternate prey. Specific treatments include tree planting initiatives, coarse woody debris management along linear features, and efforts for multi-company and multi-stakeholder coordinated habitat restoration on caribou range. Lessons learned from these programs have been incorporated into large scale habitat restoration projects near Grande Prairie, Cold Lake, and Fort McMurray. A key outcome of our review is the opportunity to provide a unified approach for restoration program planning, best practices, key performance indicators, and monitoring considerations for future programs within Canada.

Key words: Alberta; federal recovery strategy; habitat restoration; woodland caribou.

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Introduction

In 2012, the federal Recovery Strategy for the Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population in Canada was publicly released and it described coordinated actions to reclaim woodland caribou habitat as a key step to meeting caribou population and distribution objectives (EC, 2012). Actions include restoring industrial landscape features such as roads, seismic lines, pipelines, cut-lines, and cleared areas in an effort to reduce landscape fragmentation and the changes in caribou population dynamics associated with changing predator-prey dynamics in fragmented landscapes. The importance of habitat restoration as a recovery action within the federal recovery strategy is high, considering all local Alberta populations have less than 65% undisturbed habitat, which is identified in the strategy as a threshold providing a 60% chance that a local population will be self-sustaining. All local Alberta populations are considered either "not self-sustaining" or as "likely not self-sustaining", with 10 of 14 populations with long-term empirical data known to be in significant decline (Hervieux et al, 2013). Alberta's Provincial Woodland Caribou Policy also identifies habitat restoration as a critical component of long-term caribou habitat management and population recovery (GOA, 2011).

There is on-going economic pressure in Alberta to disturb caribou habitat within "not self-sustaining" local populations, since caribou ranges overlap with oil and gas and bitumen reserves. As a result, the demand to build additional infrastructure to produce and support market delivery of those reserves is also increasing. The challenge is whether continual development of energy sector projects, such as seismic, road and pipeline development is possible within caribou ranges while reducing net residual effects to caribou and caribou habitat. To address this challenge, a number of large-scale and project specific habitat restoration initiatives have been implemented by multi-

Figure 1. Natural regeneration of a typical conventional seismic line in the boreal forest. Photo courtesy of Brian Coupal.

stakeholder groups and individual companies in recent years, including restoration projects near Grande Prairie, Cold Lake, and Fort McMurray. The objectives of these initiatives have been to restore habitat on historical anthropogenic footprint in an attempt to create intact habitat areas for caribou and/or to slow down predation rate as a result of the footprint. Given a lack of formal guidelines on habitat restoration objectives or techniques, as well as a lack of reporting on program learning's, Golder Associates Ltd. (Golder) organized a Restoration Workshop in Edmonton, Alberta, in June 2013. More than 80 participants from industry, government, academia, and consulting attended the one day workshop to discuss caribou restoration efforts in Northern Alberta. The intent of the workshop was to provide an opportunity to improve common understanding from on-the-ground restoration programs in terms of key performance indicators, successes, best practices and outcomes and to link the results of these programs back to provincial guidance on habitat restoration considerations. The workshop balanced learning and discussion with knowledge sharing presentations by government and industry. Breakout groups focused on a series of key questions regarding restoration efforts. Here we outline how lessons learned from past restoration initiatives educate the objectives and techniques for implementation of habitat restoration for current and future restoration projects.

Habitat restoration initiatives

A Caribou Range Restoration Project (CRRP) was first established within Alberta in 2001 (Szkorupa, 2002) in an effort to address growing concerns with the relationship between industrial development and declining local caribou populations. At that time, research from James (James, 1999) suggested wolves were gaining a predation advantage using linear features created by industry, and that indirect habitat loss for boreal caribou was occurring through the avoidance of habitat adjacent to human disturbance (Dyer, 1999; Neufeld, 2006; Oberg, 2001). In addition, seismic lines were reported to have very slow reforestation rates (Revel *et al.,* 1984; Osko and MacFarlane, 2000), with slow tree regeneration attributed to root damage from the original disturbance, compaction of the soil in tire ruts, insufficient light reaching the forest floor, introduction of competitive seed mixes (i.e., plant seed mixes), drainage of sites, and repeated disturbances (e.g., all terrain vehicles) on seismic lines (MacFarlane, 1999 and 2003; Sherrington, 2003). Rehabilitation of existing anthropogenic disturbances within caribou range was expected to reduce the degradation of functional habitat over the long-term, with caribou no longer exhibiting avoidance of the disturbance feature (e.g., Oberg, 2001). The CRRP piloted techniques with the objectives of promoting revegetation of these features, while discouraging access for predator, primary prey, and human use.

The CRRP was a multi-stakeholder group initiated and steered by the provincial government agency Alberta Sustainable Resource Development (ASRD), and the Boreal Caribou Committee (BCC) (Dzus, 2001). Although the CRRP was not extended beyond 2007, the project did incorporate silviculture methods based on knowledge of forestry treatments, focusing on access control treatments and enhancing the vegetation recovery rate of historical seismic lines, pipelines, and lease roads. Based on the outcome of treatments and learnings on linear restoration, the CRRP prepared a Guidance Document (CRRP, 2007a) which included recommended practices for implementing a habitat restoration program, from the planning through to the treatment stages. A monitoring protocol document for revegetation (unpublished) (CRRP, 2007b) was also prepared. Key learnings during the CRRP included recognition that restoring linear development features is not equivalent to replanting a typical mono-

Figure 2. Use of coarse-woody debris on a 4 m wide seismic line. Photo courtesy of Canadian Natural Resources Ltd. Primrose and Wolf Lake Project.

culture or mixed stand forestry cutblock. Linear development features vary with respect to the width and type of initial disturbance, compaction levels, soil types, moisture regimes, and light levels. In addition, restoration objectives often differ, including discouraging predator and human access, and the establishment of vegetation which is not preferred browse for moose or deer.

A number of initiatives and trials established since the CRRP have focused on establishing vegetation and access control treatments on linear development features located within caribou range. Restoration programs have been developed under requirements to meet project approval conditions (provincially through Alberta Environmental Protection and Enhancement Act approval conditions for in-situ projects and federal pipeline approvals through the National Energy Board) as well as voluntary programs. Habitat restoration programs have included implementing treatments to encourage native vegetation establishment such as creating microsites using an excavator, seedling planting (tree and shrub species, frozen seedlings) (e.g., Golder, 2005; DES, 2004; Enbridge, 2010; Golder, 2010; Golder, 2011; Golder, 2012a; OSLI, 2012a), spreading coarse woody debris (Vinge and Pyper, 2012; Pyper and Vinge, 2012) and tree-felling (Cody, 2013; OSLI, 2012a) (Figures 1 to 6).

Lessons learned from these programs have been incorporated into large scale habitat restoration projects focused within caribou areas near Grande Prairie (CRRP, 2007c), Cold Lake (Golder, 2010; Golder, 2012a; Golder, 2015a;

Cody, 2013; Golder, 2015b), and Fort McMurray (COSIA, 2014; OSLI, 2012a), Alberta.

Existing knowledge

Conventional seismic lines, which are generally 6 to 8 m wide, have been reported to have very slow reforestation rates (Revel *et al.,* 1984; Osko and MacFarlane, 2000; Lee and Boutin, 2006). Tree regeneration along seismic lines is influenced by the characteristics of the adjacent forests (e.g., site productivity, tree and shrub species and heights) (Bayne *et al.,* 2011), method of clearing from the original disturbance, compaction of the soil from human use, insufficient light reaching the forest floor, maintenance of apical dominance from surrounding stands, introduction of competitive species such as graminoid dominated seed mixes, naturally poor drainage of sites and repeated disturbances (e.g., all-terrain vehicles, animal browsing, repeated exploration) (Revel *et al.,* 1984; MacFarlane, 1999; 2003; Sherrington, 2003; Lee and Boutin, 2006). The slow pace of recovery of plant communities on seismic lines has been recommended as an area where direct management activities, including access control to reduce repeated disturbance, and silviculture preparations to address site deficiencies, should be applied to set a line on a natural successional trajectory (MacFarlane, 2003).

Positive results for establishing native vegetation on seismic lines and pipeline rights-of-way (ROWs) have been recorded using techniques such as planting tree and shrub seedlings, and creating microsites by methods such as mounding that are conducive to seedling growth and natural vegetation encroachment (DES, 2004; CRRP, 2007b; Golder, 2010; 2011; 2012a; OSLI, 2012a; Macadam and Bedford, 1998; MacIsaac *et al.,* 2004; Roy *et al.,* 1999). Measures such as the use of coarse-woody debris (slash rollback) can address site condition issues including competition from non-target or undesired plant species, erosion, frost, and heat or moisture deficiencies, as well as to create microsites for germination (CRRP, 2007b; Pyper and Vinge, 2012; Vinge and Pyper, 2012).

Transplanting native vegetation has been at-

Figure 3. Alder shrub seedling planting on a pipeline ROW after 1 growing season. Photo courtesy of Enbridge Pipelines (Athabasca).

tempted along seismic lines and pipelines but is challenging to implement on a large scale due to the inconsistent availability of vegetation suitable for transplant, the potential for degradation of neighboring vegetation communities if transplants are sourced from adjacent stands, approval requirements to move vegetation, and less than ideal storage conditions for plant materials due to weather. Other treatments such as seeding and seedling planting have been shown to be more successful and predictable in comparison (Golder, 2012b).

Both natural revegetation and seedling planting initiatives on both seismic lines and pipelines have benefited from minimal disturbance construction during frozen ground conditions that reduce or avoid grubbing and grading and minimize disturbance to the duff layer (e.g., DES, 2004; TERA, 2011; 2012; Enbridge, 2010; TCPL, 2014).

The ability of linear developments to regenerate to native species is affected considerably by human use. Oberg (2001) identified that recovery of conventional seismic lines within the foothills to functioning caribou habitat occurs within 20 years following disturbance in west-central Alberta. Within a boreal caribou area, seismic lines that were allowed to regenerate naturally achieved an average height of 2 m, across all boreal vegetation types, within 20 to 25 years, if the line had not undergone a repeated disturbance (e.g., re-cleared to ground level for winter access or exploration use). The average age of trees on the revegetated seismic lines was only 10 years, suggesting sites that are continually disturbed or re-cleared by seismic exploration or vehicular access take longer to regenerate. Restoration efforts are also negated when human use destroys or damages seedlings after planting (Enbridge, 2010; Golder, 2011; 2012a).

Subjective expert ratings suggest that effectiveness of access control measures such as gates, berms, mounding, slash rollback, and visual screening vary considerably between negligible and high effectiveness in controlling human access within caribou ranges (CLMA and FPAC, 2007). Effectiveness of access control measures are dependent on suitable placement (e.g., placed to prevent detouring around access control point), enforcement, and public education of the intent of the access control, which facilitates respect of the control measures (AXYS, 1995). Excavator mounding is a well-researched and popular site preparation technique in the silviculture industry (Macadam and Bedford, 1998; Roy *et al.,* 1999; MacIsaac *et al.,* 2004). Mounding has been found to discourage human access such as off road vehicular use and also creates microsites that improve vegetation establishment (CLMA and FPAC, 2007). Physical access control measures provide short-term solutions to manage access and allow for natural regeneration (Golder, 2009). It has been suggested that once linear features have regenerated to a pole sapling or young forest structural stage, they no longer facilitate vehicular access (Sherrington, 2003).

A number of the techniques used to block human access use of regenerating industrially disturbed features also contribute to initiatives to block line–of–sight. Short-term management for access and line-of-sight blocking is understood to lead to long-term access control by providing the necessary conditions for the disturbance to regenerate to natural vegetation conditions (CLMA and FPAC, 2007). Expediting growth of visual barriers along linear features can be achieved by concentrating reclamation efforts on productive upland habitats, since tree and shrub (e.g., alder which is less palatable for prey species) species grow more quickly on these sites compared to lowland sites. On deciduous and mixedwood upland sites, encouraging deciduous tree species and shrub growth is important to quickly establish visual and physical barriers in the short-term. Tree-felling has recently been applied through the Cenovus Energy Linear Deactivation (LiDEA) project in northeastern Alberta and early results suggest it is effective in providing an immediate access control through remote camera monitoring (Cenovus, 2014). Although regeneration of conifer species is the endpoint for caribou habitat use and minimizes habitat creation for other prey species, conifer species growth rates are slower than the growth rates of deciduous species. Faster growth rates provide for access control and line-of-sight barriers more quickly (DES, 2004). Recent field trials suggest that planting shrubs along with conifer tree species may allow trees to grow healthier, faster and with less competition for nutrients and water from fast-growing grasses than when planted without shrubs (OSLI, 2012a). Planting shrubs may also provide important habitat benefits for wildlife, compared to only planting tree seedlings, by providing hiding cover (Bayne *et al.,* 2011).

The OSLI program (now COSIA) includes on-going studies to determine what the most efficient vegetation introduction techniques are for peatland areas, such as planting frozen seedlings in the winter instead of summer planting, and whether to use seed or seedlings, dependent on site conditions and other variables. The OSLI/COSIA program also involves voluntary restoration of legacy footprint within caribou critical habitat in an effort to restore large, late seral stage patches of caribou habitat to increase habitat intactness and discourage corridor use (OSLI, 2012a).

The Government of Alberta has not provided a manual for reclamation that can be utilized for developing silvicultural prescriptions for large scale habitat restoration programs. However, a revegetation matrix was developed by Alberta Environment and Parks and published within the Cumulative Effects Management Association (CEMA) document 'Stony Mountain 800 Linear Footprint Management Plan' (CEMA, 2012). The revegetation matrix examined vegetation trajectories associated with the natural recovery of linear features over time. The values provided in CEMA (2012) are based on practitioner opinion as well as estimates based on ecosite and tree species growth potential. The revegetation matrix can be used to simulate how vegetation height may change over time (CEMA, 2012).

While there has been some effort to assess wildlife use of regenerating seismic lines (e.g., Bayne *et al.,* 2011) and reclaimed areas (e.g., Hawkes, 2011), few researchers have documented the relationship between natural habitat recovery and wildlife responses to recovery with respect to caribou. A pilot study to measure the effects of revegetating linear disturbances on wildlife use and mobility collected data for a group of predators (i.e., cougar, wolf, coyote, lynx, grizzly and black bears) and prey (i.e., moose, deer and caribou) (Golder, 2009). Results indicated that revegetated seismic lines with a minimum of 1.5 m of consistent vegetation regrowth were preferred by both predator and prey species (including caribou) compared to open, low (< 1.5 m vegetation) vegetation control lines. The line-of-sight measured on the revegetating lines was typically less than 50 m. In general, control lines were used primarily for travel by both predators and prey species. Human use was primarily limited to the control lines. Golder (Golder, 2009) suggested that moose and deer may have been attracted to the revegetated lines for forage availability and perceived cover protection. The preference for regenerating seismic lines by wolves may be explained as a response to increased prey use of these lines. More recently, pre-treatment (Dickie, 2015) and post-treatment wolf movement data is being gathered through the University of Alberta to look at the effectiveness of line-blocking within the Cold Lake region of Alberta. Wolves selected conventional seismic, pipelines, railway, roads, trails, and transmission lines, but did not select low-impact seis-

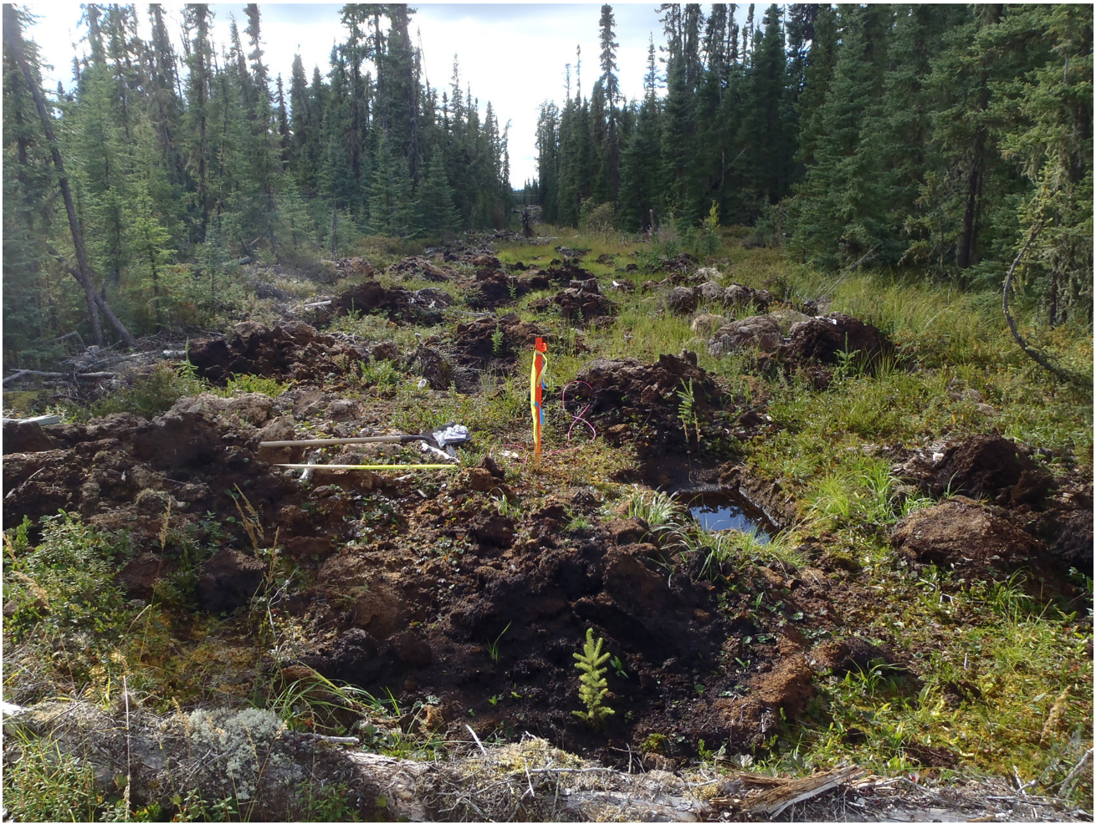

Figure 4. Mounding and seedling transplanting treatment location. Photo courtesy of Canadian Natural Resources Ltd. Primrose and Wolf Lake Project.

mic in summer (Dickie, 2015). Wolves selected all linear disturbance footprints in winter with the exception of trails (Dickie, 2015). Wolves moved faster on linear disturbance footprints as compared to surrounding forest, with the exception of low-impact seismic in both summer (30% reduction in travel speed) and winter (53% slower on low-impact seismic lines than in surrounding forest) (Dickie, 2015). While using linear features, wolves selected for shorter vegetation, changing their movement on linear features with increasing vegetation height, with a breakpoint of 1m in summer and 2.7m in winter. When travelling on linear features, wolf travelling speed decreased by 20% after linear features reached a height of 1m in summer, and travelling speed decreased by 26% after lines reached 2.7m in winter (Dickie, 2015).

The focus of habitat restoration initiatives has been on revegetation and access control, and limiting plant species that are favourable to wolves' primary prey, with the goals of creating line-of-sight breaks, directly restoring habitat with transplanted vegetation, planting shrub and conifer tree seedlings, sowing native shrub and tree seed, and controlling human access to reclaimed areas to allow undisturbed vegetation growth. Vegetation recovery in the medium and long-term following the creation of linear disturbances has not been extensively documented, however, the attributes of naturally revegetated linear features have been documented by the CRRP (CRRP, 2007b), the Foothills Research Institute (FRI, 2014), and van Rensen *et al.,* (2015). Natural regeneration does occur, with linear development features in mesic sites, the most likely to regenerate naturally without treatement, whereas a linear development feature in a bog or fen is least likely to regenerate naturally; and a narrow (<3m) line has improved regeneration over a wider line (van Rensen *et al.,* 2015). Natural regeneration to 3 m vegetation height is inversely related to terrain wetness, line width, proximity to roads as a proxy for human use of lines, and lowland ecosites (fens, bogs) (van Rensen *et al.,* 2015). Areas adjacent to major rivers illustrate high probability of regeneration. Overall, terrain wetness and the presence of fens has the stongest negative effect on natural regeneration (van Rensen *et al.,* 2015). Lack of time sequence recording for regenerating seismic lines and other linear developments reduces the ability to estimate natural rates and types of vegetation recovery, however predictive models do exist (e.g., van Rensen *et al.,* 2015).

Workshop results

Although the federal Recovery Strategy (EC, 2012) for boreal caribou describes the requirement for habitat restoration, it is not clear what defines successful habitat restoration. During the workshop participants discussed a proposed definition of habitat restoration: Restored (decades) - disturbed caribou range is returned to functional habitat that can support self-sustaining caribou population without ongoing intervention (e.g., predator control). Participants identified that habitat restoration needs to consider spatial and temporal scales, trajectories, as well as predator/prey dynamics.

During the restoration workshop, a number of the presentations discussed key elements of program planning, including authorization to implement restoration measures. Government of Alberta representatives acknowledged that an approval process needs to be developed that provides a consistent approach to authorize implementation of restoration treatments on historical seismic lines, and that development of the process is under discussion. As well, Alberta Environment and Parks presented draft restoration priority areas mapping, available upon request, to help direct where restoration efforts should be focused (D. Hervieux pers. comm., 2013).

Learnings from existing restoration programs were presented and included an awareness that not all linear disturbances are equal and that restoration on linear disturbances differs from silviculture prescriptions applied to cutblocks, given the higher variability in site conditions. As a result, the toolbox for restoration treatments needs to consider a number of variables, in particular the lack of a seed bed and mineral layer for plant growth and compaction.

It was discussed, based on previous initiatives, that prior to applying treatments on the ground, linear feature (and polygon) inventories of the existing footprint are the first steps in designing a restoration program. Collecting inventories help ensure an efficient allocation of resources committed to habitat restoration. For example, a pilot habitat restoration program in west-central Alberta approximately four townships in size and another pilot northwest of Cold Lake, approximately eight townships in size, reported that approximately 85% of linear features observed were already on a natural recovery trajectory and revegetation treatments were not recommended (CRRP, 2007c; Golder, 2010) (Fig. 1). Inventories are gathered using remote sensing to spatially map linear disturbances and the level of natural regrowth (e.g., van Rensen *et al.,* 2015). In addition to the amount of natural regrowth, field truthing of candidate treatment sites is completed to document detailed ground conditions. Data is collected on classifying the type(s) of disturbance (roads are considered severe disturbance whereas a cutline is often minimal disturbance), level of human (e.g., all-terrain vehicle) and wildlife (game trails) use, width and orientation of a line (impacts light penetration and moisture level), compaction level (impacted from construction practices), soil mineral layer (nutrients) and microsite availability, adjacent ecosite phase / forest attributes (very wet to very dry, upland/transitional/lowland), coarse woody debris level/availability/fuel loading considerations from a fire management perspective, and historical seeding practices which often results in high levels of competing vegetation to conifer seedlings (Vinge, 2013; CRRP, 2007c).

During the remote sensing and ground truthing of site conditions, treatment sites and prescriptions are finalized and often located into priority areas for restoration and to areas where human access control treatments will prevent repeated use. This ensures that the 'right lines for restoration' are selected. For large scale restoration programs, future development plans in the area (e.g., forestry harvest plans, lease areas, development footprints, pay depth to bitumen, etc.) (ALT, 2009), provincial priority areas, as well as a focused plan to create large, contiguous intact habitat areas should be considered. Restoration program development considers not only a planning scale, but a tactical scale with efficiency of operational implementation considerations. For example, the OSLI/COSIA program used a modelling approach called Landscape Ecological Assessment and Planning (LEAP) to enhance efficiency in bringing landscape data sources together to assess and develop restoration scenarios, strategic to tactical implementation plans, and monitoring plans (OSLI, 2012a).

Restoration toolbox

The objectives of past and current habitat restoration programs for caribou have been to restore habitat on existing anthropogenic footprint to create large contiguous habitat patches that can support self-sustaining caribou populations with historical predator-prey encounter rates. This objective implies that habitat restoration must address revegetation, predator and primary prey access, predator efficiency, and forage for primary prey species. Although the federal recovery strategy and analyses to set caribou recovery management measures indicate that habitat restoration is linked to improving caribou population projections, the feasibility (cost, large scale application, rate of restoration as compared to rate of ongoing development pressure) and predicted outcomes of restoration activities remain highly uncertain (ALT, 2009; Wilson *et al.,* 2010). This uncertainty includes the time lag required to recover disturbed areas to effective habitat to support self-sustaining caribou populations.

Based on monitoring of revegetation of existing disturbances, it is expected that vegetation recovery of disturbed areas will take decades, with or without intervention. To address the time lag associated with natural revegetation of linear features (Fig. 1), industries and governments have built a toolbox of habitat restoration treatment best practices, focused on establishing vegetation similar to adjacent forest communities, creating line-of-sight breaks, and discouraging human, predator and primary prey usage of linear features. The treatment best practices, including their objectives and recommended specifications, are summarized in Table 1. Inclusion of a reference in Table 1 was based on if the results of implemented treatments were successful, for the objective outlined (e.g., if a treatment met the objective of establishing vegetation along a segment of linear feature where vegetation did not exist prior to the treatment). Specifications and considerations for each treatment are also provided, based on positive evidence of success.

The treatments designed to promote revegetation of linear features are intended to address micro-site deficiencies, and are well recognized silvicultural practices modified for linear feature application. When implemented properly, these practices will meet their objective of establishing vegetation. Additional monitoring on site preparation treatments such as mounding and spreading woody debris are currently being researched in NE Alberta to determine their efficacy in achieving the goal of discouraging predator and primary prey usage of linear features. Although long-term results are not yet available, preliminary results indicate these methods are achieving this objective (Cenovus, 2014).

Implementing practices to reduce a new project's impacts at the construction phase will reduce the need for, and the amount of, habitat restoration required following construction. Construction practices which enhance the ability of a site to restore naturally will reduce the level of effort and cost of site preparation (e.g., mounding) and tree/shrub planting over the entire project. For example, three practices that can be implemented during or immediately following the construction phase of a project are minimizing line width (e.g., low impact seismic <3m width; Dickie, 2015), minimal disturbance vegetation removal (e.g., DES, 2004), and controlling off road vehicle access (Revel *et al.,* 1984).

Since the ability of cleared areas to quickly regenerate to native species following construction is affected considerably by human use, applying human access control measures, with effectiveness monitoring, along linear features should occur immediately following construction. Woody debris treatments, excavator mounding, berms, tree-felling and steel gates are treatment types that are effective immediately and can be considered; but require monitoring. The type of control can be determined by the amount of expected human use at the location, width of the linear feature, ecosite phase, and topography. For example, a seismic line seldom used by humans, crossing a newly constructed pipeline ROW, may be treated with excavator mounding and planted seedlings (e.g, Fig. 4), while a pipeline crossing of a winter access road, well-used by humans, may be treated with a greater density of excavator mounding, planted seedlings, along with a steel gate.

Reclamation criteria and guidelines for forested areas should be consulted prior to determining specifications and design of a tree and shrub seedling planting program. For example, the Government of Alberta guidelines for forest reclamation in the oil sands region (AENV, 2010) specify ranges of seedling planting densities that vary by the site type and species planted. These guidelines are not specific to caribou habitat restoration, and may need to be modified with consideration to measureable objectives for caribou habitat restoration. The Science and Community Environmental Knowledge branch of the Government of British Columbia has recently commissioned the creation of a Boreal Caribou Habitat Restoration Operational Toolkit for British Columbia that contains reclamation recommendations specific to caribou ranges, focusing on linear feature restoration (Golder, 2015a). Considerations for determining species, planting density and locations of planting should include site type (dry, moist/poor, moist/rich, wet rich), surrounding vegetation community, disturbance level (high with no LFH layer, low with LFH layer intact), coarse woody debris level, and site preparation (Vinge, 2013).

A critical component of a successful habitat restoration program is protection of the treatment locations from disturbance. Sites that have been developed using methods that promote speedy natural revegetation or planted to enhance revegetation, line-of-sight break locations, or access control treatments should be clearly marked in the field and protected with physical barriers if necessary. For example, seedlings planted on an upland graded site can be damaged or destroyed from human use of the ROW unless they are protected by a sufficient layer of coarse woody material.

Monitoring

Monitoring of construction practices, the success of treatments to establish vegetation, lines undergoing natural revegetation trajectories and the effectiveness of access control methods

Figure 5. Lease road prior to treatment with mounding, tree-felling, tree-bending, and tree transplanting. Photo courtesy of MEG Energy.

Figure 6. Lease road after treatment with mounding, tree-felling, tree-bending, and tree transplanting. Photo courtesy of MEG Energy.

is necessary for any habitat restoration program. Monitoring programs should be linked to restoration objectives and measureable targets for the program to determine success or opportunities for adaptive management measures within restoration priority areas. During the workshop participants discussed monitoring programs and the overall consensus was that there is a need for consistent design in what's being measured, that there should be near term variables measured to determine if a site is on trajectory (with consideration of revised reclamation certificate criteria); successional trajectories or milestones should be determined and monitored against; and there is a disconnect between the end goal of caribou population lambda and the desire to consider habitat restored as early as possible. Adaptive management on restoration programs will need to be implemented by adjusting and/or supplementing restoration measures, where warranted, to achieve the objectives of the habitat restoration initiatives. Monitoring programs will need to consider a number of response metrics including the wildlife response to restoration (multi-species including caribou population trends, wolf movement and behavior, and primary prey population response) and the site level response both short-term and long-term with successional trajectories or milestones developed (Cody, 2013). Given the relatively short time period since large scale habitat restoration programs have begun to be implemented, field results are currently in the early stages of reporting regarding the success of caribou habitat restoration methods meeting their objectives. Monitoring outcomes will inform adaptive management, allowing for modification of unsuccessful measures to continuously improve, and are an important means of addressing uncertainty.

Discussion

At the national scale, Alberta's woodland caribou are among the least viable in Canada (EC, 2011). Under the *Species At Risk Act*, in 2012 the federal government released its recovery strategy for woodland caribou, with a clearly outlined habitat threshold to meet critical habitat levels (EC, 2012). In four caribou ranges in northeastern Alberta underlain by oil sands deposits, on average only 24% of caribou habitat remains undisturbed, far below the recovery plan target of 65% undisturbed habitat (Pembina Institute, 2012). For any new project planned or project expansion within a caribou range in northeast Alberta, under the SARA, the new project footprint could be deemed destruction of critical habitat for the species. As such, planning for approved future development projects within caribou ranges will need to consider the entire mitigation hierarchy: avoidance of caribou range; minimizing impacts through project planning, utilizing the least footprint necessary, overlapping land uses (e.g., coordinated access planning, integrated land use management planning); planning out a comprehensive habitat restoration plan; and include off-sets to address residual project effects due to the time lag and uncertainty around habitat restoration success to caribou recovery.

Habitat restoration has been highlighted within the federal recovery strategy, as well as within the Alberta Caribou Policy (GOA, 2011) as a critical component of long-term caribou habitat management. Given the current range condition for caribou in Canada, recent National Energy Board and Federal Joint Review Panel conditions for pipeline ROW occurring within caribou ranges have included preparing, implementing and monitoring Caribou Habitat Restoration Plans (e.g., NGTL, 2014a; 2014b). These Caribou Habitat Restoration Plans provide details on the objectives of restoration plans, the criteria used to identify potential habitat restoration sites, the process to identify restoration actions to be used at different types of sites, quantifiable targets and performance measures that will be used to evaluate the effectiveness of restoration measures to offset impacts to habitat, as well as a follow-up monitoring program (NEB, 2013). Long-term vegetation removal and the time-lag associated with vegetation re-establishment to suitable caribou habitat are considered residual effects and are to be addressed with habitat off-set measures for caribou (NEB, 2013).

Although habitat restoration activities have moved from pilot projects beginning in 2001 to large scale project implementation since the release of the recovery strategy, some cautionary details need to be considered. First, there is currently no direct link to indicate that implemented restoration treatments are having a positive effect on caribou populations. Although modelling scenarios of management options for caribou indicate that restoration of habitat should have benefits in the long-term by contributing to the restoration of large contiguous habitat patches that are preferred by caribou (e.g., ALT, 2009), additional management measures must be applied by governments to address the proximate cause of caribou declines. Specifically, governments must look to implement immediate population management of predators with effective habitat conservation measures (Hervieux *et al.,* 2014) and primary prey (CAPP, 2012). It has been noted that industry actions and planning around minimizing and eliminating project footprints will be of no value if caribou populations are not stabilized through aggressive wildlife (i.e., predator and alternate prey) management and long-term habitat conservation. It is recognized that the full benefits of habitat recovery will not be realized for decades because there is a 30 to 50 year lag time following reclamation before re-establishing vegetation becomes old enough to be considered low quality for other prey, and suitably old to be used by caribou (ALT, 2009). At a minimum, predator management through wildlife control will need to be continued for this entire lag period (ALT, 2009). Intuitively, extirpation risk of local herds will be reduced if habitat restoration begins as soon as possible (CAPP, 2012). Lastly, there is not a clear understanding of the desired objectives provided by regulators regarding landscape level habitat restoration programs. With no official framework, legislation or best practices within the provincial jurisdiction, it is difficult to implement consistent caribou habitat restoration and monitoring programs (Golder, 2013).

The driver to implement large-scale habitat restoration programs has been to lower the anthropogenic footprint within caribou ranges, and to address how caribou, wolves and primary prey species utilize habitat within restored areas. Although we have identified the planning and physical measures that can be implemented for a restoration program to begin restoring caribou habitat following construction or along historical linear features, it is unreasonable to directly associate local caribou population trends with these programs due to the time lag to grow vegetation; as well as the other factors contributing to these population trends, specifically the effects of apparent competitioninduced mortality on secondary prey such as boreal caribou (DeCesare *et al.,* 2010; Hervieux *et al.,* 2014), and the current rate of development. Monitoring and adaptive management of the restoration toolkit measures, and the wildlife response to these measures, will be a critical element of industry led habitat restoration programs.

Acknowledgments

This paper includes a summary of restoration programs and studies which have been conducted in Alberta since 2001. We would like to acknowledge the efforts of those collaborators involved in the CRRP for initiating habitat restoration treatments on linear disturbances as well as to those industry collaborators and government partners who have been involved in larger scale projects in northern Alberta. Tim Vinge, Alberta Environment and Sustainable Resource Development, provided guidance on the restoration toolbox available to industry. The *Historic and Current Habitat Restoration Initiatives Summary*, was originally prepared for, and is shared by, the Ministry of Forests, Lands and Natural Resource Operations, Fish Wildlife and Habitat Management Branch, of the Government of British Columbia.

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Table 1. Caribou habitat restoration treatment best practices.

Treatment	Objective(s)	Specifications	Positive Experiences with this Technique	Considerations to take into account	Ideal Timing for Treatment	References
Mechanical site preparation: Mounding and/or ripping using an excavator (Figures 4 and 6)	• Create microsites in areas where it is deemed to be effective for increasing survival and growth rates of planted seed and seedlings, and natural regrowth of woody species • Access control	• For access control purposes, mounds should be created using an excavator. The holes left behind by the mounds should generally be approximately 0.75 m deep, if feasible. The excavated material is positioned right beside the hole, creating the mounds. • Ripping should focus on upland sites where excessive moisture is not a concern. • Troughs created by ripping should be positioned to reduce erosion potential. • Target density of mounding for access control and/or microsite creation purposes can vary from 600 to 2,000 mounds/hectare (ha), depending on the size of the hole and mound. • When completing in synergy with seedling planting, seedlings are generally planted near the hinge of the mound: • Slightly higher up from the hinge for lowland and transitional sites • At or slightly lower than the hinge for upland sites	• For the purposes of creating microsites for planted seedlings, mounding is a well-researched site preparation technique in the silviculture industry. It is commonly used in wetter, low-lying areas to create elevated, welldrained microsites for seedlings • Mounding treed fen and bog areas can enhance a site to promote natural revegetation over time, as higher, drier spots are created that seed can eventually settle into and germinate • Mounding has been used as an access control measure on decommissioned roads, seismic lines, and pipelines to discourage off-road vehicle activity. It is effective immediately following implementation • Ripping is a standard forestry site preparation method that has been modified in this case for tighter workspaces	• Sufficient frost is required to access sites in the winter when crossing lowland areas: This varies from winter to winter • Research regarding machines that can operate in lowlands during non-frozen conditions is underway in NE Alberta	• Winter (frozen-ground conditions)	Macadam & Bedford, 1998 Roy *et al.*, 1999 MacIsaac *et al.*, 2004 Golder, 2010 OSLI 2012a, 2012b Nexen, 2013 CRRP, 2007b Archuleta & Baxter, 2008 USDA, 2009 BC MFR, 2014 BCFS, 1998 BC MOF, 2000 BC MFR, 1998

Table 1 continued.

Treatment	Objective(s)	Specifications	Positive Experiences with this Technique	Considerations to take into account	Ideal Timing for Treatment	References
Tree/shrub seedling planting (Figures 3 and 4)	• access control • erosion control • reduce line-of-sight • restore habitat	• Tree/shrub species are determined based on site conditions, the adjacent forest stand and restoration objectives (e.g., low palatability for ungulates). Coniferous tree species (Spruce sp., Pine sp.) are recommended to meet caribou habitat needs. Considerations for the use of shrubs: • Alder is generally planted because it forms an effective access control and line of sight break in a relatively quick period of time • Alder has a similar palatability rating for ungulates as conifer species (CRRP 2007b) • Willow is avoided due to the high palatability rating for ungulates (CRRP 2007b) • Shrub and tree seedlings are often planted together, depending on site conditions and anticipated natural revegetation of both species	• Seedling planting is considered a long-term restoration treatment due to the length of time it takes to establish effective hiding cover and access deterrents • Seedlings should ideally be sourced at least six months prior to planned planting dates • Seedlings and/or seed for growing seedlings may not be available for every species prescribed and therefore seed may need to be collected and grown in the nursery • Seedling planting during winter is generally restricted to lowland and transitional sites with organic soil that have been treated with mechanical site preparation immediately prior to planting • Seedling planting density for reclamation purposes has generally been based on adjacent site type and quickly providing hiding cover; it can range from 2,000 to 2,500 stems/hectare	• Use of frozen seedlings need to consider preparation of nursery stock, storage, planting temperature, and use of snow packing following planting to avoid winter freeze/thaw seedling mortality	• Seedlings can be planted on frozen sites in the winter (OSLI, 2012a; MEG, 2014; Cenovus, 2013) • Non-frozen stock are generally planted as summer stock in consideration of the Least Risk Timing Windows (BC) or Restricted Activity Period (AB) for caribou	AENV, 2010, 2011 BC MFR, 1998 Cenovus, 2013 CRRP, 2007b DES, 2004 Golder 2005, 2010, 2011, 2012b, 2012c MEG, 2014 OSLI 2012a, 2012b Nexen, 2013 NEIPC, 2010

Table 1 continued.

Treatment	Objective(s)	Specifications	Positive Experiences with this Technique	Considerations to take into account	Ideal Timing for Treatment	References
Spreading of woody material (Figure 2)	• control of human access during snow free periods • erosion control • protect planted seedlings from extreme weather, wildlife trampling, and damage from ATVs • provide site nutrients when the wood decomposes • provide microsites for natural seed ingress	• Spread woody material evenly across the entire corridor or polygon feature • Ensure woody material is consistently dense enough on the ground to discourage ATV and wildlife use • The Guide to Fuel Hazard Assessment and Abatement in British Columbia (2012) recommends woody loads do not exceed 99 tonnes/ha (~175 m^3/ha). An exemption may be allowed for larger volumes from the local fire centre under Section 25 or 26 of the Wildfire Regulation. • Vinge and Pyper recommend applying between 60 to 100 m^3/ha of woody material to reclaimed sites to mimic the natural range of variability for woody material in the forest • Implement at sites left for natural recovery when woody material is available as well as sites that are planted with seedlings	• The length of a treated segment is dependent on sufficient quantities of woody material available. Longer segments are a more effective treatment at controlling human access since ATV riders will be less inclined to attempt to travel through the woody material or traverse around it in adjacent forest stands if the woody material continues for an extended distance • Woody material can also conserve soil moisture, moderate soil temperatures, provide nutrients after it decomposes, prevent soil erosion, provide a source of seed for natural revegetation, provide microsites for seed germination and protection for introduced tree seedlings, and protect seedlings from wildlife trampling and browsing • Spreading of woody material is effective as an access control immediately following implementation • Woody material can be brought to a site from another location that has identical tree species	• Potential for fuel loading is a concern. The BC MFLNRO specifies acceptable levels of woody material while considering fire management objectives. Consultation with the local fire regulations recommended prior to treatment • Storage and use of woody materials may be compromised if bark beetle is a concern in the area and would be discussed with the local forest officer • Storage of woody material for extended periods without increasing fire hazard can be challenging and should be discussed with district fire managers as part of the planning process when using woody materials	• Winter (frozen-ground conditions)	CRRP, 2007b Enbridge, 2010 Osko & Glasgow, 2010 Golder, 2010, 2011 Government of Alberta, 2013 OSLI 2012a, 2012b BC MFLNRO, 2012 Pyper & Vinge, 2012 Vinge & Pyper, 2012

Table 1 continued.

Treatment	Objective(s)	Specifications	Positive Experiences with this Technique	Considerations to take into account	Ideal Timing for Treatment	References
Tree-felling/ Tree Bending (Figures 5 and 6)	• access control • reduce line-of-sight • reduce shade effect	• Bend (hinge) mature trees partially across the line with an excavator while treating the features for mounding purposes or spreading woody material • Fell mature trees across the line on upland and transitional sites (e.g., white spruce, pine, aspen, and black spruce) • An excavator is preferred for felling trees by pushing them over, if site conditions are suitable for excavator access • Trees can be felled with a chain saw if site access is suitable to address safety concerns • Trees are to be felled perpendicular to the line. Trees are not to be felled parallel to the line to reduce a fire hazard • Treatment locations to occur approximately every 20 m on lowland and upland sites • At each treatment location, 2 or more trees to be felled, from opposite sides of the line, to create an access control and line of sight break • Treatment locations should occur where sufficient sized timber is present. • Treatment locations should be as frequent as possible to discourage wildlife use, understanding that locations will be variable depending on forest stand adjacent to line • More trees to be felled near access points and intersections to restrict access and predator movement. Additional trees can be felled along identified lines where the adjacent trees are of suitable height (depends on width of line, need to cover across entire corridor)	• Tree-felling and tree bending across the line is mimicking natural processes that occur in the forest. • Tree-felling from the adjacent eco-site can reduce the shade effect on the corridor, leading to more sunlight and warmer soils, creating an enhanced environment for plant growth	• Tree-felling will result in tree mortality. Tree bending may keep trees alive with longer term needle cover • Potential for fuel loading is a concern. The BC MFLNRO specifies acceptable levels of woody material while considering fire management objectives. Consultation with the local fire centre is recommended prior to treatment. • Felling and bending is difficult to implement using hand fallers due to difficulties with access, and safety considerations. Mechanical equipment and site safety supervision should be considered • A permit may be required to fall trees that are outside the restoration site	• Winter (frozen-ground conditions	Cody, 2013 Cenovus, 2013 CRRP, 2007b MEG, 2014 Keim *et al.*, 2014

Table 1 continued.

Treatment	Objective(s)	Specifications	Positive Experiences with this Technique	Considerations to take into account	Ideal Timing for Treatment	References
Installing fences	• access control	• Fences can be installed at intersections with linear corridors and/or along a corridor where predator/human access control and line-of-sight breaks are required • Where natural topography or bends in the corridor do not break the line-of-sight, fences can be placed to limit access and sight lines • Wooden panels should be pre-constructed off-site, fastening the panels together at the treatment site to create a fence	• Fences could also be established using poles and geotextile or similar style decomposable matting • Gates can be installed on fences if desired to allow some operational access	• Fences are logistically challenging to establish in areas without pick-up access. • Used infrequently in the past and unknown efficacy • Installing fences during summer may be difficult to implement due to access availability	• Winter (frozen-ground conditions)	CRRP, 2007a