An Exploration of Conservation Breeding and Translocation Tools to Improve the Conservation Status of Boreal Caribou Populations in Western Canada

Pre-workshop Document

Calgary Zoo Centre for Conservation Research

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IUCN

SSC Species Survival Commission

RSG

SPECIES CONSERVATION PLANNING SUB-COMMITTEE

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1 CONSERVATION CONTEXT

Boreal populations of woodland caribou (hereafter, 'boreal caribou') are declining in most of their Canadian range, and are listed as "Threatened" under the Federal Species At Risk Act (SARA) and in numerous provinces and territories. In 2012, Environment Canada released its Boreal Caribou Recovery Strategy, a national framework to recover boreal caribou across its Canadian range. The Recovery Strategy aims to stop current population declines, increase individual populations to at least 100 animals, and improve habitat condition to at least 65 per cent undisturbed in each range. Provinces and Territories are charged with creating their own SARA-compliant Range Plans and Action Plans to meet the above recovery objectives on the habitat and population elements, respectively.

Unsustainable levels of predation are broadly agreed to be the proximate cause of caribou declines throughout their Canadian range. Landscape-level habitat changes resulting from human and natural disturbances in the boreal forest are thought to ultimately drive this unsustainable predation. Habitat conservation and restoration is a foundational element of caribou recovery, but it is likely that concomitant intensive management techniques will also be required to immediately address low calf survival and recruitment in boreal caribou populations. Successful recovery of boreal caribou populations will likely require the implementation of numerous habitat and population management tools concurrently. The proportion of these tools relative to one another will likely vary by individual caribou local populations (based on local population / range characteristics, demography, landscape condition, socio-political values, etc.).

2 THE WORKSHOP, JANUARY 26 – 28, 2016

This International Union for Conservation of Nature (IUCN) facilitated Workshop is designed to scope the utility of a broad range of population augmentation tools that may be implemented alongside other habitat-based tools. While habitat management is recognized as the foundational element of caribou recovery, workshop participants are reminded that habitat management tools will not be explicitly discussed at this workshop, except in a context where they are used in conjunction with more direct approaches to caribou population increase. Habitat management projects, programs and policies are being addressed and advanced through numerous other avenues, but are not the focus of this Workshop.

2.1 Workshop Goals and Objectives

The overall purpose of this Workshop is to explore the scientific background, conservation utility (including feasibility, practicality and impediments), and social acceptability of population management tools (i.e. conservation translocations, including maternal penning, predator exclosure fencing, wild-to-wild translocations and captive breeding and release) as tools to support the recovery of boreal caribou in Western Canada[1].

Key objectives for the Workshop are to:

- share management experience, scientific knowledge and social perspectives on caribou conservation breeding/translocation techniques, building on experiences from caribou, reindeer and other ungulate species worldwide;
- assess and evaluate a range of conservation breeding and translocation techniques (scientific research questions, management requirements, risks, population benefits, merits, and limitations) as they apply to boreal caribou in Western Canada;
- discuss potential criteria that could be used to assess and prioritize boreal caribou ranges as to their candidacy for conservation breeding and translocation tools.

Outcomes of this Workshop may be used to identify (for example) funding sources, partner agencies or organizations, delivery models, or proposals for feasibility studies and pilot projects that will allow for continued exploration and proving of these conservation tools, subject to government and regulatory approval in respective jurisdictions.

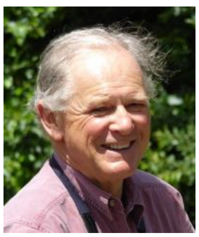
2.2 Workshop Facilitation

The involvement of key Chairpersons and members of IUCN SSC Specialist Groups in this workshop will draw on a body of global experience that has not yet been applied to boreal caribou conservation and recovery challenges in Alberta or anywhere in Canada. The IUCN is a global organization that supports scientific research, manages field projects all over the world, and brings governments, NGOs, the United

[1] Alberta (AB), British Columbia (BC), Northwest Territories (NWT), Saskatchewan (SK), Yukon (YT)

Nations and companies together to develop policy, laws and best practice. The IUCN, being the world's oldest environmental organization and the largest professional conservation network, is the leading authority on the environment and sustainable development. The Species Survival Commission (SSC) is one of six commissions within the IUCN and comprises a global network of over 8,000 volunteer experts spread between >120 specialist groups, task forces and working groups. The SSC advises on technical aspects of species conservation, as well as organizes action for threatened species.

Key IUCN representatives, all of whom have invaluable experience with conservation breeding, translocations and recovery planning for species at risk, are:



Dr. Mark Stanley Price (Workshop Lead Facilitator)
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Given the complexity of situations and the extent to which threat factors interact, decisions on best solutions will require at least qualitative structured decision-making. The workshop process will assume sound working knowledge of the pre-workshop document. Its headline conclusions will be discussed with focus on their generality to the range of situations faced by boreal caribou. This should lead to preliminary assessments of management approaches of most promise. Through a mix of plenary sessions and working groups, probably focusing on sets of caribou ranges, 'best chance' management interventions will be explored. By the end of the workshop, there should be a plan to explore the feasibility of management techniques that could be implemented at specific sites.

2.3 Pre-workshop Document

This pre-workshop document has been prepared to ensure participants have a common understanding of the pertinent topics prior to the Workshop so that the group may focus on moving forward with discussion of potential population augmentation tools during the Workshop. Specifically, this document presents relevant information on:

- IUCN guidelines for reintroduction and other conservation translocations;
- status of boreal caribou populations in western Canada;
- summary of caribou population augmentation tools previously (or presently) implemented in various jurisdictions;
- summary of key translocation (including fencing) programs for other ungulates worldwide to examine benefits and challenges associated with various techniques.

The information presented in this document is based on an extensive review of publicly-available material. In the interest of brevity, only key information is presented within the document; additional details are found in the accompanying appendices.

2.4 Post-workshop Report

Dr. Mark Stanley Price will lead the development, with support from Calgary Zoo researchers, of a post-workshop report (targeted availability the end of May 2016). The post-workshop report will corroborate areas and techniques that may produce positive conservation results based on workshop outcomes and a structured decision making process. This report may include:

- a determination of potential local populations where conservation translocations could contribute to population stabilization or recovery outcomes;
- key parameters or criteria to be addressed in area- and method-specific feasibility studies; and
- identification of individuals/organizations/funding sources that could/would collaborate in specific feasibility studies or pilot assessments.

3 IUCN GUIDELINES

The International Union for Conservation of Nature (IUCN) has drawn upon its extensive collective knowledge base and decades of experience to develop guidelines for translocations and ex situ management as tools for species conservation. These guidelines were strategically designed to be applicable to a range of different species and situations. As conservation translocations and captive-breeding are key tools of interest in this workshop, we rely heavily upon the knowledge and advice contained within these guidelines, as well as that of experts present at the workshop.

3.1 IUCN Guidelines for Reintroductions and Other Conservation Translocations

* All information contained within this section is summarized from IUCN/SSC 2013.

The IUCN's Reintroduction Specialist Group (RSG) and Invasive Species Specialist Group (ISSG) developed the 'Guidelines for Reintroductions and Other Conservation Translocations' (hereafter 'IUCN translocation guidelines') to prepare and advise conservationists on how to appropriately integrate translocations into species conservation strategies. **We recommend that any program intending to incorporate translocations should use or at least reference the IUCN translocation guidelines to responsibly and effectively plan and implement conservation translocations.**

The IUCN translocation guidelines define translocation as 'the human-mediated movement of living organisms from one area, with release in another'. **A 'conservation translocation' is an intentional translocation that aims to generate a measurable conservation benefit for a population, species or ecosystem.** Individuals to be translocated can be sourced from either wild (i.e. wild-to-wild translocation) or captive populations (i.e. captive breeding and release).

Types of conservation translocations are differentiated depending upon whether animals are released inside or outside of the species' indigenous range, and on the overall purpose of the translocation. **Translocation types most relevant for caribou conservation are 'population restoration' translocations, those that occur within a species' indigenous range, of which there are two types:**

(1) **reinforcement**: animals are released into an existing population to increase its viability, and

(2) **reintroduction**: a population no longer exists within the area and releases aim to re-establish a viable population.

In comparison, 'conservation introduction' translocations intentionally move and release a species outside of its indigenous range, and are conducted for two purposes:

(1) assisted colonization, to 'avoid extinction of populations of the focal species', and

(2) ecological replacement, for the species to 'perform a specific ecological function'.

The IUCN Guidelines recommend a number of steps that should be followed when considering, designing, implementing and following-up with any translocation (Figure 1).

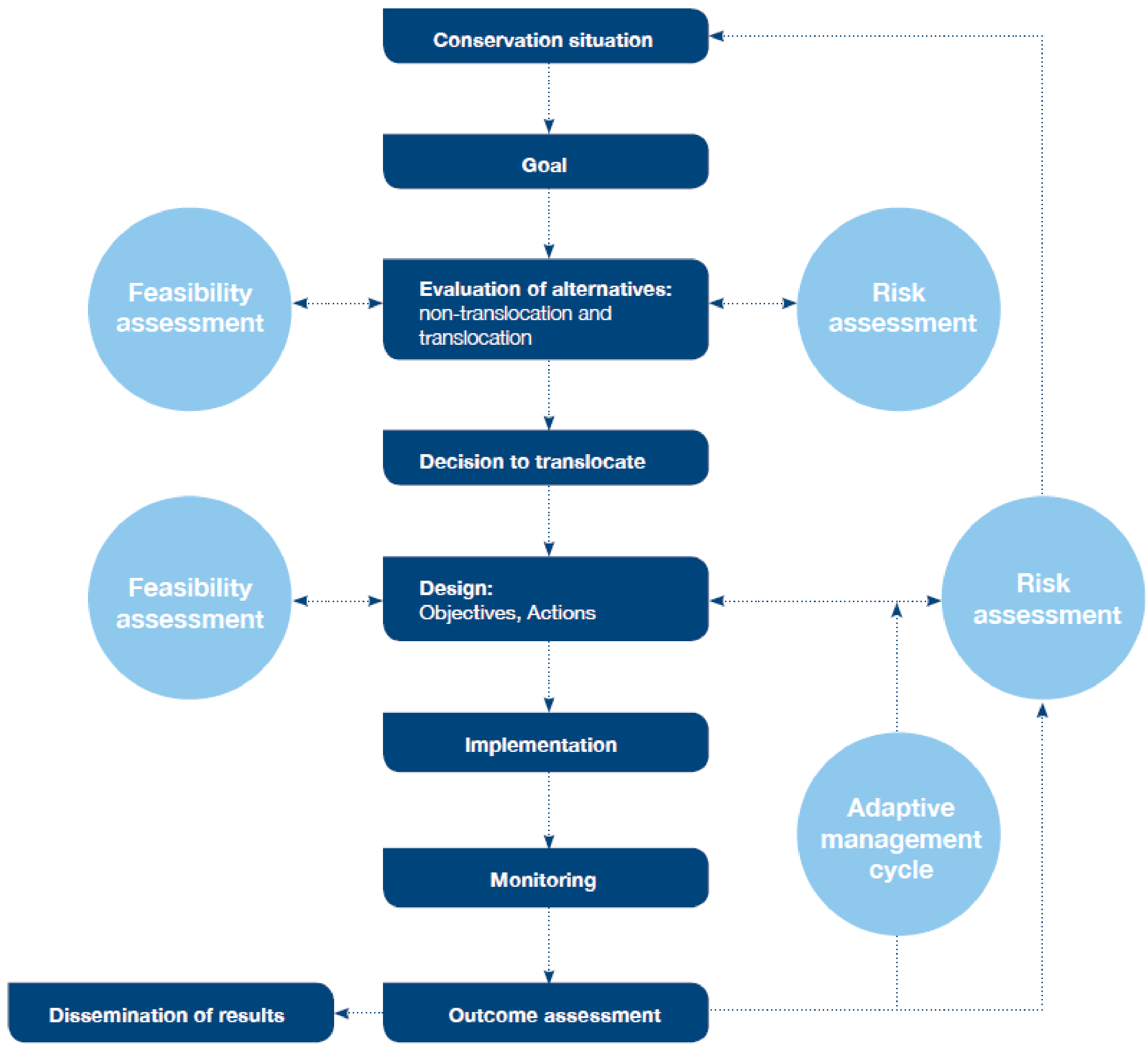

Figure 1: 'The conservation translocation cycle' (from IUCN 2013)

3.2 IUCN Guidelines on the Use of Ex Situ Management for Species Conservation

* All information contained within this section is summarized from IUCN/SSC 2014.

The IUCN Species Survival Commission (SSC) developed the 'IUCN Species Survival Commission Guidelines on the Use of Ex Situ Management for Species Conservation' (hereafter 'IUCN ex situ guidelines') to provide guidance on whether inclusion of ex situ management in a species conservation strategy is justified to address conservation goals/objectives. **These IUCN ex situ guidelines are intended to be complementary to the above IUCN translocation guidelines.**

The term 'ex situ' can encompass a range of management techniques that fall along a continuum of management intensity. **The IUCN ex situ guidelines define ex situ as 'conditions under which individuals are spatially restricted with respect to their natural spatial patterns or those of their progeny, are removed from many of their natural ecological processes, and are managed on some level by humans'**.

The guidelines outline the potential utility of ex situ management to:

- 'address the causes of primary threats',
- 'offset the effects of threats',
- 'buy time' (by establishing an 'insurance' population), and
- 'restore wild populations' (through translocations).

Ex situ management should be evaluated within the context of overall objectives for a species' conservation. **The SSC outlines a 5-step logical decision-making process for evaluating whether ex situ management is appropriate to include within a species' conservation strategy:**

1. 'Compile a status review of the species, including a threat analysis.'
2. 'Define the role(s) that ex situ management will play in the overall conservation of the species.'
3. 'Determine the characteristics and dimensions of the ex situ population needed to fulfil the identified conservation role(s).'
4. 'Define the resource and expertise needed for the ex situ management programme to meet its role(s) and appraise the feasibility and risks.'
5. 'Make a decision that is informed (i.e. uses the information gathered above) and transparent (i.e. demonstrates how and why the decision was taken).'

Though appropriate caution must be taken when considering ex situ management as a conservation tool, waiting until a species is very near extinction reduces the chance that such an approach will be successful. If a decision is made to proceed with an ex situ management program, a number of other steps must be taken to ensure the program is conducted responsibly and effectively. These include relying upon the IUCN translocation guidelines, as well as other appropriate guidelines.

4 BACKGROUND INFORMATION ON THREATS

Closely examining and understanding threats to a species' persistence (as well as its classification, life history, ecology) is a critical step in any responsible and effective translocation program (IUCN/SSC 2013).

Predation is recognized as the predominant proximate threat to boreal caribou populations throughout their Canadian range (ASRD and ACA 2010, Festa-Bianchet et al. 2011, EC 2012). Wolves appear to be a major predator of caribou (e.g., Stuart-Smith et al. 1997, Rettie and Messier 1998); however, black bears are increasingly acknowledged as important predators of caribou, especially caribou calves (e.g., Latham et al. 2011a, Leclerc et al. 2014). In addition, other species such as lynx, coyote and golden eagle have been documented preying on caribou calves; however their impact as population limiting factors is not known.

Unsustainable predation rates are thought to ultimately stem, at least in part, from habitat fragmentation and alteration resulting from industrial and agricultural land use as well as natural disturbances (particularly forest fires; Thomas and Gray 2002, EC 2012). The relationship between disturbance and increased predation on caribou may be explained by 3 major mechanisms (ASRD and ACA 2010):

- disturbance drives increases in densities of alternate prey and in turn, predators,
- disturbance compromises caribou's ability to spatially separate themselves from alternate prey and predators, and/or
- disturbance increases the occurrence, movement and hunting efficiency of predators on caribou.

As caribou require large, continuous tracts of suitable habitat, habitat fragmentation affects caribou populations by reducing both food and space necessary for caribou to adequately meet their life requisites (Badiou et al. 2011). Cumulative disturbance from both human and natural sources may reduce functional habitat (reduced use of suitable habitat due to avoidance behavior or increased mortality risk) for caribou more than direct habitat loss alone, which may ultimately influence caribou population dynamics (Weclaw and Hudson 2004, Sorensen et al. 2008, Johnson et al. 2015). Therefore, **habitat alteration is regarded as the ultimate threat to boreal caribou** (Badiou et al. 2011, EC 2012).

Additional factors, such as disease and parasites, hunting, forage quantity and quality, and stochastic events may affect caribou populations to differing extents (EC 2012). Although specific consequences remain uncertain, climate change is also predicted to have potentially serious impacts on caribou populations through both direct and indirect mechanisms (Thomas and Gray 2002, Hummel and Ray 2008, Vors and Boyce 2009, EC 2011, Vors 2013, Dawe et al. 2014).

Ultimately, many of these threats likely act in combination to have cumulative impacts on caribou that are not evident when examining individual threats separately (Weclaw and Hudson 2004, Culling and Cichowski 2010, EC 2012).

5 POPULATION STATUS (OF BOREAL CARIBOU IN WESTERN CANADA)

Nationally, the boreal population of woodland caribou is listed as Threatened under Schedule 1 of SARA due to widespread population declines and increasing threats posed by human activities, which inform a projected population decline of greater than 30% over 3 generations (~20 years; EC 2012, COSEWIC 2014). Environment Canada (2012) has concluded that the recovery of all boreal caribou local populations is both technically and biologically feasible, and has set a recovery goal to "achieve self-sustaining local populations in all boreal caribou ranges throughout their current distribution in Canada, to the extent possible".

Conservation status for boreal caribou in Western Canadian provinces and territories are:

- *Alberta*: Threatened under the Alberta Wildlife Act
- *British Columbia*: On the provincial Red List and a priority 1 species under goal 3 of the BC Conservation Framework
- *Saskatchewan*: Not listed; though a status report released in 2000 recommended listing woodland caribou as threatened (Government of SK 2013a)
- *Northwest Territories*: Threatened under the Species at Risk (NWT) Act
- *Yukon*: Not listed, but may be irrelevant for boreal caribou (see below)

Global conservation status:

- *IUCN*: Least Concern (*Rangifer tarandus*)
- *NatureServe*: G5TNR [caribou globally secure (G5), but boreal population not yet ranked (TNR)]

Of 51 boreal caribou local populations in Canada, 14 are considered "self-sustaining", 26 are "not self-sustaining", 10 are "as likely as not self-sustaining" and 1 is "unknown" (Figure 2), wherein self-sustainability refers to the ability of a range to support the local population and depends upon the amount and quality of suitable habitat (largely determined by the extent of disturbance; EC 2012)[2]. **Of 37 local populations for which data on population trends is available, 81% are declining** (COSEWIC 2014).

[2] Environment Canada used a habitat-based Bayesian decision support system to predict whether the extent of disturbance within a given caribou range would support a viable local population (EC 2011, 2012). These predictions were supported by results of an empirical, long-term, multi-population monitoring study in Alberta (Hervieux et al. 2013).

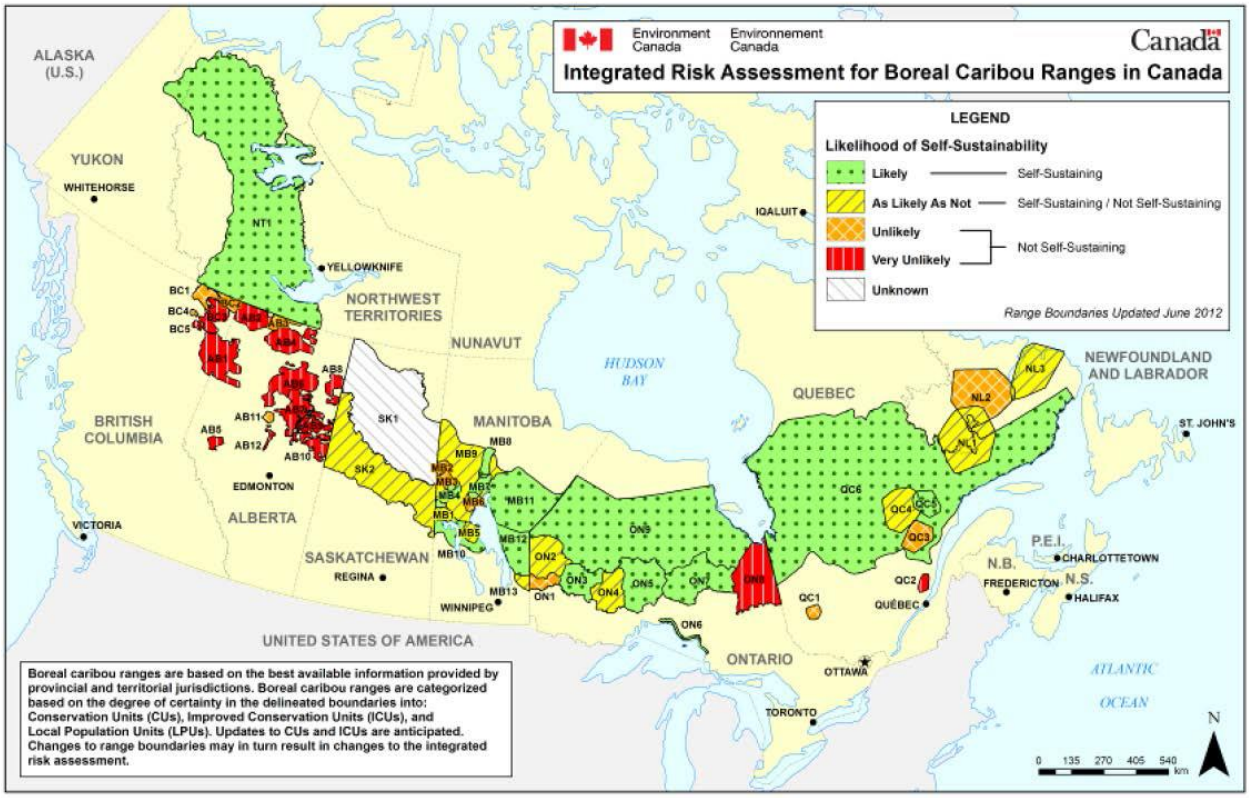

Figure 2: "Integrated risk assessment for boreal caribou ranges in Canada, reflecting the capacity of each range to maintain a self-sustaining local population of boreal caribou" (from EC 2012)

Caribou population trends are likely driven by both adult female and calf survival rate rather than just one vital rate (Hervieux et al. 2013). Assuming an average adult boreal caribou female survival rate of 85%, Environment Canada (2008) suggested that a recruitment rate of 15% female calves into the total population is needed to achieve population stability; assuming a number of demographic variables, calf recruitment must be at least 28.9 calves:100 females to achieve this. Using a non-spatial PVA, Environment Canada (2008) also estimated that populations numbering greater than 300 animals can "persist indefinitely when range conditions support average adult female and calf survival", populations numbering 50 - 300 animals are vulnerable to stochastic events and are at risk of 'quasi-extinction', and populations of fewer than 50 individuals face particularly high risk of extinction.

We compiled published data on estimated population sizes and trends (Figure 3) and demographic information (adult survival and calf recruitment; Table 1). Additional information on boreal caribou within SK and NWT/YT (which are poorly represented in the figures/tables) is also included.

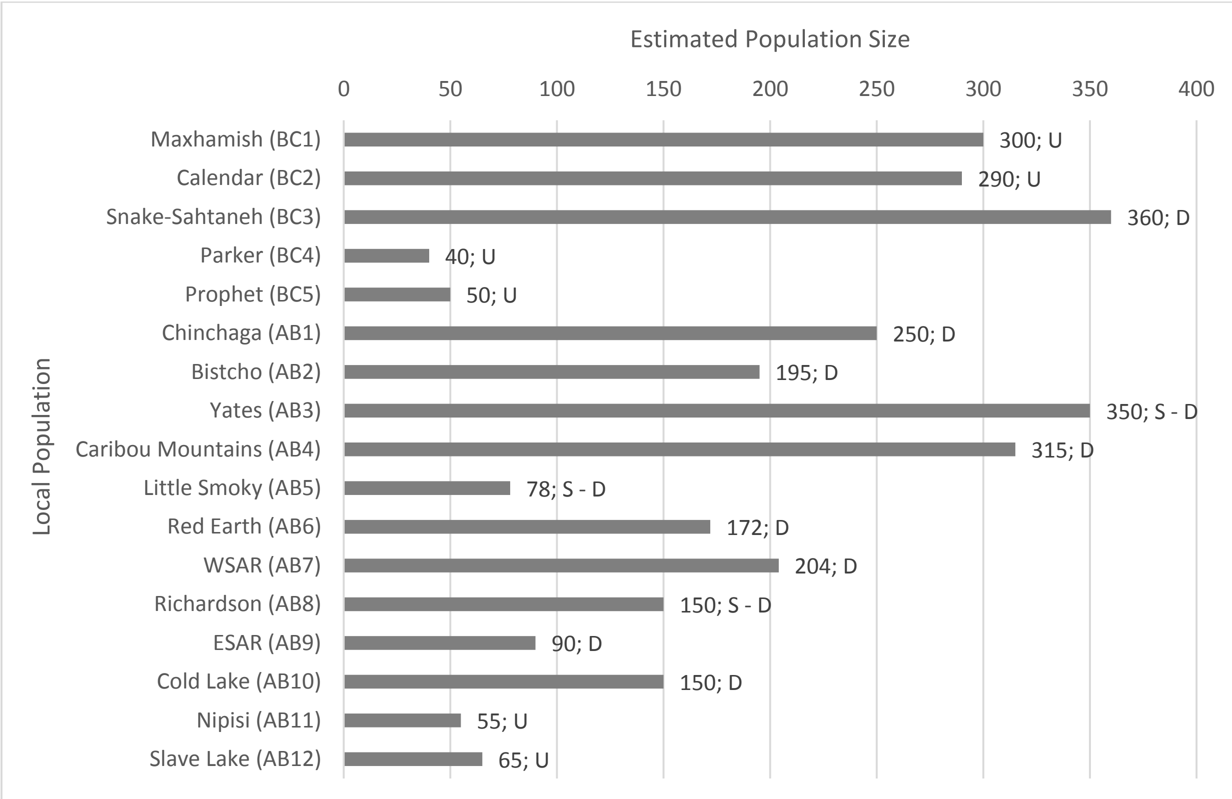

Figure 3: Estimated population sizes and trends for recognized boreal caribou local populations in BC and AB (note: some are cross-border). Numbers indicate total estimated population size; Letters indicate population trend: D = Decreasing, S = Stable, U = Unknown. ESAR = East Side Athabasca River, WSAR = West Side Athabasca River. Data is from EC 2012 (for population size: lower estimate), except for AB population trend information updated from Hervieux et al. (2013). EC 2012 population count for Chinchaga includes the BC portion. Note: BC population size information from Culling and Culling 2014 (minimum late winter population count 2014): Maxhamish = 102, Calendar = 79, Snake-Sahtaneh = 241, Parker = 40, Prophet = 37, Fort Nelson = 10 (an additional area of habitat outside of defined ranges for which a growing body of evidence supports formal inclusion in future revisions of BC's boreal caribou range map); Chinchaga and Chinchaga RRA combined count for BC = 214. Estimates for ranges in SK and NWT/YT have not been included due to lack of data in EC 2012 and little to no updated information available since EC 2012. In addition, data may potentially be misinterpreted when comparing estimates in BC and AB, where herds within small geographical areas have been distinguished, to SK and NWT/YT, where boreal caribou are considered to be dispersed over a much larger area of land (e.g. Estimated 6,500 animals total within NWT/YT, EC 2012). Available information on population size and trends for SK and NTW/YT is detailed below. Note: As of the 2012 Recovery Strategy, all local populations in AB and SK were considered ‘not self-sustaining’.

Table 1: Demographic information for Western boreal caribou local populations

Recognized Population[1]	Range Identification	Recruitment (9 - 11 months)	Annual Adult Female Survival	Year	Reference
Maxhamish	BC1	10 calves: 100 females[2]	0.72 (annual finite avg. all populations[3])	March 2014 (recruitment); April May 2013 – April 2014 (female survival)	Culling and Culling (2014)
Calendar	BC2	13 calves:100 females			
Snake-Sahtaneh	BC3	11 calves:100 females			
Parker	BC4	32 calves:100 females			
Prophet	BC5	10 calves:100 females			
Fort Nelson[4]	n/a	0 calves:100 females			
Chinchaga	AB1	13.4 calves:100 females[5] (AB)	0.831	1994 - 2012 (10 years, averages)	Hervieux et al. (2013)
Bistcho	AB2	17.1 calves:100 females	0.776	1994 - 2012 (5 years, averages)	
Yates	AB3	20.6 calves:100 females	0.907	1994 - 2012 (5 years, averages)	
Caribou Mountains	AB4	14.4 calves:100 females	0.858	1994 - 2012 (17 years, averages)	
Little Smoky	AB5	15.3 calves:100 females	0.901	1994 - 2012 (13 years)	
Red Earth	AB6	15.7 calves:100 females	0.819	1994 - 2012 (15 years, averages)	
West Side Athabasca River	AB7	19.8 calves:100 females	0.849	1994 - 2012 (18 years)	
Richardson	AB8	17.9 calves:100 females	0.903	1994 - 2012 (3 years, averages)	
East Side Athabasca River	AB9	14.7 calves:100 females	0.853	1994 - 2012 (17 years, averages)	
Cold Lake	AB10	10 calves: 100 females (AB) 12.6 calves:100 females (SK)	0.814 (AB), 0.866 (SK)	1994 - 2012 (12 years, averages)	
Nipisi	AB11	n/a[6]	n/a	n/a	n/a
Slave Lake	AB12	n/a	n/a	n/a	n/a
South Slave/SE Dehcho	NWT	23 calves:100 females (South Slave) 34 calves:100 females (Dehcho-south)	0.859 (South Slave) 0.76 (Dehcho-south)	2003/04 – 2009/10 (South Slave) 2005/06 – 2009/10 (Dehcho-south)	Studies cited in SRC (2012)
Dehcho (N/SW)		34 calves:100 females (Dehcho-north)	0.794 (Dehcho-north)	2005/06 – 2009/10	
North Slave		n/a	n/a	n/a	n/a
Sahtu		n/a	n/a	n/a	n/a
Inuvialuit		n/a	n/a	n/a	n/a
Gwich'in		n/a	n/a	n/a	n/a

Note: Little data available on SK populations. See section below for more information.

[1] EC 2012, except for NWT which are from COSEWIC 2011

[2] Avg. (March 2013, March 2014): 19c:100F BC1, 24c:100F BC2, 17.5c:100F BC3, 18c:100F BC4, 14.5c:100F BC5

[3] Including Chinchaga

[4] Additional area of habitat outside of defined ranges

[5] Milligan and Etthithun cores and Chinchaga RRA (combined) for BC: 10c:100F March 2014, 14c:100F March 2013 (Culling and Culling 2014)

[6] Data not available

Supplementary Information on Population Status for SK and NWT/YT

Saskatchewan

Research on boreal caribou in SK only began as of the late 1980s (Thomas and Gray 2002) and there remains a general lack of information concerning boreal caribou in SK to date (EC 2012, Government of SK 2013a). Research conducted in central SK in the 1990s estimated an average adult female survival rate of 84% and average calf recruitment of 28 calves:100 females (Rettie and Messier 1998). Between 2004 and 2008, radiocollared caribou in the Prince Albert Greater Ecosystem (PAGE) experienced an average annual adult female survival rate of 73% (Arsenault and Manseau 2011).

As of the 2012 Recovery Strategy, population size estimates and trends were unavailable for both of the two conservation units (low certainty) within SK (EC 2012). One range was considered ‘as likely as not self-sustaining’ (based on given habitat conditions) and the other ‘unknown’ due to high fire and low anthropogenic disturbance factors, which could not be modeled (EC 2012). As of 2000, there were an estimated 4,300 caribou in SK (Government of SK 2013b).

Northwest Territories and Yukon

Boreal caribou only just enter the northeastern corner of the Peel watershed within the Yukon—some caribou from the NWT MacKenzie core study area move into the Yukon during certain seasons (Nagy et al. 2004). Therefore, the Yukon Territory may have limited impact upon boreal caribou management (R. Farnell pers. comm. 2015).

As boreal caribou in NWT do not appear to form cohesive herds, one continuous range is defined for boreal caribou in NWT (and YT; EC 2012). As of the 2012 Recovery Strategy, this NWT ‘improved conservation unit’ (medium certainty) was estimated to hold 6,500 animals and was considered ‘self-sustaining’ (EC 2012).

Trends for the entire NWT population are not known, but estimated growth rates for specific regions indicate that numbers were increasing in the Gwich’in study areas (Nagy 2011), decreasing in the Dehcho and Cameron Hills study areas (Larter and Allaire 2010, Kelly and Cox 2011) and decreasing to stable in the South Slave study area (Kelly and Cox 2011). However, these results should be interpreted within the context of abundance, as numbers vary between study areas—53% of NWT boreal caribou are found in areas where numbers are declining or stable (Dehcho and South Slave regions), 8% in areas where numbers are increasing (Gwich’in region) and 39% in areas where trends are unknown (Inuvialuit, Sahtu and North Slave regions; SRC 2012).

6 MANAGEMENT ACTIONS/TECHNICAL SOLUTIONS

While habitat management will be a key element of caribou recovery going forward, intensive population management will likely also be required for most boreal caribou populations in Western Canada in order to address both proximate/symptomatic (predation) and ultimate/systemic (habitat change) threats to boreal caribou. The range of population augmentation or management tools addressed in this Workshop include:

- predator and alternate prey control (including lethal and non-lethal methods),
- wild-to-wild translocations,
- captive breeding and release,
- captive rearing and release, and
- predator exclosure fencing.

For each of these tools, a body of scientific and technical evidence, practical experience, and logistic and cost considerations can be integrated into a structured decision making process to assess the suitability of individual methods for different caribou ranges and jurisdictions. Below we present a summary of these population management tools in the context of some relevant key considerations.

6.1 Predator and Alternate Prey Control

We summarized 12 predator control programs conducted within North American caribou ranges from 1967 to current-day (Appendix 1). We only included predator control programs in which caribou were the (or one of the) ungulate species targeted for recovery; a number of other programs have been conducted to address moose declines (see NRC 1997 and Russell 2010 for reviews).

6.1.1 Predator Control

Overall, in areas where predators are abundant and are the primary cause of mortality, sufficiently intense reductions in predators have been associated with caribou population growth, which appear to have been driven mainly by improved calf recruitment (e.g. Gasaway et al. 1983, Boertje et al. 1996, Bergerud and Elliot 1998, Hayes et al. 2003).

In a review of wolf management programs in Alaska, YT, BC, AB and NWT, Russell (2010) concluded that wolf control is effective if:

- wolf predation is a limiting factor to ungulate populations,
- predators can be reduced to sufficient levels (65 – 80% of pre-control wolf levels),
- reductions are conducted until a population goal is reached or for at least 4 years,
- predators are reduced over an adequate area (at least 10,000 km^2),
- habitat is not limiting caribou population growth,
- hunting of caribou is diminished (ideally banned).

a) Have caribou survival and/or recruitment responded to past predator reductions?

Most wolf control programs we examined saw improvements in calf recruitment and some were also associated with increases in adult survival, but the statistical significance of these observations varied across studies that tested for significance. The highest recruitment rates recorded during wolf control were an average of 42 calves:100 females (in October) for the Aishihik northern mountain caribou herd in the Yukon (when the wolf population was reduced by 69-83% below 1992 pre-treatment densities between 1993-1997; Hayes et al. 2003), and 39-65 calves:100 females (in September/October, 1976 - 1981) for the Delta barren-ground caribou herd in Alaska (when the wolf population was reduced by 55 – 80% below pre-control numbers between 1976 – 1981; Boertje et al. 1996). However, increases in recruitment observed during wolf control within the Little Smoky boreal caribou population (mean recruitment = 12 calves:100 females pre-treatment, 19 calves:100 females post-treatment) were weaker than those observed in the Yukon (Hervieux et al. 2014).

b) Have caribou populations responded following past predator reductions?

Many caribou populations (though not all) appear to have responded to decreases in wolf densities, often beginning to grow in numbers the year of or immediately following initial reductions (e.g. Gasaway et al. 1983, Boertje et al. 1996, Bergerud and Elliot 1998, Hayes et al. 2003). The highest caribou population growth rates recorded were during wolf control in the Yukon and Alaska—the Finlayson northern mountain caribou herd in the Yukon increased at a finite rate of increase of $\lambda = 1.18$ (1986 – 1990) when the wolf population was reduced by 42% from the original population size in 1983 and by 83-86% from the original population size from 1984 to 1989 (Yukon Department of Environment unpublished data); the Delta barren-ground caribou herd in Alaska increased by $\lambda = 1.16$ over 7 years of wolf control (70-80% removal from the pre-control population during the first 5 years, 55-60% during the last 2 years; Boertje et al. 1996); the Aishihik northern mountain caribou population in the Yukon increased by $\lambda = 1.15$ over 5 years of wolf control (69 – 83% removal from the pre-treatment density; Hayes et al. 2003, Farnell 2009). However, though the annual rate of population change of the Little Smoky boreal caribou population increased 4.6% between pre- (2000 – 2005/06) and post- (2005/06 – 2012) control periods, wolf control (~45% removal of mid-winter wolf population each year) did not generate caribou population growth (λ post wolf control = 0.99; Hervieux et al. 2014).

c) How intensive must removal be to have an impact on caribou?

Relationships (i.e. linear or otherwise) between wolf reductions and caribou responses have not been established. However, **it appears likely that at least a 60% and ideally an 80% reduction (threshold) in a wolf population is required to generate responses in caribou survival, recruitment and/or population size**. In a review of a number of wolf control programs, Adams (2010 in Russell 2010) assessed 3 programs as 'successful' (i.e. short term goals for ungulate populations were met), all of which removed 69 – 77% of wolves for 6 – 7 years (moose and caribou populations grew 10 – 15% per year).

When planning predator control, the ecology of the predator species, including its movement, is an important consideration. **If predators are highly mobile and predator control occurs only on a local scale determined by small caribou ranges, predator movement into the target area may undermine control efforts** (Mosnier et al. 2008).

d) Is predator control sustainable over the long-term?

All reduction projects we examined in which wolf densities were measured reported rapid recovery of wolf populations following control (e.g. Boertje et al. 1996, Bergerud and Elliot 1998, Hayes and Harestad 2000, Hervieux et al. 2014)**.** Perhaps accordingly, caribou populations have been found to decline after wolf control has ended, though this may occur after a lag period (e.g. Boertje et al. 1996, Yukon Department of Environment unpublished data).

These findings suggest that predator control, if used alone, must be continuously conducted to maintain low wolf numbers (Thomas and Gray 2002, Festa-Bianchet et al. 2011). Therefore, while it may be an effective short-term option to protect caribou herds while other responses are being developed or in tandem with other actions (e.g. translocations or maternal penning), habitat management will be required over the long term.

e) Logistics- methods used to reduce predator populations and costs

i. Lethal Control

The majority of wolf reduction programs have used annual aerial control (shooting from helicopters) to successfully reduce wolf populations (e.g. Boertje et al. 1996, Bergerud and Elliot 1986, 1998, Hayes et al. 2003, Hervieux et al. 2014). **Biologists from the BC Mountain Caribou Science Team and Ministry of Environment strongly recommend aerial control as the most humane and cost-effective strategy** (Wilson 2009). Wilson (2009) provides detailed recommendations on predator-prey management (including cougars, bears) to support mountain caribou recovery in BC.

Costs associated with predator control programs will depend upon various factors, including the species, control area size and location/accessibility, control method chosen, intensity of control, measured effectiveness of control, duration of the control program and associated monitoring. Wolf control within the Little Smoky boreal caribou range in Alberta cost approximately \$35 CAD/km^2 per year (D. Hervieux pers. comm. in Schneider et al. 2010). Based on this value, Schneider et al. (2010) estimated that costs of conducting wolf control programs within 12 woodland caribou herds for 0 – 16 years (depending upon the herd) in northern Alberta would range between \$0 – 8.38 million CAD (average \$3.59 million CAD) in total[3].

[3] The length of time wolf control was conducted depended upon how long it was needed to recover each herd. Wolf control was applied when caribou density was < 0.045 animals/km^2 and stopped when caribou density was > 0.06 animals/km^2.

ii. Non-lethal control methods

Surgical sterilization

Surgical sterilization of dominant wolf pairs has been tried in combination with lethal control methods in the Yukon (Hayes et al. 2003, Farnell 2009) and BC (Hayes 2013), and with translocation of subordinate individuals in Alaska (Boertje and Gardner 2003; see Appendix 1). **Sterilization has been found to successfully stop reproduction and reduce wolf population growth** (Boertje and Gardner 2003, Hayes et al. 2003, Hayes 2013), while not affecting wolf territoriality, pair bonding or survival (Farnell 2009, Hayes 2013).

Sterilization treatment in combination with other control measures has been associated with concurrent increases in caribou populations (Boertje and Gardner 2003, Hayes et al. 2003, Farnell 2009, Hayes 2013). However, to our knowledge, sterilization has never been implemented alone (i.e. not without other control measures).

Reproductive inhibitors

There has been some research and experimentation in using reproductive inhibitors, either orally or through vaccines, to control reproduction in a number of different species (Fagerstone et al. 2010, Massei and Cowan 2014, Cohn and Kirkpatrick 2015). For example, Bisdiamine (steroid) was administered to wolves in ground meat daily and seemed to suppress spermatogenesis without a change to mating behaviour (Asa et al. 1996). However, to our knowledge, experimentation with reproductive inhibitors has not yet gone past the testing phase for wolves.

Various technical, biological, economic and legal challenges arise when considering widespread application of any reproductive inhibitor, whether administered orally, through implants or vaccines (Fagerstone et al. 2010)**.** For example, steroids would require repetitive applications as they are only effective over a short period, some persist within food chains and they can have negative health effects in some animals (Fagerstone et al. 2010). PZP has been found to disrupt estrous cycles in deer, which could alter the timing of births (Fagerstone et al. 2010). Amongst the biggest challenges may be obtaining regulatory approval. Furthermore, many agents are not species-specific and may thus affect non-target animals. Finally, some treatments may change the target species' behavior, such as mating or aggressiveness.

Reproductive inhibitors must also be biologically practical to use. The relative efficiency, as measured by the percent decline in population size relative to the number of animals sterilized or removed, of using contraceptive techniques as compared to lethal control is predicted to depend on the species' age of first reproduction and average adult survival rate (Dolbeer 1998). For animals that first breed at the age of 1 or 2, lethal control is predicted to be more efficient than contraception when adult survival is higher than 0.56 and 0.23, respectively (Dolbeer 1998). Lethal control will always be more efficient, regardless of adult survival, for animals that first breed at the age of 3 or older (Dolbeer 1998)**.**

Finally, using reproductive inhibitors must be economically practical and socially acceptable. The cost of their implementation will vary depending upon costs associated with development and regulatory approval processes, as well as actual treatment, which will involve human and technical resources. Generally, reproductive inhibitors are thought to be more accepted by the public than other methods of control, particularly lethal control.

Discretionary feeding of predators

Providing predators with alternative food sources ('discretionary feeding') was attempted in 4 cases in Alaska between 1985 and 1996 as a non-lethal method of predator control (NRC 1997, Russell 2010). However, results from the four cases were mixed. Boertje et al. (1995 in Russell 2010) highlight the high time and cost requirements of discretionary feeding and rank this technique as 'low' for cost-effectiveness.

6.1.2 Alternate Prey Control

Although the utility of alternate prey control in caribou conservation has been demonstrated theoretically (e.g. Weclaw and Hudson 2004), to our knowledge there has only been one study to date that has examined the effects of reducing moose on caribou populations (Serrouya 2013; though see Steenweg 2011). Some evidence also suggests that in addition to moose, white-tailed deer should be included as a priority species in prey reduction programs for caribou management (Latham et al. 2011b).

a) Is managing hunter harvests effective in reducing target prey populations?

Serrouya (2013) monitored moose, wolf and caribou populations following a BC government policy that started in 2003 and increased hunter harvest of moose 10-fold between 2003 and 2005 (lower harvest level 2005 - 2010) in 3 southern mountain caribou ranges. **Overall, the moose population declined by 71% from 2003 to 2011** (1.58 moose/km^2 to 0.44 moose/km^2), but data suggested the actual decline began 1 – 2 years after increased moose harvest started in Autumn 2003 (Serrouya 2013). The moose population decline was thought to have been stimulated by hunting, but ultimately driven by depensatory predation by wolves (Serrouya 2013).

Steenweg (2011) also investigated the effects of reduced moose populations within a BC southern mountain caribou range. Increased moose hunting quotas within the Hart Ranges southern mountain caribou range began in 2006 and ultimately led to a decline from ~3,000 moose (1.18/km^2) in 2005 to ~1,818 moose (0.73/km^2) in 2008/09, a 50 – 60% reduction overall (Steenweg 2011).

b) Have alternate prey reductions influenced predator numbers?

The combined results of Serrouya (2013) and Steenweg's (2011) studies suggest that moose reductions may lead to a decrease in wolf numbers, likely due to wolf dispersal (as opposed to mortality). However, there is likely a time lag (possibly 2 – 3 years) between moose reductions and wolf population response.

c) Have alternate prey control programs affected caribou populations?

In Serrouya's (2013) study, **the resulting effects of moose reductions on caribou populations were mixed** (Serrouya 2013). The larger subpopulation within the treatment area, Columbia North, increased following moose reductions, while the smaller subpopulations, Columbia South and Frisby-Queest, continued to decline. Despite mixed findings, Serrouya (2013) warns against disregarding alternate prey control as an option if implemented alongside other management tools that address proximate and ultimate limiting factors.

d) How intensive must removal of alternate prey be to have an impact on caribou populations?

The required intensity of alternate prey reductions to generate responses in wolf populations remain largely unknown. Bergerud (2007) suggested that moose densities > 100/1000km^2 can support wolf densities greater than the maximum limit (6.5/1000km^2) required for caribou population stability. **Fuller's (1989) equation estimates that moose densities must be < 300 moose/1000km^2 to limit wolf densities to <6.5 wolves/1000km^2 and < 50 moose/1000km^2 to limit wolf densities to < 1.5 wolves/1000 km^2** (Wilson 2009). Wilson (2009) suggested that moose densities be reduced (under a 'natural disturbance regime') to 50 – 300 moose/1000 km^2 for mountain caribou recovery, depending on the status of the target caribou population.

6.2 Wild-to-wild Translocations

We summarized information from 57 caribou and reindeer translocations to Canada (40), the USA (11) and abroad (6) (Appendix 2); in some cases, caribou were released to a given location on more than one occasion and summarization of results from 22 introductions to Newfoundland are presented from Bergerud and Mercer (1989) rather than summarized individually. Caribou have been previously translocated for purposes other than conservation, which were also included.

Of the translocations examined, woodland caribou were used as source populations in 37 cases (65%) and reindeer or barren-ground caribou in 20 (35%). Boreal caribou were reintroduced to Charlevoix, Quebec in the early-1970s and to the Lake Superior region in Ontario in the 1980s. Boreal caribou from Saskatchewan and Quebec may also have been used in earlier reintroductions to Minnesota and Nova Scotia in the 1930s and 1960s, respectively. To our knowledge, translocations of boreal caribou in Western Canada have never been attempted, but several translocations of mountain caribou have occurred in British Columbia since the late 1980s.

a) How successful have previous translocations been overall?

It is difficult to define what constitutes a 'successful' translocation as objectives are rarely identified and there is no set end date (IUCN/SSC 2013). Objectives may include demographic targets (such as survival, reproduction and/or abundance), behavioral responses, ecological changes, genetic diversity and disease infection rates (IUCN/SSC 2013).

We did not attempt to state whether translocations were successful in establishing viable populations, but **of the 57 translocations examined, 37 populations (65%) were still present or presumed present as of the most recent information**, 17 (30%) were extinct and the status of 3 (5%) was unknown. We separately examined 38 translocations within North America (including the 22 Newfoundland introductions reviewed in Bergerud and Mercer 1989) that have occurred since 1960 and in which caribou are not ranched. Of these translocations, 28 populations (74%) were still present as of the most recent information, whereas 10 (26%) were extinct or presumed extinct.

Of 37 caribou translocations in Canada and the USA reviewed by Kinley (2009), in which 15 – 146 caribou were released, 67% were successful in establishing new herds or reinforcing existing herds. An earlier review by Bergerud and Mercer (1989) summarized 33 translocations of caribou in eastern North America between 1924 and 1985. Of these, 19 (58%) were deemed to be successful in establishing viable populations by the time of publication (Bergerud and Mercer 1989).

b) What are some factors that may affect translocation success?

iii. Low predation risk

Bergerud and Mercer (1989) concluded that caribou translocations will fail in areas where wolf density is greater than 10 wolves/1000 km^2. In 10 translocations reviewed by Kinley (2010) into areas with predators (including wolves and cougars) 6 reintroductions or reinforcements were successful, 1 reintroduction (of 6 animals) and 1 introduction (of 8 animals) were unsuccessful (likely due to wolf predation) and the long-term viability of 2 reintroductions was still undetermined, although they appeared to have been successful over the short-term.

It appears that definitive statements concerning the extent to which predation influences caribou translocation success cannot be made. However, **given knowledge of the substantial threat predation poses to existing boreal caribou herds and outcomes of previous translocations, predation likely poses a significant risk to released caribou.**

iv. White-tailed deer and disease

As carriers of *Parelaphostrongylus tenuis*, meningeal worm, white-tailed deer have been implicated in the failure of various caribou translocations in eastern North America (Bergerud and Mercer 1989). Of the introductions reviewed by Bergerud and Mercer (1989), all those released into ranges with high densities of white-tailed deer infected with meningeal worm failed and the authors thus concluded that caribou translocation cannot succeed in areas where white-tailed deer carry *P. tenuis*.

Translocated animals may also endanger an existing resident population by bringing new diseases to a region (IUCN/SSC 2013). Therefore, a comprehensive assessment of disease is a key component of any translocation plan (IUCN 2013) and all individuals to be translocated must be screened for potential disease and parasites prior to release.

v. Characteristics of source populations

Woodland caribou exhibit locally adapted behaviors, so similarities/differences in characteristics between source and target populations stand to affect a translocation's probability of success. **Ideally, translocated caribou would exhibit similar behavioural characteristics, have experience with comparable species and densities of predators, and make use of similar seasonal habitats as the target population.** However, translocations of caribou to the South Selkirks mountain caribou herd from both mountain and northern ecotype source populations indicated that translocations involving different ecotypes may be feasible (Compton et al. 1995).

vi. Size and composition of release groups

Generally, the probability that a translocation will succeed increases with the number of animals released (Griffith et al. 1989, Wolf et al. 1998, Forsyth and Duncan 2001). However, for large ungulate species, **some evidence suggests that there exists an asymptote at approximately 20 – 40 animals released at one time** (Griffith et al. 1989, Wolf et al. 1998, Forsyth et al. 2001).

Kinley (2010) recommends including at least 3 bulls (>2.5 years old) in every group of 20 caribou translocated to promote breeding and as many younger females (but >1.5 years old) as possible for the remaining 17 animals. Kinley (2010) further recommends excluding calves due to their lower chance of survival and difficulties in recognizing their sex from the air when monitoring.

vii. Dispersal

Dispersal away from the target area was observed in various caribou translocations. Adult caribou may be more likely to disperse away from the target area than calves when caribou are moved relatively short-distances (<100 km) due to efforts to return back to their original range (Young et al. 2001). Some evidence also suggests caribou released nearby (< 50 km) existing herds may leave the release area to join resident caribou (Bergerud and Mercer 1989).

Gonzales et al. (2015; Figure 4) recently **developed a Bayesian Belief Network as a structured decision-making tool to examine the feasibility of translocating woodland caribou** to reinforce a population in Pukaskwa National Park, Ontario. Though the network was developed specifically for the Pukaskwa caribou population, similar factors and network structure may be relevant to caribou in Western Canada.

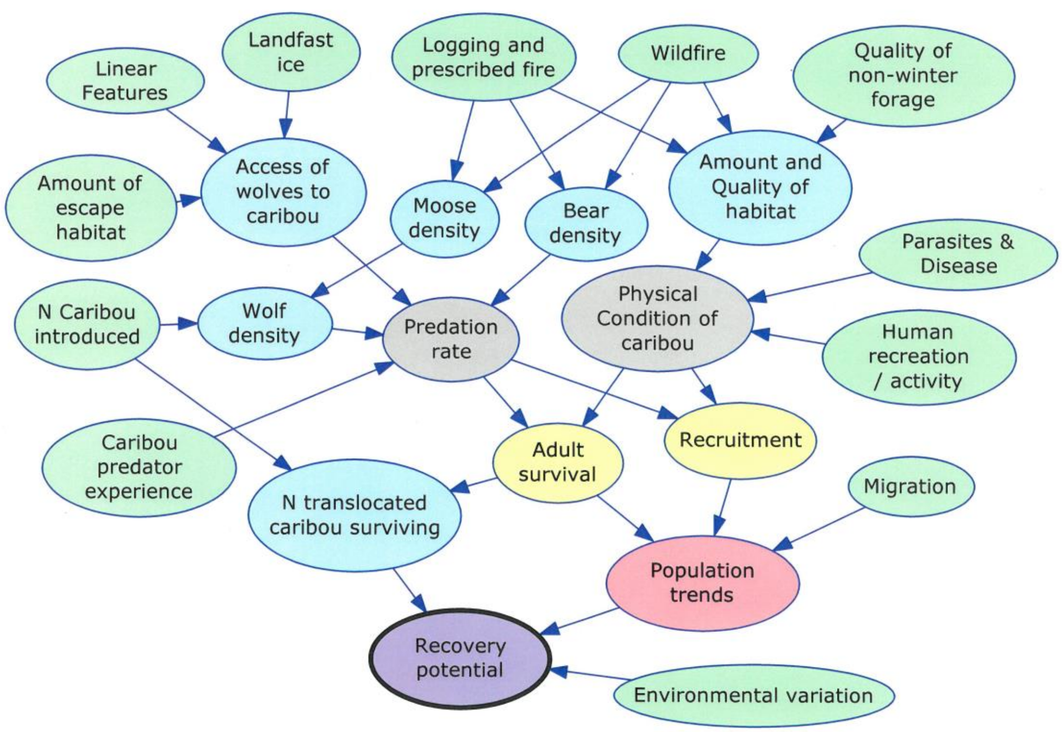

Figure 4: "Influence diagram underlying a Bayesian Belief Network for a proposed woodland caribou translocation into Pukaskwa National Park." (taken from Gonzales, E. K., Nantel, P., Rodgers, A. R., Allen, M. L., & Drake, C. C. 2015). Colours indicate network levels: Green = First level, 'parent'/'input' nodes, either environmental conditions or management modifications that affect caribou persistence. Blue = Second level, 'child nodes', ecosystem variables that are directly affected by current conditions/parent nodes. Grey = Third level, variables that drive population dynamics. Yellow = Fourth level, demographic rates, which determine population trends (fifth level). The final level, i.e. the final "child node" is the outcome. (P. Nantel, pers. comm. 2016).

c) *Logistics and costs*

Prior to translocation, permits from all provincial, territorial and federal governments must be obtained and thorough consultation with each provincial / territorial government, First Nations and other stakeholders would need to occur, all of which may take a considerable amount of time (Kinley 2009).

Costs would vary depending upon a number of factors, including the number of caribou translocated and methods used to capture, move and release them, the locations and accessibility of source and target areas, the number of years over which translocations occur, monitoring plans and any complementary predator, alternate prey and habitat management, all of which also determine the number of staff required.

For a reintroduction to Banff National Park, assuming 20 animals translocated/year for 2 years from source herds in BC or southern Yukon, Kinley (2009) estimated that the cost of consultation, translocation and short-term monitoring (i.e. excluding any maternal penning or additional population surveys) would be $128,000 over the first year, $168,000 in year 2 and $61,000 in year 3, plus required staff time each year.

Note: See Kinley 2009, 2010 for further details on potential logistics for caribou translocations.

6.3 Captive-breeding

a) Captive-breeding vs. captive-rearing

Both captive-rearing and captive-breeding can be considered 'ex situ' management techniques (as described in Section 3.2) and fall along a continuum of techniques involving keeping caribou in captivity for a given period of time. While it can be difficult to categorize projects, it is important to distinguish between these methods, as differences in their management can have implications for released animals. Within this document, we consider 'captive-rearing' to be any situation in which caribou were held within a confined area for a short period of time (weeks to months), but breeding between individuals was not planned or managed. Formal 'captive-breeding' programs are considered to be those in which select animals are bred over a defined period to allow for releases involving breeders and/or offspring to an identified area for a conservation purpose.

b) Advantages and disadvantages of captive-breeding and release programs

One advantage of captive-breeding and release programs is that a limited number of founder animals could produce a large and predictable source population for later releases (i.e. it may not compromise wild populations to the same extent as multiple wild-to-wild translocations). Furthermore, if release into the wild is not deemed appropriate at the current time, captive-breeding without release may be used to establish 'insurance populations' that ensure the continued existence of the species until a suitable time for release (IUCN/SSC 2014).

However, captive-breeding and release programs for caribou are also likely to involve a number of challenges. In evaluating the utility and feasibility of translocation in the recovery of the Banff caribou population, Kinley (2009) suggested that wild-to-wild translocation be chosen over captive-breeding due to the lack of any large-scale breeding facility for caribou, greater planning and higher costs associated with captive-breeding, higher risks [of disease] involved in rearing caribou in close proximity to other animals, the possibility that captive-reared caribou would be more 'naïve' than wild-born caribou (i.e. may experience higher mortality risk from predation) and the time required to build the captive stock. There has also been some suggestion that captive caribou may experience lower fecundity than wild caribou (B. Irving pers. comm. in Whittington 2011). Finally, as with other techniques, the true conservation utility of captive-breeding is realized once translocations and releases into the wild are done and wild populations are increasing; therefore, the problem is not solved solely by creating and establishing a captive herd.

c) Previous experience with captive-breeding

Reindeer husbandry has been practiced in Eurasia since as early as the 9th century, and herding and ranching of reindeer for subsistence purposes was introduced to Canada at the end of the 19th century (Haigh 1991). Although the extent to which reindeer are domesticated has varied, captive management of the species for ranching purposes is relatively well-established. In addition, breeding and rearing of caribou commonly occurs at zoos around the world.

Captive-breeding and release programs for conservation purposes have been theoretically considered and concluded likely feasible (see references in Kinley 2009). However, **to our knowledge, formal**

captive-breeding of caribou for a conservation translocation has never been conducted. Several translocation projects examined held caribou within enclosures for a relatively long period of time (up to several years), during which time calves were born and raised within the enclosures (see Appendix 2: Charlevoix, Minnesota, Baxter State Park and Finland translocations). However, it is not clear whether these projects can be considered formal captive-breeding programs as the degree to which breeding was managed is unclear.

d) Logistics

Fundamental techniques have been established for successful captive-breeding and release programs, but the logistics of implementing a formal captive-breeding project for caribou conservation remain largely unknown. Therefore, research into all aspects of captive-breeding and release processes would be required to help determine best practices for success.

A captive-breeding and release program can be considered in 4 iterative stages: identification of founder animals for captive breeding, growth of the captive population, release(s), and post-release monitoring. Each of these phases would be planned in light of potential alternative actions, and in terms of risk—adequate risk assessments would need to be conducted as outlined in the IUCN/SSC translocation guidelines (2013). Calgary Zoo (2014) identified a number of research questions for each of the four stage that would need to be addressed for a caribou captive-breeding and release program:

1. Founding Stage
    - How many caribou (males and females) founders need to be captured to maintain desired genetic diversity in captivity over time, and to produce offspring for release?
    - Over how many years are captures from the wild necessary to satisfy genetic / demographic needs of the captive population over time?
    - Which diseases / parasites should be avoided in forming the captive population? Which parasites / serological adaptations are desirable to be retained?
    - Where are the most appropriate (i.e., genetics, behaviour, size) source population(s) of caribou to accommodate long-term goals of assisting recovery in British Columbia and Alberta?
    - What short / long-term effect does capture from the wild have on remaining source populations?
    - How can captures of wild individuals, transport, and subsequent acclimation to captivity best minimize stress and mortality?

2. Breeding Stage
    - Will behaviours change in individuals or among generations that might increase / decrease the suitability of caribou for release?
    - Which behaviours / physiological characteristics are associated with successful breeding?
    - Are behaviours associated with successful management in captivity (e.g. 'tameness'), well-aligned with post-release requirements for survival / breeding (e.g. 'predator avoidance')?
    - Can pre-release training improve behavioural suitability for release?

3. Release Stage
    - How many individuals can be released during which periods to maximize gains in the wild, while minimizing genetic diversity / demographic losses in captivity?
    - Which transport / release method(s) minimize post-release dispersal, maximize group cohesion, and for reinforcements result in herd formation with wild individuals?
    - Which transport / release method(s) result in maximum short and long-term survival / reproduction of released individuals?
    - What pre / post-release management (e.g. predator limitation, food supplementation, human access-restriction) can maximize short / long-term survival / reproduction of released individuals?
    - Under what conditions would released animals be re-captured and returned to captivity?

4. Management of reintroduced / reinforced populations
    - Which behaviours, habitats and predator densities result in maximum short or long-term survival / reproduction of released individuals?
    - Is maternal penning useful / necessary to assist released individuals?
    - Is predator removal necessary for reinforcement / reintroduction success and if so under what habitat conditions?
    - Is genetic diversity retained in wild populations, do populations differ in genetic structure, and are certain genotypes most aligned with survival / reproduction?
    - Does reintroduction / reinforcement significantly improve the viability of mountain caribou?

Further break-down of these 4 stages (Figure 5) outlines numerous steps in a potential caribou captive-breeding and release program, each of which will involve decisions that must be guided by sound science and will affect overall project logistics and costs.

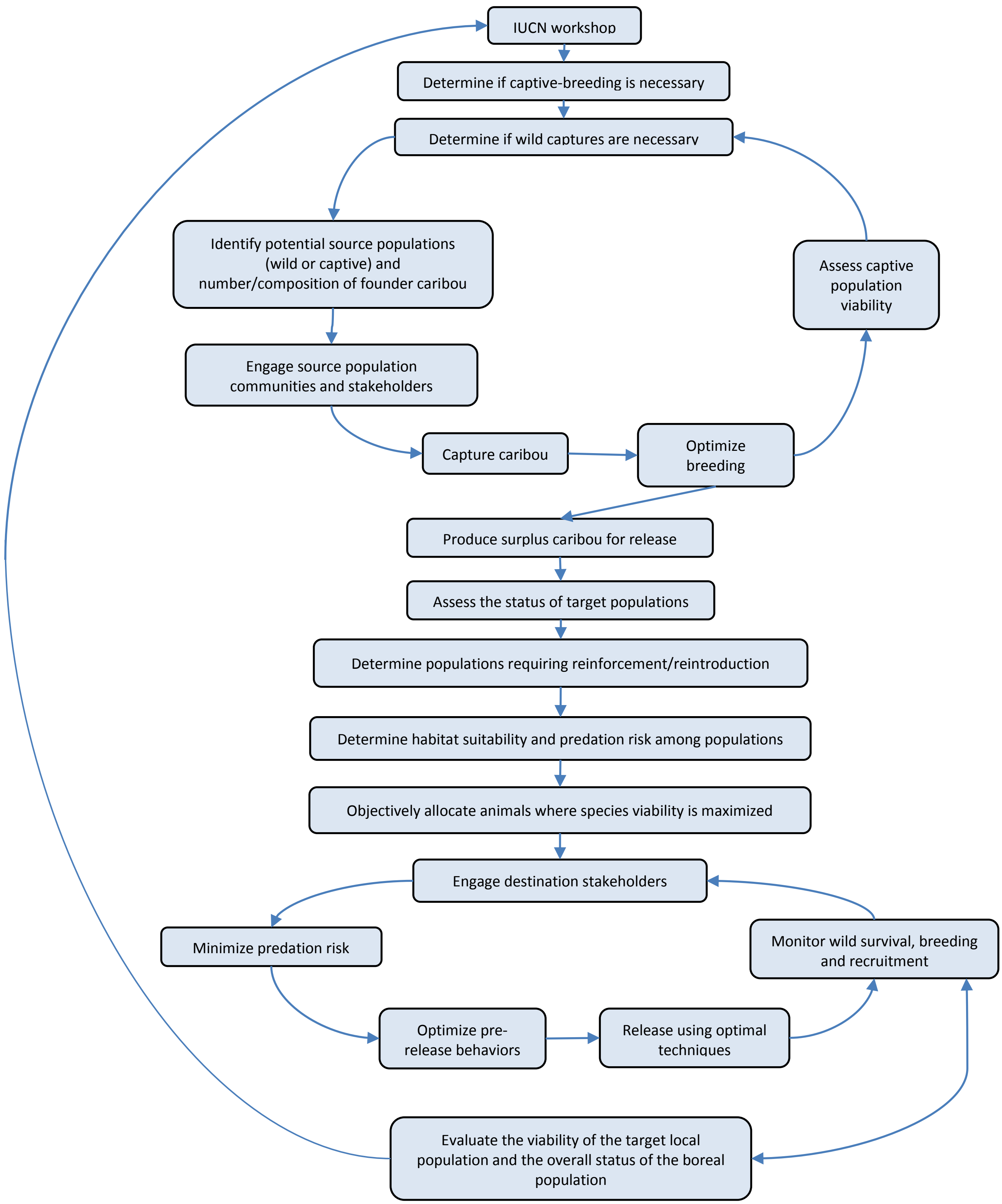

Figure 5: Steps in a potential caribou captive-breeding and release program (based on Calgary Zoo's 2014 flow chart)

6.4 Captive-rearing (Maternal Penning)

Maternal penning, also known as 'caribou rearing in the wild', is a head-starting technique that aims to increase calf survival by protecting calves from predation in the first few weeks of life (when mortality is generally highest).

"Professional Forester Kevin Bollefer helps a pregnant caribou as they toboggan her into the protective maternity pen" (RCRW 2015)

Pregnant females are captured in late-March or early-April and relocated to a protected pen within the herd's native range to give birth and raise their young. Specific methods have varied depending upon the herd size, project scale and resources available. Typically, pens have consisted of a ≥ 1.5 m geotextile fence, surrounded by one or two outer electric fences. Pen sizes have varied depend upon the number of captured individuals and project resources available, ranging between approximately 2 – 12 ha. Penned caribou are fed natural lichens upon first entering the pen and gradually transitioned to commercial reindeer pellets; feed is reverted back to lichen prior to release. In June or July, all calves and adults are released back into the wild.

We summarized 4 maternal penning projects that have been conducted within YT, AB and BC from 2003 to present-day.[4]

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[4] Note on another case of captive-rearing (not maternal penning): In another translocation project summarized (Adak Island, Appendix 2) 72 caribou calves were captured and hand-reared for 2 months prior to being released. Calves were reared in captivity for approximately 2 months prior to release, over which time they were bottle fed and most are reported to have become tame (Jones 1966). Nearly all losses while in captivity occurred within the first 2 weeks, totaling 68% of the captive population in 1958 and 69% in 1959, which suggests captivity-induced stress may have been the cause (Jones 1966). Once released, the caribou remained in a group and showed signs of imprinting until 1962, after which time the band dispersed and signs of imprinting were lost (Jones 1966). The population rapidly expanded to an estimated 2,600 – 2,800 in June 2012 (USFWS 2014b).

a) Are survival rates and/or recruitment higher for pen-born as compared to wild-born calves?

Previous projects have found mixed results regarding the effectiveness of maternal penning in improving calf survival, but conclusions are difficult to draw given the few projects conducted thus far, their scale (mostly pilot projects) and concurrent predator or alternate prey control programs that confounded results.

b) Does capture and penning itself appear to negatively affect captured caribou?

There appear to be few negative effects of penning on the well-being of caribou. Adult females seem to adapt well to captivity and give birth to live and healthy young. Adults and calves have also experienced relatively high survival while in the pen. **However, any potential long-term effects are unknown.**

"Revelstoke Caribou Rearing in the Wild (RCRW) team releases two pregnant caribou in the safety of the maternity pen and prepare to do an ultrasound on a third. Photo by Rob Buchanan." (from RCRW 2015)

c) *Have maternal penning projects impacted caribou population growth?*

Other than the Chisana project, all penning programs have been conducted as pilot projects (10 -18 penned female caribou, conducted over only 1-2 years). Furthermore, all 3 penning projects implemented in AB and BC have had predator or alternate prey reductions conducted within the herd's range over at least one of the years in which maternal penning occurred. Therefore, there is little empirical evidence that maternal penning projects impact caribou population growth.

Though the relatively large number of captured females (and calves raised) over the course of the Chisana project was too low in proportion to the wild population's size to generate population growth, **maternal penning may be an effective short-term option in the recovery of smaller at-risk populations of caribou** (CCRT 2010). Adams et al. (2006 in Kinley 2009) suggest ≥ 1/3 of all females within a population should be penned for this technique to be effective. Similarly, Serrouya et al. (2015) estimated that approximately 30% of the Columbia North subpopulation would need to be penned to generate 2% growth per year ($\lambda = 1.02$).

d) Logistics and costs

By utilizing natural fostering methods within a herd's native range, maternal penning may avoid problems regularly experienced in long-term captive-breeding and release programs including reductions in genetic diversity, disease risk, loss of natural instincts, and high costs. It may also present a more publicly-acceptable recovery option than some other techniques (e.g. predator control).

That being said, maternal penning is only likely to be successful in recovering caribou populations if conducted in conjunction with other conservation actions, such as habitat and predator-prey management (CCRT 2010, Smith and Pittaway 2011, Serrouya et al. 2015).

Maternal penning is generally expensive relative to other intervention options, such as translocation and predator control (though probably not more expensive than captive-breeding). However, the cost varies between projects depending upon the herd size and project-specific population growth objectives, which determine the number of caribou that must be penned (plus associated costs related to construction, staff, etc.), and the accessibility of the penned area.

The cost of penning individuals of the Little Smoky herd in 2006 was approximately $40,000 CAD per calf (Smith and Pittaway 2011). The anticipated cost of maternal penning for the Klinse-Za herd in 2014 (including planning/permitting, lichen/feed, camp construction, collars, pen construction, capture and transport, administration, and shepherding, but excluding any predator removal) was approximately $452,000 CAD (Klinse-Za maternal penning steering committee 2014a). Maternal penning, excluding any capture costs, was estimated to cost approximately $250,000 over the first year and $200,000 each subsequent year when considered as part of a translocation-aided recovery program for Banff (L. DeGroot pers. comm. in Kinley 2009).

Given the variability in cost between projects, cost:benefit ratios must be determined for individual cases and are expected to be more favorable for small, highly-endangered herds in relatively accessible areas (CCRT 2010).

6.5 Predator Exclosure Fencing

6.5.1 Concept

Fencing of large areas to protect endangered animals from threats has been attempted in a number of locations to date (Hayward and Kerley 2009). In particular, Australia, New Zealand and South Africa have adopted large-scale fencing to protect target animals from predation, overgrazing and over-hunting (Hayward and Kerley 2009). Fences can protect enclosed populations from any threat arising from direct human influence (e.g. habitat loss and alteration, hunting; Hayward and Kerley 2009). **'Predator-proof fencing' may address unsustainable levels of predation by preventing predator access to animals protected within an enclosed area** (Hayward and Kerley 2009).

6.5.2 The 'Big Fence' caribou project

The Oil Sands Leadership Initiative Land Stewardship Working Group (OSLI LSWG) first began investigating the technical feasibility of using large-scale predator exclosures for boreal caribou conservation in Alberta in 2012 (OSLI LSWG 2012). Since the assimilation of OSLI into Canada's Oil Sands Innovation Alliance (COSIA), this concept has been adopted and is now being advanced by COSIA as one prong of a multi-pronged strategy by the oil sands sector to contribute to recovery of boreal caribou in the oil sands region of northeast Alberta (Amit Saxena pers. comm.).

The concept involves enclosing a large area of caribou habitat (potentially hundreds of km^2) with a predator-proof fence to protect caribou from predation. The objective of a large predator exclosure would be to establish and maintain a viable caribou population that could also ideally be used to supplement other caribou populations. An existing population that is declining and facing a high risk of extirpation would be enclosed, although depending upon the population chosen, caribou may need to be captured from elsewhere to supplement the population.

A 'viable' population is here considered one "with stable or positive population growth; that is large enough to withstand random events (e.g., severe weather) and human-caused pressures; but requires ongoing management intervention to persist" (OSLI LSWG 2012). The established population would eventually be freed from the fence once suitable habitat has been restored (> 40 years) to generate a self-sustaining free-ranging population.

Participants of a workshop held by OSLI LSWG in May 2012 identified a number of potential benefits associated with conducting such a project within Alberta, which included promoting caribou population growth and generating a source population for releases and other translocations (OSLI LSWG 2012).

Fences can pose possible risks to the species they are meant to protect, which must be evaluated and weighed against potential benefits. Risks may be greater for species that require large areas due to the difficulty in creating enclosures big enough to meet their habitat requirements (Hayward and Kerley 2009). The 2012 workshop participants also recognized several challenges and assumptions, including the need for active management of fenced populations, the project's integration into a broader management program, and the potential risks of fire and predator breaches. (For additional information on benefits, challenges and specific planning resulting from the May 2012 workshop, see OSLI LSWG 2012.)

In 2014, the team tested fence designs to determine whether fences would hold up against predators in the area and which design works best to do so (Alexander 2014). A one-hectare test site was enclosed by a 2.5 metre high, page-wire fence with smooth sheet metal along the top (to prevent black bears from climbing over) and a metre-long skirt along the bottom to impede digging (Alexander 2014). Bait, such as moose carcasses, was placed inside the enclosure to attract predators and thereby test for any shortcomings in the fence's design (Alexander 2014). The fence appeared to be effective in excluding most predators; only one black bear penetrated the fence (Alexander 2014).

Given the technique's viability, experts and managers are now determining project details, such as how large an area the fence should enclose and how many caribou could be held within the enclosure (Alexander 2014). COSIA is currently advancing the big fence concept by conducting a scoping study to determine technical, regulatory and stakeholder engagement requirements for such a project. This project is currently underway, and reports or deliverables have not yet been completed (Amit Saxena pers. comm.).

6.5.3 Logistics and costs

Fence type and construction must be tailored to the prey and predator species of interest, as no one design appears to be effective in every situation (Hayward and Kerley 2009). As mentioned, a design for caribou fencing was recently tested and found to be relatively effective. It is unlikely that any fence will be completely effective 100% of the time (Long and Robley 2004) (e.g. the caribou experimental fence was breached by one black bear). Managers must decide what effectiveness level (i.e. how often the fence is breached) is considered sufficient to adequately recover the protected population (Long and Robley 2004). Bode and Wintle (2010) developed a return on investment framework as a systematic method to compare costs and benefits of fence designs.

Fencing projects are generally very costly due to high initial costs of construction, as well as longer-term expenses (Hayward and Kerley 2009). Maintenance is considered one of the most important determinants of fence effectiveness and the largest issue with any fencing project (Hoare 2003 as cited in Ferguson and Hanks 2010). 2012 Caribou workshop participants considered the standard life expectancy of a fence to be 25 years, with replacement required thereafter. Participants also expected that staff would need to inspect the entire fence at least weekly (with additional monitoring using remote cameras) and over the long-term, the fence would eventually need to be removed.

Logistics and associated budgets must also include any associated additional management activities, such as management of caribou, predators, alternate prey and other species, veterinary services to monitor and address caribou health, research and monitoring activities, habitat restoration, access management, etc.

Feasibility assessments commissioned by OSLI LWSGI (2012) suggested that implementation of a large-scale predator exclosure within caribou range would require a financial commitment in the order of $10 million to cover the costs of construction, annual maintenance and operating, and eventual fence removal (OSLI LSWG 2012 in Golder 2014).

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Appendix 1: Predator and Alternate Prey Control

Targeted herds (DU), Province	Years	Target species	Removal (#/proportion/densities)	Reduction methods	Responses in the targeted predator/alternate prey populations after control ended	Survival and recruitment responses in caribou populations	Caribou population responses	References
Charlevoix (Boreal), QC	1967 - 1979	wolves	8 wolves killed 1978 -79 (no info found on other years)	Snares and traps + shooting from helicopters 1978/99 (other years unknown)	The government continued to encourage wolf harvests after completion of the program (harvesting rate ~ 40 - 41% in the 1990s/2000s), but wolf numbers remained high.	The maximum calf survival 1978 - 1981 was recorded in 1979 (9/9 calves survived), which coincides with wolf reduction over the winter of 1978/79. Calf mortality averaged 21% between 1978 - 1981 and 57% 1998 - 2001. Adult survival was lowest in 1979 (87%), immediately following wolf control.	Stable at ~ 50 animals during wolf control; only started to increase in the 1980s, once the program had ended. Stabilized again at 100 – 125 individuals in the 1990s. Declined in 2000s, now stable at ~83 animals. Remain at-risk.	Sebbane et al. 2003, 2008, 2011; Jolicoeur et al. 2005; St. Laurent and Dussault 2012; Sepaq 2015
Gaspésie (Atlantic-Gaspésie), QC	First program: 1990 - 1996 Second program: 2001 - present?	Black bears and coyotes (some golden eagle)	Since 2001: Black bears: ~ 31 individuals/year removed (67% removal) Coyotes: ~15 individuals/year (136% removal)	Mainly trapping. Shooting and feeding sites tried, but abandoned.	Predator numbers have remained relatively high since control was restored in 2001, likely from immigration to target area.	Recruitment: ~20 calves:100 females in 1991, >50 calves:100 females in 1998, fell again in 1999 to <20 calves:100 females in 2000; increased to remain > 30 calves: 100 females from 2001 to 2008; starting in 2007, fell to < 20 calves:100 females by 2009.	Principal indicators of recovery were at a 10-year all-time high in 1997; population continued to increase for 2 years following the end of the program; but declined to 140 animals in 2001. After recommencement of control in 2001, the population expanded to ~200 animals in 2007, but later declined.	Mosnier et al. 2008, St. Laurent et al. 2009; ERCG 2006, 2011
Delta (Grant's), AK	1976 - 1982	wolves	70 - 80% below precontrol population (14.4 – 4.4 wolves/1000km²) each year 1976 - 1980; 55 - 60% below precontrol pop'n (6.6 - 8.4 wolves/1000km2) each year 1980 - 1982	Shooting from a helicopter or fixed-wing aircraft. Public trapping and hunting continued after control ended, but not enough to significantly affect wolf population.	Following the end of wolf control, the wolf population rebounded to near pre-control densities by 1985 (11.5 wolves/1000km2) and wolf numbers exceeded those before control by 1991 (15.7/1000km2).	Calf recruitment significantly increased following the start of wolf reduction in 1976. Calf survival from 1976 - 1979 also increased significantly in the 2 control herds, but to a lesser extent. Calf recruitment within the Delta herd during wolf control (1976 -1981) was between 39 - 65 calves:100 females; highest recruitment observed in 1979, 3 years after control began. Calf recruitment to 6 months was significantly negatively correlated to wolf numbers.	Pop'n increased from 1975 to 1989 (λ = 1.12 over entire period, 1.16 during 7 years of wolf control 1975 - 1982, 1.06 following wolf control, 1982 - 1989), followed by a decline from 1989 to 1993 (λ = 0.78). Delta increased in density from 183 to 891 caribou/1000km2 during the 14 years after control began vs. the Denali and Macomb (reference) herds remained between 100 - 370 caribou/100km2. Wolf control and favourable weather thought to have jointly allowed for caribou pop'n growth.	Gasaway 1983; Boertje et al. 1996; NRC 1997
	1993 - 1994	wolves	Removal of 62% of pre-control autumn 1993 population (15.4 wolves/1000km²) and 56% of pre-control 1994 population (10.6 wolves/1000km²)	Trapping and occasional shooting from the ground, but no shooting from aircrafts.	Wolf population rapidly rebounded to near pre-control levels.	Mortality of 4 - 16 month old female caribou declined from 60% before wolf control to 38% following wolf control, but mortalities attributed to wolves did not decrease. No change in survival of female caribou > 16 months before versus after wolf control. Avg. recruitment to Sept/Oct in the Delta herd: 7.4 calves:100 females 1992 -	The Delta caribou herd stopped declining and stabilized over the first year of wolf control; increased over the next 2 years at a rate of approximately 12%, but again declined through 2000. Population trends and recruitment estimates were similar within the	Boertje et al. 1996; Valkenburg et al. 2004

						1993 vs. 21.5 calves:100 females 1994 - 1995. Not significantly different from the Denali herd, but significantly higher than the Macomb Herd. Recruitment sig. declined after the program ended (1994 -1998) in the Delta herd, but not the Denali or Macomb herds.	adjacent Denali caribou herd. The Macomb herd declined from 1990 to 1995, then increased, but the trends were not as drastic as those seen in the Delta or Denali caribou herds.	
Kechika region, including Horseranch herd (northern mountain), BC	1978 - 1981	wolves	1978: 22 of 36 (61%) 1979: 25 of 29 (86%) 1980: 23 of 27 (85%) Wolf densities in reference herds 9 - 10 wolves/1000km² (1978 - 1981); Horseranch contained 10 wolves/1000km² before reductions, 0.8-3.8 wolves/1000km² following reductions.	Poisoned the first winter and shot from a helicopter the second and third winters.	See below.	Calf survival doubled during wolf control and the proportion of calves in the fall population significantly increased from ~6% in 1977 to 16 – 17% 1978-1980 in the Horseranch population. Calf survival decreased once again as the wolf population recuperated after program completion (<5% of populations by 1982). The percentage of calves in reference herds varied over the years, but averaged 10 - 13%. Examining all populations, the average percentage of 5-month old calves in the population was 15.1% vs. 7.5% and the average calf mortality was 70% vs. >85% in years when there were fewer wolves compared to years when there were more wolves, respectively. No direct measurement of adult survival available, but hunting suggested natural adult mortality rates were 8% for adults in the Horseranch herd when wolves were reduced and 12% when wolves recovered after control, versus 18 - 21% for adults in the reference herds.	The Horseranch population increased over the course of the program, from 246 animals in 1977 to 337 in 1982 (λ = 1.06), whereas the population of adjacent herds in which wolves were not reduced declined over the same time period (λ = 0.88, 0.89).	Bergerud and Elliot 1986
	1982 - 1987	wolves	70/88 (85%) in 1982, 89/107 (83%) in 1983, 105/138 (76%) in 1984, 157/242 (65%) in 1985 from entire Kechika region, including Horseranch Mountains	Shooting and local hunting	By Feb-March of each year following wolf reduction, the wolf population recovered to 81 – 97% of the pre-control population (recolonization). The more wolves that were removed, the more immigrated into the area. In 1987 (1 year after control ended), overall wolf density within Kechika was 17.5/1000km² (higher than the pre-removal density in 1982).	Caribou recruitment to 5 months within Kechika was significantly negatively related to the density of wolves prior to parturition. For all ungulates studied (caribou, sheep, elk, moose), recruitment of calves 5 - 9 months old in both the Kechika and Muskwa regions was correlated with the density of wolves before parturition. When wolves were reduced, average calf survival increased 2 - 5 times compared to control populations.	Mean λ values for the Kechika caribou population were 0.93 in years without reductions versus 1.14 with reductions (significantly different). The population generally increased when recruitment was high in years with low wolf densities.	Bergerud and Elliot 1998; Environment Canada 2012; Hegel and Russel 2013

Finlayson (northern mountain), YT	1983 - 1989	wolves	To 58% of original population size in 1983 (i.e. 42% removed), to 14 - 17% of original pop'n thereafter (1984 - 1989) (i.e.. 83 - 86% removed); original density: 10.3 to reduced: 1.4 - 1.8 wolves/1000km² over last 6 – 7 years of removals	Shooting from helicopter; incidentally trapped	Following the end of the program, wolves rebounded from 24 wolves in March 1989 to 240 wolves by March 1994 (~ 10.4 wolves/1000 km²).	Recruitment increased over wolf control, from 17 calves:100 females before control in Oct. 1982 to an average of 50.6 calves:100 females 1984 – 1990. When wolf control ended, recruitment decreased to 9 – 44 calves:100 females between 1990 and 2006. Calf recruitment was significantly higher during treatment vs. post-treatment years. Adult mortality decreased from 10 – 45% in 1982 – 1983 before wolf control to 0 – 22% in 1984 – 1987. Adult caribou mortality was strongly correlated to the number of wolves. Hunting may have increasingly added to mortality (3.8%) with caribou population decline.	Over the period of wolf control, the Finlayson caribou herd nearly doubled in size between 1983 and 1990. Between 1986 and 1990 alone it increased from 3,073 to 5,950. Exponential annual growth rate r= 0.17 (1986 – 1990). However, following the end of the program caribou showed a reciprocal response to wolf increases; the herd declined to 2,077 – 3,100 animals by March 2007and exhibited a decreasing population trend (r = -0.16, 1990 – 1999).	Farnell and McDonald 1986 in Thomas and Gray 2002; NRC 1997; Hayes and Harestad 2000; Hayes et al. 2000; Farnell 2009; Environment Canada 2012; Yukon Department of Environment, unpublished data
Aishihik (northern mountain), YT	1993 - 1997	wolves	69 - 83% below 1992 pre-treatment density; 8.2 wolves/1000km² (1992), 1.5 - 2.8/1000km² following wolf control (1993 - 1998)	Shooting from helicopter 1993 - 1997; surgical sterilization 1994 - 1997; experimental 'chemical-immuno-contraception experiments were conducted on alpha wolf pairs after 1997, but the data has not been released.	Unknown * Sterilization results: Sterilization prevented 12 breeding events that would have otherwise produced ~ 68 pups from 1994 - 1997. Sterilizations thus reduced wolf population increases by 11 - 58% (1995 - 1998). Territoriality, pair bonding and survival rate were unaffected by surgical sterilization. The authors concluded that sterilization was an effective method to control wolf population expansion.	Annual recruitment in the Aishihik caribou herd significantly increased from 15 calves:100 females pre-control to 42 calves:100 females during wolf control, whereas none of the other herds showed this trend. Recruitment was highest in Oct 1996, with 47 calves:100 females. The Aishihik herd lost proportionately fewer calves between July – Oct. during treatment (7 - 18%) as compared to pre-treatment (34 - 41%) and loses for the Aishihik herd were lower than that of the Wolf Lake herd (17 - 60%) over all treatment years (P<0.01). Although pre-treatment data for Aishihik adult survival rates were too variable to estimate trends, there was no evidence to indicate adult survival changed in response to wolf control and census interpolation methods estimated mean annual adult survival to be 0.87 and 0.91 before and during wolf control respectively.	When wolves were reduced, the Aishihik caribou herd stopped declining and then rapidly increased at a finite rate of 1.15 during wolf control. In comparison, the Wolf Lake herd remained stable and the Chisana herd declined, but the Ibex herd increased at rate similar to Aishihik (i.e. had similar responses without wolf control).	NRC 1997; Hayes et al. 2003; Farnell 2009; Hegel and Russel 2013
Wells Gray (southern mountain), BC	1987 - 1988	wolves	4 in 1987 (on east side of lake, few or no wolves remaining on east side; i.e. nearly 100% removed), 7 in 1988 (4 east side, 3 west side, 30 - 50% of population in the area killed) Estimated density ~ 1/100km²	Shot (from ground?)	Unknown; some evidence of recolonization	After reductions in 1987, calf survival was higher on the east side of the lake (wolves reduced) vs. west side. After reductions in 1988, some wolf predation still occurred; calf survival to Oct marginally improved, but few calves remaining by March. Over all years, when both natural wolf absence and control were included in analysis, calf survival to Oct and estimated March recruitment was significantly higher in year where wolves were absent. When wolf control	The Quesnel Lake population declined from 220 caribou in 1986 to 94 in 1989 (finite rate of increase = 0.754). The Wells Gray population increased from 231 in 1987 to 265 in 1989 (finite rate of increase = 1.04).	Seip 1992

						alone was assessed, calf survival to Oct was significantly higher in areas where wolves were controlled as compared to areas where wolves were present and uncontrolled; however, by March, there was no difference in recruitment between controlled and uncontrolled areas.		
Wells Gray North subpopulation and Bakerville herd, together termed the "Quesnel Highland" caribou (southern mountain), BC	1st phase: 2001 - 2004 2nd phase: 2007 - 2012	wolves and moose	Phase 1 (2003/2004): 5 - 9 wolf packs fertility treated, mean pack size decreased ~8 to 4.5 wolves and wolves reduced by 13% March 2003, 27% 2004 Phase 2 (2008/2009): 9 - 13 wolf packs fertility treated (sterilized 3 wolves within each group) March wolf density reduced by 36-48% after 2009 + Moose harvests increased after 2001 (but no comparison to pre-treatment)	Combination of fertility treatment and lethal methods (aerial capture then killed, no direct shooting) + moose hunting harvests increased.	After program halted in 2004, wolves increased to 9.2 wolves/1000km^2 (similar to original unexploited density in 2001) by Dec 2007. * Sterilization results: No evidence that treatment during Phase 1 (5-9 packs fertility treated) reduced the wolf rate of increase. Phase 2: 9 - 13 packs fertility treated by 2009; treatment successfully arrested reproduction, stabilized wolf population at a substantially lower density since 2009.	There was no observed change in recruitment with reduced wolf densities.	The number of caribou in the treatment herd showed an overall increase since 2002, but the comparison herds also increased until 2006. From 2006-2012, the treatment herd was the only group that increased, but there no evidence that the trend was significantly different from comparison herds.	CCLUP Caribou Strategy Committee 2009; Hayes 2013
Little Smoky (boreal), AB	2005/06 - 2012	wolves	~45% of mid-winter pop'n each year = average removal of 11.6 wolves/1000km^2 = 841 wolves total	Aerial and toxicant methods (+ minimal removal by fur trappers)	Relatively consistent removal rates over all years of the program suggest the wolf population maintained high numbers despite reductions each year.	Mean recruitment within LSM significantly increased over time. Mean recruitment in the RPC reference herd (0.19 and 0.17 pre- and post-control respectively). Experimentally, recruitment was not significantly different between populations or between before and after treatment periods (0.12 pre-control, 0.19 post-control). Mean adult female survival within the LSM herd did not significantly increase over time. Mean adult survival in the RPC herd was low (0.83 and 0.79 pre- and post-control respectively). Experimentally, adult female survival was significantly different between the LSM and RPC herds, but not between before and after treatment periods (0.89 pre-control, 0.91 post-control), nor was there an interaction between treatment and population.	The empirical/stochastic λ for LSM increased from 0.95/0.94 prior to control to 0.99/0.99 following control (4.6% increase in mean population growth). But increase began just prior to wolf control, with the largest λ (1.1) recorded the year before wolf control began. The RPC (control population) experienced a reduction in population growth (λ = 0.908/0.90 to 0.861 /0.86; 4.7% decline). The BACI design showed that LSM and RPC population trajectories were not significantly different prior to wolf control, but significantly diverged following treatment. Projections indicated wolf control generated a 20% difference between realized and projected population size.	Hervieux et al. 2014

Fortymile (Grant's), AK	1997 - 2001	Wolves	Treated 15 packs; 8 sterilized packs remained as of April 2003	Surgical sterilization of the dominant pair and translocation of other wolves	Unknown * Sterilization results: Sterilizations was successful in stopping reproduction. Sterilization did not affect the probability of dispersal, but territory size of each pack was reduced.	Calf survival was >= 50% during the period of wolf reduction from 1998/99 to 2001/02. In comparison, when herd size was stable, calf survival was 33 and 41% in 1994 and 1995. But no evidence to suggest wolf predation had decreased following treatment (maybe b/c caribou moved out of the treatment area and wolves from outside the treatment area hunted within the area). Recruitment in 1998 - 2002 was improved compared to pre-treatment. Adult survival rates exceeded 87% during the period of wolf reduction from 1998/99 to 2001/02. In comparison, when herd size was stable, adult survival was 75 and 80% in 1990 and 1991.	The Fortymile herd nearly doubled in size from 22,558 caribou in June 1995 to 43,375 caribou in June 2003, but this increase began prior to wolf treatment.	Boertje and Gardner 2003
	2004 – 2008 (ongoing as of 2010)	Wolves	Pre-control harvest (2001 – 2004): 47 wolves removed per year 2001 – 2004; Targeted wolf control (2004 – 2008): average 107 wolves removed per year 2004 – 2008 (+ opened regulations on Grizzlies)	Harvest and trapping only 2001 – 2004; Harvest, trapping, ground shooting, snaring 2004 – 2008	Wolf population increased from pre-control to 2008.	Caribou recruitment avg. 35 calves:100 females for 2 control years vs. avg. 27 calves: 100 females 5 years pre-control.	The caribou herd estimated: 43,375 in 2003, 38, 364 in 2007 and 41,000 in 2008 (i.e. stable).	Russell 2010
Columbia North, Columbia South, Frisby-Bolder/Queest (southern mountain), BC	2003 - 2005	moose	10-fold increase in hunter harvest 2003 – 2005, reduced harvest 2005 - 2010; moose pop'n declined 71% from 2003 - 2011 (1.58/km^2 to 0.44/km^2)(decline started 1-2 years after treatment began)	Moose harvest	Moose pop'n declined by 71% between 2003 and 2011, beginning 1-2 years after the start of harvest increases in 2003. Likely triggered by hunting, driven by depensatory predation by wolves. Declines in moose numbers appeared to also reduce the wolf population, likely due to dispersal (but maybe also starvation).	Recruitment was not significantly different before vs. after treatment in either the treatment or reference areas. Wolf reductions may not proportionately decrease predation risk to caribou. Remaining wolves spent more time in caribou habitat. But no evidence that caribou increased in diet (based on scat and kill-sites).	Caribou populations within the treatment area had mixed responses to moose reductions; the larger subpopulation, Columbia North, increased following moose reductions, while the smaller subpopulations, Columbia South and Frisby-Queest, continued to decline. Both subpopulations within the reference area showed continued declines over the long term.	Serrouya 2013, 2015
Hart Ranges (southern mountain), BC	2006 - Ongoing?	moose	2005 population: ~3000 moose (1.18 moose/km^2) 2008 - 2009: ~1818 moose (0.73 moose/km^2) (50 - 60% reduction)	Moose harvest	2009 max dispersal in treatment area was sig. diff. from 2007 max dispersal in treatment area and 2009 dispersal in control area (non-overlapping confidence intervals). Possible lag time 2 – 3 years. Trends indicated increasing dispersal over time	n/a	n/a	Steenweg 2011

					in treatment area (not statistically sig.). No evidence for change in mortality.			
South Selkirks (southern mountain), BC	Jan 2015 - Ongoing	wolves	11 (Of the wolves targeted, seven to 10 remain)	Shooting from helicopters	Unknown	Unknown	Unknown	"U.S.-ranging Selkirk …" 2015; "Wolf cull…" 2015; Meissner 2015
Quintette, Moberly/Klinse-za, Scott and Kennedy-Siding (central mountain), BC	Jan 2015 - Ongoing	wolves	73 (most around the Moberly and Quintette caribou herds)	Shooting from helicopters	Unknown	Unknown	Unknown	"Wolf cull…" 2015; "B.C. wolf cull…." 2015

Appendix 1 Table References

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Appendix 2: Translocations

Note: Includes wild-to-wild translocations, as well as those that involved captive-breeding or rearing

Target Location/Herd	Source Location/Herd	Source Subspecies/DU	Year(s)	Type of Translocation	# released	Major Outcomes	Present or Extinct	References
North American Translocations since 1960, herding excluded								
Purcells-South caribou herd (mountain ecotype), Purcell Mountains, BC	Level-Kawdy herd, BC	Northern mountain DU (woodland)	March 2012	Reinforcement (~20 resident caribou)	19	10 died within the first 4 months after release. As of July 2013, 17 of the 19 translocated caribou were confirmed dead as a result of cougar predation (6), wolf predation (2), accidents (3), unknown causes other than predation (3), unknown causes which may include predation (2) and malnutrition (1). 1 bull confirmed alive, 1 cow status unknown. 2014 estimate: 19 caribou in herd, population stable, but expected to decrease over long-term. Second phase deferred to 2015.	Present	Gordon 2013; Environment Canada 2014; Leech 2015
Telkwa caribou herd (northern ecotype), Telkwa Mountains, BC	Chase/Sustut Herd, BC	Northern mountain DU (woodland)	1997 - 1999	Reinforcement (6-8 resident caribou)	32 total (28 F, 4 M; 1997: 12, 1998: 16, 1999: 4)	Translocated caribou remained in the target area and herd increased to a peak of 114 caribou in 2006; Stronen et al. (2007) suggested translocation successful over short-term; then declined to approximately 40 animals by 2010; estimated at ~25 animals Oct 2013, declining population trend.	Present	Houwers 2006 and G. Schultze pers. comm. in Kinley 2010; Stronen et al. 2007; Cichowski 2014; Environment Canada 2014
South Selkirks herd (mountain ecotype), South Selkirk Mountains, Idaho, Washington, BC	Northern type from Itcha and Ilgachuz Mountains, BC; Mountain type from Columbia Mountains, BC	Northern and Southern Mountain DU (woodland)	1987 - 1990, 1996 - 1998	Reinforcement (~25 resident caribou)	103 total (1987: 12 northern and 12 mountain, 10 F/2M each; 1988: 14 northern and 10 mountain, each with 4 bulls; 1990: 12 mountain; 1996 - 1998: 43 mountain)	Survival over the first 3 years (April 1987 – March 1990): 67% for mountain, 33% northern caribou (33%) ($p = 0.026$). 5-year weighted average (March 1987 – Feb. 1992) annual survival rates: 74% mountain vs. 73% northern ($p = 0.97$). Northern caribou may have been more at risk of malnourishment b/c of attempts to feed on terrestrial lichens rather than the more abundant arboreal lichens (observations). Translocated mountain caribou (i.e. the same as the target herd) generally showed more similar movement patterns and had more interaction with the resident vs. northern ecotype that showed higher variability in habitat use and movement patterns. But all 7 dispersals from the release area were by mountain caribou. Translocated caribou that moved out of the release area established a new subpopulation south of the native residents; both subpopulations < 50 individuals in the early 1990s. Recruitment in 1991 and 1992 estimated 0.14 and 0.06, respectively. As of 2000, 26 of the 43 caribou transplanted in 1996 - 1998 had died (4 cougar predation, 1 grizzly bear predation, 1 accidental fall, 2 poaching, 18 unknown causes). Temporary increase in population following translocations, but declined to ~33 in 2004, ~ 27 caribou in March 2013.~18 animals in 2014.	Present	Compton et al. 1995; Warren et al. 1996; Almack 2000; USFWS 2008; DeGroot and Wakkinen 2013; "U.S.-ranging Selkirk …" 2015; "Wolf cull…" 2015

Charlotte Alplands, BC	Itcha Ilgachuz Herd and Rainbow herd, BC	Northern mountain DU (woodland)	1984, 1986, 1987, 1988, 1991	Reintroduction (1950s/60s)	[52 total] (1984: 13 adults, 2 calves to McClinchy Cr.; 1986: 8 adults, 1 yearling to McC. Cr. + 2 yearlings, 4 calves to Trumpeter Mtn.; 1987: 11 calves to Trp. Mtn.; 1988: 1 yearling, 2 calves to Trp. Mtn.; 1991: 8 calves to Trp. Mtn.)	Most animals released 1984 - 1986 (mainly adults) emigrated from the release area and returned to their original range. The 28 caribou released between 1986 and 1991 (mainly calves) remained in the release area. => Adult caribou may be more likely to disperse away from the target area than calves when caribou are moved relatively short-distances (<100 km) due to efforts to return back to their original range. Herd likely mixes with both its source herds. 1989 - 1993: ~72 animals (~11% growth); decline to 23 individuals in 2001, ~ 7 individuals as of 2012, expected continued decline (wolf or grizzly predation suspected cause)	Present	Young et al. 2001; CCLUP 2009; Kinley 2010; Environment Canada 2014
Baxter State Park, Maine	Newfoundland	Newfoundland DU (woodland)	December 1963, May 1989 & April 1990	Reintroduction (1908)	56 total [1963: 24 (19 F, 5 M), 1989/1990: 32]	1963 release: Animals dispersed; failure. 1986: 27 caribou captured on Avalon Peninsula, Newfoundland. 22 survived, held in a ~6 ha enclosure for ~3.5 years. 1989: 12 captive-raised caribou were released into the park; after realizing that penned caribou experience a higher risk of meningeal worm infection, the remaining 20 wild and captive-raised caribou were released in 1990. Caribou reported to have become 'skittish around people' once released into the wild despite having been relatively tame in captivity, which researchers interpreted as a positive sign that captive-raised calves kept some natural instincts. One caribou also seen successfully evading two coyotes. After initial releases: 14 caribou were observed in 1964, but then dispersed and disappeared entirely after 1966. Only 1 of the 12 animals released in 1989 was confirmed alive by the end of the year. By November 1990, 25 were confirmed dead (12 killed by bears or coyotes), status of 7 unknown. Meningeal worm and black bear predation likely caused decline.	Extinct	Bergerud and Mercer 1989; Dunn 1965 in Audet and Allen 1996, Gold 1989, "Predators Kill…" 1990, McCollough and Connery 1991 in Audet and Allen 1996
Gargantua Peninsula + 2 small offshore islands, ON	Slate Islands, ON	Boreal DU (woodland)	October 1989	Reintroduction	39 (10 M, 26 F, 3 calves)	High initial mortality. By June of the following year, only 1 of 17 radiocollared animals remained alive (wolf predation). Some caribou escaped to surrounding islands, still exist at low densities in the area.	Present	Gogan and Cochrane 1994; Bergerud and Mercer 1989, Bergerud et al. 2007, OWCRT 2008, (and Kinley 2010)
Nushagak Peninsula, Alaska	North Alaska Peninsula herd, Alaska	Grant's/barren-ground	Feb 1988	Reintroduction (>100 years)	146 (Composition: 82.2% females, 9.6% males and 8.2% calves)	Rapid growth in 1st 6 years (1988 - 1994) to >1,000 animals in 1994 ; peaked at 1,399 caribou in 1997; decline to 526 caribou in 2006; Stayed at ~550 caribou until 2009; then increased to 902 by July 2012. Over entire period, 1988 - 2013, r = 0.226. Annual female survival rate 1988 - 2013: 0.876 with hunting mortality included, 0.915 with hunting excluded. Hunting and predation accounted for 31.8% and 11.4% of all mortalities, respectively. Nutrition is thought to be the ultimate limiting factor.	Present	Collins et al. 2003; Aderman 2013
Kenai Peninsula , Alaska	Nelchina herd, Alaska	Grant's/barren-ground	1965, 1966, 1985, 1986	Reintroduction (1912)	124 total [1965: 15, 1966: 29, to 2 diff. sites; 1985 & 1986:	Releases in 1965/66 resulted in the formation of 2 caribou herds, the Kenai Mountain Herd and the Kenai Lowland Herd. 1985/86 releases formed 3 new herds: Twin Lakes Herd, Killey River Herd and Fox River	Present	Burris and McKnight 1973 in Audet and Allen 1996; Tunseth 2002; Alaska

					80 total (28, 18, 16, 18) to 4 diff. sites]	Herd. In 2002, the Twin Lakes and Killey River Herds grew and merged (~700 animals), but 3 avalanches in 2001 – 2003 killed >150 caribou. Today: Kenai Mountain herd: 200 - 400 animals; Kenai Lowland herd: 130 - 150; Twin Lakes/Killey River herd: ~250; Fox River herd: 50 - 75		Department of Fish and Game 2003; Paul 2009; USFWS 2014a
Leach Island, ON	Slate Islands, ON	Boreal DU (woodland)	1986	Reintroduction	3 (1 M, 1 F, 1 calf)	Only four caribou, possibly all female, remained by 1990.	Presumed extinct	Gogan and Cochrane 1994
Bowman Island, ON	Slate Islands, ON	Boreal DU (woodland)	October 1985	Reintroduction (1940s)	6	By April 1986, all but one of the caribou had died (predation, emigration).	Extinct	Bergerud 1985, Bergerud and Mercer 1989; Bergerud et al. 2007; OWCRT 2008
Montreal Island, ON	Slate Islands, ON	Boreal DU (woodland)	1984	Introduction	9	1988: 14 caribou, 1993: 16 caribou seen, but wolves reached island in 1994: predated some, others moved off island.	Extinct	Gogan and Cochrane 1994; Bergerud et al. 2007; OWCRT 2008; Kinley 2010
Michipicoten Island, ON	Slate Islands, ON	Boreal DU (woodland)	1982 & 1983	Reintroduction (1800s)	8 total (1 M, 3 F, 3 calves in 1982, 1 M in 1983)	In 1988, after 6 calving seasons, at least 26 caribou present (finite rate of increase of $\lambda = 1.22$); 2001: 160 animals (finite rate of increase over 19 years = 1.18); 2003: > 200 caribou.	Present	Gogan and Cochrane 1994; Bergerud and Mercer 1989; Bergerud et al. 2007; Ontario Parks 2003
Newfoundland	Newfoundland (native herds)	Newfoundland DU (woodland)	1961 - 1982	Introductions	384 total (22 different sites, 4 - 33 depending upon the site)	By 1982, herds numbered ~1,500 animals; 17 of the 22 sites maintained viable populations. Evidence suggests caribou released nearby (< 50 km) existing herds may leave the release area to join resident caribou.	Presumed those deemed successful are still present.	Bergerud and Mercer 1989
Belcher Islands, Nunavut	Reindeer Reserve, Tuktoyaktuk, NWT	Reindeer	March 1978	Introduction of reindeer, reintroduction of *Rangifer* (disappeared in 1800s)	60 (10 M, 50 F)	March 1982: 222 animals (3.7 times increase in the population since reintroduction). Estimated population over last 20 years ~ 700 animals, following an increasing population trend.	Present	Ferguson 1985
Le Parc des Grands-Jardins/ Laurentides Wildlife Reserve (Charlevoix), QC	Cote-Nord, QC	Boreal DU (woodland)	1969 - 1972	Reintroduction (1920s)	83 total (over 3 years, all calves)	48 caribou (13 in 1966, 35 in 1967) captured in the Cote-Nord region, transported to a 0.5 ha enclosure in Grands-Jardins National Park initially, later to a 2.1 ha enclosure in Laurentides Wildlife Reserve. 7 of the 48 caribou died from myopathy soon after release. Remaining caribou held in captivity for 3 years, adapted well and successfully bred to grow the population to 102 animals (adults and calves) by Summer 1969. 83 caribou born in captivity released into wild. Original 48 caribou never released (worried they would return to capture site). Herd remained stable at ~40 - 50 individuals until ~1980 (during wolf control); increased in the 1980s; stabilized at 100 – 125 individuals in the 1990s; declined in 2000s; stabilized at ~83 animals (2008); still highly threatened (small size, isolation from other herds, low recruitment). Bergerud and Mercer (1989) highlighted the Charlevoix reintroduction as the only example of a successful translocation of caribou into an area frequented by predators, but St-Laurent and	Present	Karns 1978 in Kinley 2010; Vandal 1984 in Bergerud and Mercer 1989 and Kinley 2010; McCollough and Connery 1990 in Kinley 2010; Sebbane et al. 2003, 2008, 2011; St-Laurent and Dussault, 2012; Sepaq 2015

						Dussault (2012) warn against such a conclusion given the continuing fragility of the herd.		
Cape Breton Highlands National Park, Nova Scotia	QC	woodland caribou (eastern migratory or boreal DU)	1968 & 1969	Reintroduction (1920s)	51	Seen in the park for one year, but then quickly declined and were last seen in 1972 (meningeal worm suspected).	Extinct	Dauphine 1975 in Bergerud and Mercer 1989 and Audet and Allen 1996
Southampton Island, Nunavut	Coats Island, Nunavut	Barren-ground	1968	Reintroduction (1955)	48 (19 females, 7 yearling females, 2 female calves, 6 bulls, 6 yearling males, 8 male calves)	Rapid growth, >30,000 individuals by 1997. Population decline starting in 2003, estimated 7,761 in 2011 (reproductive disease, reduced pregnancy rates and harsh icing events).	Present	NWMB 2011; "Southampton Island…" 2012; "Battle brews.." 2014
Other North American translocations:								
Anchorage area, Alaska	Hagemeister Island, Alaska	Reindeer	1992 & 1993	Management-focused	411	Presumed present	Present	Stimmelmayr and Renecker 1998
Great Cloche Island , ON	Norway	Reindeer	January 1969 (transported); released May 1969	Introduction	12 (9 F, 3 calves)	First signs of neurologic disease August 1969 (~3 months after released into enclosure); By July 1970, only 5 alive and showed signs of disease.	Extinct	Anderson 1971, Bergerud and Mercer 1989
Wisconsin	Unknown	woodland caribou	Unknown (5-year period, 1960s/70s?)	Reintroduction	14	Caribou released to enclosure in June. By September, half of the caribou were infected with P. tenuis and died; by January all caribou dead.	Extinct	Trainer 1973 (also cited in Bergerud and Mercer 1989 and Audet and Allen 1996)
Hagemeister Island, Alaska	Unknown	Reindeer	1965	Herding	unknown	Managed by native reindeer herders in Togiak since introduction. Herd grew to unsustainable levels and had to be removed (95% reduction in lichen, 1990 mass starvation).	Present	Stimmelmayr and Renecker 1998, "Alaska Journal;…" 1992
Adak Island, Alaska	Nelchina herd, Alaska	Grant's/barren-ground	July 1958 & July 1959	Introduction	24 (calves)	Calves released in 1958 bred as yearlings and produced the first wild-born calves in 1960. Population rapidly expanded to 2,600 – 2,800 in June 2012, started to emigrate to surrounding islands; USFWS can shoot when off Adak.	Present	Jones 1966, Burris & McKnight 1973 in Audet and Allen 1996; Paul 2009; USFWS 2014b; Rosenthal 2015; Joling 2015
St. Matthew Island, Alaska	Nunivak Island, Alaska	Reindeer	August 1944	Introduction	29 (yearlings: 24 f, 5 m)	Population increased rapidly to 6,000 animals by Summer 1963, then crash die-off (depleted lichen, severe weather). Extinct by 1982.	Extinct	Klein 1987; Klein 1968 in Audet and Allen 1996
Liscombe Game Sanctuary, Nova Scotia	Newfoundland	Newfoundland DU (woodland)	April 1939	Reintroduction	12 total (9 F, 3 M)	Introduction failed.	Extinct	Tufts 1939 in Bergerud and Mercer 1989; Tufts 1939 and Bensen and Dodds 1977 in Audet and Allen 1996
Red Lake Herd, Minnesota	SK	Assumed boreal (could be barren-ground)	1938 - 1940	Reinforcement	15 - 20	10 caribou (2 bulls, 8 calves) captured in Saskatchewan in 1938; 1 bull released immediately; 8 surviving animals held within a 1,000 ha enclosure in Minnesota until being released with their offspring (1940). By 1946, all caribou in the area had died (predation, poaching, meningeal worm implicated).	Extinct	Karns 1978 in Audet and Allen 1996; Cringan 1957 in Luensmann 2007; Bergerud and Mercer 1989

Mackenzie River Delta, NWT	Alaska	Reindeer	1929 - 1935	Herding	2,382 (1,498 females, 611 bucks, 273 steers)	Numbers increased to 8,346 by 1942; then fluctuated between 5,000 - 9,000; 2,800 in 1967; ~3,000 animals managed (frequently escape to join native barren-ground caribou)	Present	Scotter 1972; Treude 1979; Haigh 1991; "NWT's only reindeer herd..." 2008; "Mackenzie Delta..." 2015
Baffin Island, Nunavut	Norway	Reindeer	1921	Herding	627	Due to mortalities, dispersal, and inadequate forage and care, most of the herd disappeared by 1925 and the herding project was cancelled in 1927.	Extinct	Scotter 1972; Haigh 1991; Government of Nunavut 2013
Pribilof Islands (St. Paul and St. George Islands), Alaska		Reindeer	1911 + later releases from Nunivak and Umnak Islands	Herding	25	Populations severely declined (poaching, severe weather, inadequate forage), but later releases boosted populations. Hundreds or reindeer present as of 2007.	Present	Hanna 1922; NOAA 2008
St. Anthony, Newfoundland	Norway	Reindeer	1908	Herding	300	Rapid increase to 1,300 animals in 1912, but then decreased to 230 by ~1920 (poor management); eventually moved to Anticosti Island	Extinct	Scotter 1972; Haigh 1991
Alaska (unknown location)	Siberia	Reindeer	1891-1902	Herding	1,280	Estimated 10,000 reindeer in Alaska by 1905 (managed). Population declined in 1930s to 25,000-50,000 by 1950s (poor management). Reindeer herding now found only on Seward Peninsula (~15,000 - 20,000 caribou, 20 herders).	Present	Scotter 1972; Naylor et al. 1980; Haigh 1991; Backi 2004; Willis 2006; Finstad et al. 2002, 2006
International translocations:								
Falklands Islands	South Georgia	Reindeer	2001	Introduction	59 (calves)	Produced offspring in 2003; further results unknown	Present	Bell and Dieterich 2010
Godthaab area, West Greenland	Norway	Reindeer	1952	Herding	unknown	Unknown	Unknown	Klein 1980
Broggerhalvoya, Svalbard	Unknown	Reindeer	1978	Reintroduction (~100 years)	15	Increased to ~200 by 1989	Presumed present	Staaland et al. 1993
Salamajarri National Park, Finland	Unknown	wild forest reindeer	Dec 1981 and 1983	Unknown	Unknown (calves)	Wilf forest reindeer captured winters 1981 & 1983. Held in captivity until ~1.5 years old, then released. Released individuals successfully bred in the wild, but other outcomes are not available.	Unknown	Nieminen and Laitinen in Audet and Allen 1996
Western Greenland	Northern Norway	Domestic reindeer	September 1952	Unknown	225	Reindeer brought the warble fly and nostril fly. Infected the indigenous wild Greenland caribou. Greenland caribou reduced in number.	Unknown	Rosen 1955 in Olney, Mace and Feistner 1994
South Georgia	Norway	Reindeer	1911 - 1925 (3 occasions)	Introduction	Unknown	Resulted in 2 herds that today number ~ 2,6000 animals and have become a nuisance	Present	Leader-Williams 1988 in Bell 2010

<u>Appendix 2 Table References</u>

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Appendix 3: Captive-rearing (Maternal Penning)

Targeted Herd	Years	Pen Size	# Animals captured	Births	Deaths in captivity	# Adult Females & Calves released	Calf Survival	Recruitment	Population responses	References
Chisana northern mountain caribou herd, YT	2003 - 2006	2003: 6.0 ha 2004: 9.5 ha 2005: 12.0 ha 2006: 12.2 ha	146 pregnant females total (2003: 17; 2004: 29; 2005: 50; 2006: 50)	146 total (All pregnant females gave birth, 2 stillborn)	Adults: 6 total, but 5 resulted from capture for stocking pen. Calves: 10; 8 were suspected to be of natural losses; 2 were study-related; Avg. survival in pen 93% (wild-born 33%)	140 females, 136 calves total	Averages 2003 – 2006: During penning (birth/6 days - ~1.5 months): 93% pen-born, 33% wild-born (156 monitored). After release (6 days/~1.5 months to 5 months): 70% pen-born, 52% wild-born. Birth to 5 months: 65% pen, 17% wild	Pre-penning (1989 – 2002): 6.8 calves:100 females Post-penning (2003 - 2006): 22.1 calves:100 females with penned animals, 18.2 calves:100 females without penned calves. I.e. mostly natural increase; maternal penning contributed avg. 26%	Pop'n at < 720 animals in 2003 (pre-penning), 766 in 2007, 682 in 2010. Modeling suggests penning augmented the herd size by ~11% over projected growth without penning between 2003 - 2006. Note: The scale of project too small in proportion to the wild herd's size to generate growth, but may be more effective for smaller pop'ns.	Farnell 2009; Chisana caribou herd working group 2012; Oakley et al. 2004; CCRT 2010; Kinley 2010; Hegel and Russel 2013; L. Adams pers. comm. 2016
Little Smoky boreal caribou herd, AB	2006 - 2006	4 ha	10 pregnant females	10 total (All pregnant females gave birth)	Adults: 0 Calves: 1 (myocardial degeneration)	10 females, 9 calves	Birth to late-Sept (~3 months): Pen-born: 50% pen-born, 71% wild-born (7 monitored) March 2007 (calves ~9 months): at least 3/5 remaining captive-reared and 3/5 free-ranging calves alive.	Percentage of calves in the population in September 2006 (19%) was the highest observed since 1982; proportion remained high through March 2007 (14.5%).	Pop'n ~ 80 animals (2012 Recovery Strategy); Stochastic population growth rate of the Little Smoky herd avg. 0.939 (1999 – 2005) vs. 0.988 (2005 – 2012). *Note: Concurrent wolf control program 2005/06 - 2012	Smith & Pittaway 2011; Hervieux et al. 2014
Columbia North southern mountain caribou herd, BC	2014 - Ongoing	2014: 6.4 ha 2015: 6.4 ha	2014: 12 total (9 pregnant females, 1 non-pregnant cow, 2 10-month old calves) 2015: 19 total (18 female caribou [16 pregnant], 1 10-month old calf)	2014: 9 total (All pregnant females gave birth) 2015: 15 calves born	2014: 0 2015: Adults: 1; Calves: 4 (abandonment, injury, infection) 2014 + 2015 combined: average penned adult survival rate = 96%	2014: 12 females, 9 calves 2015: 17 females, 11 calves	2014 calves: Survival to March 2015 (~9 months): 22.2% (2/9) pen-born, 19.7% wild-born 2015 calves: Survival to 4 months: 67% pen-born (10/15) (higher than expected for wild-born counterparts)	March 2015: Calves composed 11.5% of population (comparable to late 1990s rates of 8.9 - 14.3%, but lower than early 1990s rate ~ 19%, when the herd was increasing)	Pop'n 149 – 152 animals in 2013. Note: Concurrent moose reduction experiment	Alexander 2014; RCRW 2015a-e; Orlando 2015; Clayton 2015; Serrouya et al. 2015
Kinse-Za/Moberly central mountain caribou herd, BC	2014 - Ongoing	2014: 3.9 ha 2015: 7 ha	2014: 10 pregnant females 2015: 11 females (8 thought to be pregnant)	2014: 10 total (All pregnant females gave birth) 2015: 7 calves born	2014: 1 calf (entanglement in blowdown) 2015: 2 calves	2014: 10 females, 9 calves 2015: 11 females, 15 calves	2014: As of August 12, 2014, 9 females and 6 calves remained alive.		2014 census: 22 Moberly, 18 Scott (40 total) 2015 census: 34 Moberly, 8 Scott (42 total) *Note: Concurrent wolf control program initiated 2015	Klinse-Za maternal penning steering committee, 2014a,b; Seip and Jones 2013; PNCC, 2015; McNay et al. 2013; Carter 2015; Seip 2015

Appendix 3 Table References

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Appendix 4: Lessons from Other Ungulate Recoveries

We drew upon knowledge of previous ungulate translocations detailed in case studies within the IUCN Reintroduction Specialist Group's Global Reintroduction Perspectives books (Soorae 2008, 2010, 2011 and 2013). These projects included releases of:

- Arabian sand gazelle (*Gazella subgutturosa* ssp. *Marica*) to Uruq Bani Ma'arid PA and Mahazat as-Sayd PA, Saudi Arabia
- Arabian oryx (*Oryx* leucoryx) to Al Wusta Wildlife Reserve (formerly Arabian Oryx Sanctuary), Oman; Um Al Zomoul and Dubai Desert Conservation Reserve, United Arab Emirates (UAE); Wadi Rum PA, Jordan; Negev Desert, Israel; Mahazat as-Sayd PA and Uruq Bani Ma'arid PA, Saudi Arabia
- Hirola (*Beatragus hunteri*) to Tsavo East National Park, Kenya (reinforcement)
- Roe deer (*Capreolus capreolus*) to Ajloun Nature Reserve, Jordan
- Amur goral (*Naemorhedus caudatus*) to Wolaksan National Park, South Korea
- Apennine chamois (*Rupicapra pyrenaica ornata*) to Gran Sasso-Laga National Park, Italy
- Mountain gazelle (*Gazella gazella*) to Ibex Reserve and Uruq Bani Ma'arid PA, Saudi Arabia
- Scimitar-horned oryx (*Oryx dammah*) to Dghoumes National Park, Tunisia
- Przewalski's horse (*Equus ferus przewalskii*) to Mongolian Gobi
- Common eland (*Tragelaphus oryx*), wildebeest (*Connochaetes taurinus*), giraffe (*Giraffa camelopardalis*), sable antelope (*Hippotragus niger*), bushbuck (*Tragelaphus scriptus*) and Burchell's zebra (*Equus quagga*) to Shangani Ranch and De Beers Ranch, Zimbabwe
- Milu (Pere David's deer) (*Elaphurus davidianus*) to Beijing, Dafend and Shishou, China
- Lichtenstein's hartebeest (*Alcelaphus buselaphus* ssp. *Lichtensteinii*) to Malilangwe Wildlife Reserve, Zimbabwe
- Tule elk (*Cervus elaphus nannodes*) to Point Reyes National Seashore, USA
- Elk (*Cervus elaphus*) to Ontario
- Bison (*Bison bison*) to New Mexico and Native American lands in Western USA
- Wisent (European bison) (*Bison bonasus* L.) to Europe
- Wood bison (*Bison athabasckae*) to Russia
- Desert bighorn sheep (*Ovis canadensis mexicana*) to Fra Cristobal Mountains, New Mexico, USA

Many of the above projects also included long-term captive-breeding programs and/or the use of fences in recovery efforts.

As ungulates translocated within North America to areas with predators, we examined Elk (*Cervus elaphus*) reintroductions and reinforcements to Eastern North America, and Bighorn sheep (*Ovis canadensis*) reintroductions to Western U.S. in more detail. We also investigated Alpine ibex (*Capra ibex*) reintroduction and reinforcement to the Marmolada massif in Italy, Persian fallow deer (*Dama mesopotamica*) reintroductions and introductions to Iran and Isreal and a predator-exclosure project for hirola (*Beatragus hunteri*), which were not included within the IUCN case studies. Furthermore, we drew upon relevant reviews of ungulate-specific and general translocations for overarching observations.

Though not a comprehensive review, we consider these species and associated information to adequately represent an array of different situations upon which to draw general conclusions—e.g. taxa, geographical locations, habitats, wild-born vs. captive-born individuals, founder group size and composition, captive-breeding, and fencing. We drew upon experts' judgments of overall program success, challenges faced and major lessons learned, as detailed in the IUCN case studies, to relay information deemed most relevant to caribou recovery.

a) How 'successful' have ungulate translocations been overall?

All ungulate translocation programs detailed with IUCN case studies were deemed either partially successful, successful or highly successful (i.e. none were considered failures) at the time of case studies' respective publication dates. Though this may in part represent a tendency to report on success rather than failure (Fanelli 2012) it may also reflect the overall success of long-term ungulate translocation programs.

IUCN case studies reported on the overall success of a program, which sometimes involved several releases to a given location and/or releases to multiple locations. For example, a first release of Arabian oryx to the Dubai Desert Conservation Reserve in UAE was largely unsuccessful; however, later releases were successful in establishing self-sustaining herds (Simkins 2008). Similarly, the first release of Apennine chamois to the Gran Sasso-Laga National Park in Italy was unsuccessful (only 1 animal survived), but subsequent releases established stable, viable herds (Lovari et al. 2010).

For those examples examined that were not included within IUCN case studies:

- A review of elk reintroductions to Eastern North America found that about 40% of documented reintroductions failed within 5 – 94 years (most occurred in the first half of the 20th century) (Popp et al. 2014).
- Of 100 bighorn sheep reintroductions within Colorado, Montana, North Dakota, South Dakota, Wyoming and Utah between 1923 and 1997, 30 were considered unsuccessful (13 extirpated and 17 remnant populations, ie. < 30 animals as of 1997 with a low chance of recovery), 29 moderately successful (30 – 99 animals) and 41 successful (≥ 100 animals) (Singer et al. 2000).
- Of 10 reintroductions and 3 introductions of Persian fallow deer to 13 sites in Iran, translocation was considered to be a success for 4 sites (ie. 33% success rate) (Goudarzi et al. 2015).

Authors of nearly every IUCN case study emphasized the importance of appropriate planning prior to any translocation, as well as post-release monitoring and adaptive management following release. The need for proper documentation throughout the translocation process and the importance of long-term financial commitment and support was also often relayed.

Hoffman et al. (2015) recently reviewed all 235 even- and odd-toed ungulates (Cetartiodayctyla and Perissodactyla, respectively) listed as 'data-sufficient' on the 2008 IUCN Red List and compared their 2008 reported conservation status to their estimated status under scenarios wherein no conservation efforts were conducted. Overall, the authors estimated that **the decline in the conservation status of reviewed ungulates would have been 8 times worse had conservation actions not been implemented** (based on a 'best case' scenario) (Hoffman et al. 2015). Of 178 species that were affected by direct threats (mostly hunting), 30 benefited from direct targeted conservation action, such as translocations, anti-poaching patrols and species-specific hunting restrictions (Hoffman et al. 2015). In comparison, 148 benefited from indirect conservation, such as habitat protection. Of at least 25 species that were affected by indirect threats (mostly habitat loss resulting from agriculture or logging), direct action was taken for 3 species (Hoffman et al. 2015). Species whose conservation status was projected to have been lower than its 2008 status were it not for reintroduction efforts included the Arabian oryx, Przewalski's horse, Persian fallow deer, white rhinoceros (*Ceratotherium simum*), and Iberian wild goat (*Capra pyrenaica*).

b) Have populations typically grown following translocations?

Many reintroduced populations examined within the IUCN case studies exhibited population increases (e.g. Spalton et al. 1999, Wacher and Robinson 2008, Lee et al. 2010, Lovari et al. 2010, Islam et al. 2011c, Al Jahdhami et al. 2011, Wronski et al. 2011, Swanepoel and Dunham 2013, Gogan et al. 2013), especially if environmental conditions at the release site were favorable and the population was protected from predators and/or poaching pressures.

However, the overall growth of released populations was reported over varying time periods and multiple subsequent releases often occurred. Furthermore, **initial increases were sometimes followed by declines due to a number of factors** (often in tandem), including hunting (Spalton 1993, Al Jahdhami et al. 2011, Wronski et al. 2011), weather (and associated starvation) (Islam et al. 2010a, Kaczensky et al. 2011), and domestic livestock (Wronski et al. 2011). These declines were particularly severe when populations were restricted to a defined area (usually by fencing) and severe weather put added pressure on the population (Islam et al. 2010a, Clegg et al. 2013).

Not all released populations exhibited positive growth. For example, 2 of 3 reintroduced populations of Arabian oryx to Israel were reported to have exhibited negative growth due to low reproductive success (Saltz 2008). A reintroduced population of Lichtenstein's hartebeest in Malilangwe Wildlife Reserve, Zimbabwe also declined following the second of two releases, apparently due to unsustainable predation levels (Clegg et al. 2013).

For those examples examined that were not included within IUCN case studies:

- Of the 70 'successful' and 'moderately successful' reintroductions of bighorn sheep to Western US, 11 steadily increased following initial translocation, 11 increased at first but then declined, 11 increased, declined and then recovered, and 1 fluctuated largely over time (Singer et al. 2000).
- The exponential rate of increase (*r*) of reintroduced elk populations in Eastern North America ranged from - 0.05 to 0.13 (i.e. $\lambda = 0.95 – 1.13$) (Popp et al. 2014).
- Two of 13 populations of Persian fallow deer reintroduced or introduced to Iran have gone extinct (Goudrazi et al. 2015). Two extant populations are declining (growth rate, *r*, -0.11 and -0.26) and growth rates for the remaining 9 extant populations range between 0.06 and 0.32 (≥0.2 considered

good, 0.1 – 0.2 moderate and ≤ 0.1 poor, negative value indicates declining population) (Goudrazi et al. 2015). Between 1996 and 2001 a total of 124 Persian fallow deer (58 females, 66 males) were released to Nahal Kziv Nature Reserve in Israel over 10 events (Bar-David et al. 2005). The wild population continued to expand to more than 250 animals by 2014 (D. Saltz pers. comm. 2014 in IUCN 2015).

c) *What time of year were animals typically released?*

The timing of releases was determined according to the species' biology, environmental conditions and the most important limiting factors to populations (Stanley-Price 1986, Kiwan et al. 2008, Wacher and Robinson 2008, Shah et al. 2013).

d) *Have translocation programs released animals over one or multiple events?*

The majority of translocation projects examined released animals to the same release site over 2 or more events spanning multiple years (e.g. Wacher and Robinson 2008, Saltz 2008, Islam et al. 2010b, Wronski et al. 2011). Lovari et al. (2010) recommended that consecutive Apennine chamois releases should be concentrated in time to minimize dispersal.

Using optimization models, Tenhumberg et al. (2004) suggested that the optimal release strategy is to release small groups of animals over multiple years. Advantages of doing so are: (1) the success of the entire translocation program does not rest on the success of one release, (2) management can be adjusted between releases as needed, and (3) especially if individuals are sourced from a captive population, the source population may continue to grow at a higher rate (Tenhumberg et al. 2004).

e) *Have translocation programs released animals at one or multiple release sites?*

Many species examined were released to numerous sites. Doing so helps to reduce the risk of any one population succumbing to stochastic events. Animals were released to entirely different release areas (including different countries, e.g. Arabian oryx reintroduction to Jordan, Isreal, UAE, Saudi Arabia) and/or a number of different release sites within the same general area (e.g. Arabian oryx released to 3 sites in the Negev Desert, Isreal, Saltz 2008; mountain gazelle released to 4 sites within the Ibex Reserve, Saudi Arabia, Wronski et al. 2011). Multiple release sites may be managed as a meta-population to promote genetic diversity (e.g. scimitar-horned oryx, Gilbert and Woodfine 2008; wisent, Belousova et al. 2005).

f) *How has founder group size and composition affected previous ungulate translocations?*

In a review of reintroductions of Artiodactyla (even-toed ungulates), Komers and Curman (2000) found that all releases ≥ 20 animals grew, whereas trends of those that included < 20 animals were more variable. Population growth in relation to founder group begins to level off after 20 animals, which suggests that **founder groups much larger than 20 animals do not increase the probability of a translocation's success** (Griffith et al. 1989, Wolf et al. 1996, Komers and Curman 2000). However, larger groups may be necessary in some situations/for some species. For example, outcomes of bighorn sheep translocations suggest larger founder groups (~40 animals) may be required (Singer et al. 2000).

Sex and age composition of founder groups also affects translocation outcomes, with older mature animals likely to positively influence population growth (Komers and Curman 2000, Bar-David et al. 2005).

Trade-off decisions have had to be made between population growth and other factors that may affect translocation success. For example, young scimitar-horned oryx (5 – 7 months) were chosen for release to Bou Hedma National Park, Tunisia, despite the fact that doing so resulted in initially low breeding rates (Gordon and Gill 1993). This decision was made because managers felt that young animals would form a 'more integrated social unit' than adults, would better adapt to local conditions, and would involve lower transport costs (Gordon and Gill 1993).

g) Are soft or hard approaches typically adopted? Have they been observed to influence outcomes?

Most ungulate translocations examined adopted soft-release approaches (e.g. Kiwan et al. 2008, Wacher and Robinson 2008). Whether a hard or soft release was implemented depended upon the particular species and situation in question. Researchers/managers of antelope reintroductions to arid environments in Africa and Arabia strongly recommended and adopted soft-release approaches (e.g. Gordon and Gill 1993, Simkins 2008, Wacher and Robinson 2008). Relatedly, nearly all reintroductions that used captive-bred animals followed soft approaches—upon arriving at a release site, animals were held within enclosures of varying sizes for periods of days to many months depending upon the project.

This 'acclimation' period was adopted to allow animals to recover from any stress experienced during transport and to adapt to their new environment. In some cases, it was also used to encourage the formation of stable social structures (Gordon and Gill 1993) and to promote bonding between breeding pairs (Woodfine et al. 2011) prior to release.

Researchers have observed or hypothesized benefits of using a soft-release approach, including more favorable social coherence and reproductive success (Wacher and Robinson 2008), reduction of potential negative effects on vegetation (Simkins 2008), prevention of dispersal from the release area (Ryckman et al. 2010), and potentially lower initial predation-caused mortality (Kock et al. 2010).

However, in a general review of translocations (i.e. not just ungulates), Griffith et al. (1989) found no consistent relationship between program success and whether a hard versus soft release approach was used. Hard releases were found to be just as effective as soft releases in reintroductions of large antelopes to Debshan Ranches in Zimbabwe (Swanpoel and Dunham 2013). Similarly, using a soft-release approach was considered unnecessary or even unfavorable for Apennine chamois reintroduction in Italy (Lovari et al. 2010).

h) What are some observations that have been made when using wild-born vs. captive-born individuals?

Most ungulate reintroductions examined within IUCN case studies used captive-born animals and as mentioned, these programs have experienced varying degrees of success. In a general review of translocations (i.e. not just ungulates), Griffith et al. (1989) found that translocations that used exclusively wild-caught animals were more likely to be successful than those that used exclusively captive-reared animals.

However, **it appears that no definitive conclusions regarding potential outcomes of using wild-born vs. captive-born ungulates can be made across species and programs.** Some programs found that the released animals adapted well to the wild (Perelberg et al. 2003), including zoo-bred animals (Gilbert and Woodfine 2008), while others suggested captive-born animals do more poorly upon release than wild-born animals (Lovari et al. 2010, Shah et al. 2013).

'Captivity' can imply a range of different environments – from zoos to pens within the species' natural environment—and animals kept in captivity are managed to varying degrees. Therefore, the conditions that 'captive-born' individuals experience prior to being released into the wild may affect program outcomes. For example, mixed releases of Persian fallow deer to Soreq Nature Reserve (and surrounding areas) indicated that animals from the Jerusalem Biblical Zoo exhibited maladaptive behaviors that likely increased their mortality risk as compared to animals from Hai-Bar Carmel Reserve (Saltz et al. 2011).

i) Have released animals adapted well to their new conditions?

Animals in many translocations appeared to have adapted well to their new environments and showed appropriate social behaviors and group structure (e.g. scimitar-horned oryx in Tunusia, Gilbert and Woodfine 2008; Persian fallow deer to Israel, Perelberg et al. 2003).

However, **translocated animals may need time to fully adapt to new conditions at their release site** (Scillitani et al. 2013). For example, released alpine ibex in Italy required up to 3 years to acquire knowledge on location of available forage and socially integrate with resident animals (Scillitani et al. 2012). **Survival of young may be low during the transition period** when translocated animals adjust life-history characteristics to release areas, which may in turn jeopardize reintroduction success (Whiting et al. 2011). For example, female bighorn sheep translocated to Utah adapted the timing and synchrony of parturition over time to meet environmental conditions at the release site (Whiting et al. 2011); however, survival of females' young to their first winter was lower during years when parturition had not yet been adjusted (Whiting et al. 2011).

Animals may have a particularly difficult time adapting if environmental conditions at the release site are much different from those at the source site; Al Zaidaneen and Al Hasaseen (2008) suggested that translocated Arabian oryx from Shaumari reserve had difficulty adapting to their new environment in the Wadi Rum Protected Area, Jordon due to differences in climate, vegetation and topography (Al Zaidaneen and Al Hasaseen 2008).

Authors of previous projects have proposed that conspecifics can help released animals to adapt more quickly (Dolev et al. 2002, Saltz 2008). For example, Persian fallow deer released in Israel appeared to adjust to wild conditions more quickly when conspecifics were already present, as indicated by quicker establishment of home ranges by animals from later releases (though this finding was not significant) (Dolev et al. 2002). Arabian oryx that had been reintroduced to Israel appeared to help newly released oryx establish [ranges] and learn the landscape (Saltz 2008). However, repeated releases also appeared to temporarily destabilize social groups (Saltz 2008).

j) Has dispersal been a problem in ungulate translocations? What affects dispersal?

Dispersal from the release area is common, though dispersal was constrained in many translocations examined by fencing. **The probability that animals disperse from the release area and the distance to which they do so are affected by a number of factors, including:**

- **Environmental conditions at the release site and surrounding areas** (Simkins 2008, Ryckman et al. 2010): For example, elk released to the 4 sites showed differences in dispersal, which may have been influenced by differences in habitat characteristics (forage availability, human population densities, climate) between the sites and/or the presence of conspecifics (Ryckman et al. 2010). Furthermore, environmental conditions, especially vegetation, directly affected dispersal patterns of Arabian oryx reintroduction to Dubai, UAE (Simkins 2008). Habitat conditions were also found to be a key feature driving dispersal in mountain gazelle reintroduction to the western Empty Quarter in Saudi Arabia (Islam et al. 2011c).

- **Soft- versus hard-release approach** (Gordon and Gill 1993, Ryckman et al. 2010): For example, elk that were held for moderate periods dispersed significantly lower distances than those held for shorter periods (Ryckman et al. 2010). However, holding period could not completely explain variation in dispersal distances and results suggest there may be a threshold above which holding period no longer affected dispersal (Ryckman et al. 2010).

- **Time since release** (Scillitani et al. 2012, 2013): For example, adult alpine ibex males released in Italy initially explored their surroundings to find preferred resources, but eventually reduced their home ranges to sizes similar to resident animals (Scillitani et al. 2012).

- **Sex and age** (the effects of which may differ between species)(Ryckman et al. 2010, Lovari et al. 2010, Scillitani et al. 2012): For example, elk calves generally stayed closer to the release site and dispersed in a different direction than adults (Ryckman et al. 2010). Adult male elk dispersed significantly farther than calves (of either sex), but there was no difference in the mean dispersal distance between adult males and females (Ryckman et al. 2010). In contrast, behaviors of individual alpine ibex males translocated to Italy were highly variable, but there was no relationship between spatial movements and age (though all were subadults or adults; Scillitani et al. 2012). In apennine chamois reintroductions to Italy, sub-adult males were most likely to disperse, but they were less likely to do so if several mature females were present at the release site (Lovari et al. 2010).

k) How has habitat quality affected translocation success?

Many authors of the IUCN case studies stress the importance of ensuring suitable habitat is available within the release area, and if not, improving habitat before release. Otherwise, poor habitat quality can have dire consequences for a released population (Shah et al. 2013). In a general review, Griffith et al. (1989) found that translocations more likely to be successful if habitat quality at the release site was high, though there was no consistent relationship between success and habitat improvement.

If suitable habitat was not present, supplementary feeding has been used to promote growth in the reintroduced herd and to protect the ecosystem from further grazing pressure (Simkins 2008, Islam et al. 2010b, Zhigang 2013).

Environmental conditions, especially vegetation, can also affect animals' behavior and dispersal patterns after release (Simkins 2008, Ryckman et al. 2010). For example, Arabian oryx reintroduced to the UAE were more likely to use feed stations during the summer when natural forage was low and dispersed more over the longer-term as environmental conditions improved within the Dubai Desert Conservation Reserve (Simkins 2008).

l) How does the presence of predators affect ungulate translocations?

Previous translocations of ungulates have noted high initial mortality of released animals in areas with abundant predators, which may ultimately risk program success. For example:

- Wolf mortality accounted for 25% of elk mortalities (wild-sourced) up to 6 years after they were released to Ontario and bear predation on neonates was also noted as a potential factor limiting recruitment (Rosatte et al. 2007). Initial mortality was especially high at one of the four release areas due in large part to predation (Rosatte 2013).

- A reintroduced population of Lichtenstein's hartebeest (captive-bred) in Malilangwe Wildlife Reserve is predicted to go extinct without further releases due to unsustainable levels of predation of adult females by lions and other large carnivores (Clegg et al. 2013).

- Six of seven radio-collared desert bighorn sheep released to the Fra Cristobal Mountains (from a fenced refuge) in 1997 were killed by cougars within 18 months of their release (Phillips 2013).

- High populations of lions, leopards, cheetahs, spotted hyenas and African wild dogs are thought to have killed hirola that were still adapting to their new environment during early stages of reinforcement in Tsavo East National Park in Kenya (Kock et al. 2010).

In a general review, Griffith et al. (1989) found that translocations more likely to be successful if no or few predators are present.

m) How does the presence of other (non-predator) species affect ungulate translocations?

Interactions with other (non-predator) animals, including livestock, has been found to affect released animals and ultimate outcomes of translocation programs (Gordon and Gill 1993, Singer et al. 2000, Wronski et al. 2011). Potential negative effects may be due to aggressive behaviors (Gordon and Gill 1993), competition for resources (Wronski et al. 2011) and disease (Singer et al. 2000).

n) What health-related problems have occurred in ungulate translocation programs?

Disease has threatened captive and free-ranging populations in previous translocation programs (e.g. Islam et al. 2011, Scillitani et al. 2011, Goudarzi et al. 2015). Nutritional deficiency was found to cause mortality in translocated tule elk in California, which threatened the reintroduction's initial success (Gogan et al. 2013). Disease risks have also posed challenges in obtaining suitable source animals for releases and/or using reintroduced populations as future source herds themselves (Gogan et al. 2013, Rosatte 2013).

o) *What issues regarding human dimensions are commonly encountered during ungulate translocation programs?*

Unexpected social challenges have threatened the success of previous translocation programs and numerous authors of IUCN case studies stress the importance of conducting social outreach throughout a translocation program. For example:

- Reintroductions or reinforcements of animals have impacted local people, who can retaliate if they are not included in the process and do not understand the program's objectives and importance (Rosatte et al. 2007, Saltz et al. 2011, Wronski et al. 2011).
- Even unintentional illegal hunting due to misidentification of animals can undermine translocation efforts if the hunters are not educated on ungulate identification before animals are released (Rosatte et al. 2007).
- Management of translocated species has needed to be balanced with tourist expectations (e.g. Lichtenstein's hartebeest reintroduction to the Malilangwe Wildlife Reserve, Zimbabwe, Clegg et al. 2013).
- Social issues have arisen between members on the same management team, which has hindered overall progress (Phillips 2013).

Political challenges have also been noted, and are particularly tricky when reintroduction efforts occur in multiple countries or jurisdictions (Perzanowski and Olech 2013). However, some political pressure can increase a program's public profile and ultimately expand public awareness (Kock et al. 2010).

p) *What role have fences played in translocations?*

Fencing has been used in translocation programs to protect animals from threats to their existence during the captive-breeding, release and post-release stages of translocation. Many releases examined were to fenced areas, especially those of antelopes in arid habitats of Africa and the Middle East (e.g. Gordon 1993, Gilbert and Woodfine 2008, Simkins 2008, Islam et al. 2010a, 2011a,b, Clegg et al. 2013, Shah et al. 2013). These releases were usually to Protected Areas or Nature Reserves that were several hundred to several thousand km^2 large (though some were as small as 12 km^2, Eid and Ananbeh 2010, and some as large as nearly 14,000 km^2, Kock et al. 2010).

Most fences were intended to protect animals from poaching, which was a major threat to many antelope species in arid landscapes. These populations were held in relatively large fenced areas within their natural habitat, often with minimal or no support over long period of time.

Fencing was also used for soft-releases and for programs that captive bred animals in fences within their natural range. In contrast to releases of animals to large-scale long-term fenced areas, these techniques involved a high degree of support and management of the fenced population, and commonly fenced smaller areas.

q) *Has (predator-proof) fencing been effective in protecting animals from predators and stimulating population growth?*

Fencing appears to have been relatively effective in protecting enclosed ungulates from predation and stimulating growth in the protected populations; although occasional predation events have occurred when predators penetrate the fencing.

- Lichtenstein's hartebeest kept within two 500 ha enclosures (as breeding nuclei) in Zimbabwe rapidly increased (once founder numbers were high enough) (Clegg et al. 2013). However, unsustainable predation rates of adult females by lions and other carnivores threaten the free-ranging population, which is declining (Clegg et al. 2013).

- Seven roan antelope were brought into a 302 ha predator-proof enclosure in 1994 to protect them from unsustainable predation by lions (ultimately driven by habitat change that facilitated influxes of alternate prey to antelope habitat) (Harrington et al. 1999, Grant et al. 2002). The fenced population expanded to 39 animals by May 2001, in contrast to the free-ranging population, which failed to recover despite habitat management (Grant et al. 2002).

- In 2012, 48 hirola were moved to a 25 km^2 predator-proof enclosure within their historic range in Kenya (Ali unpublished work 2016). The enclosure had relatively high range quality and was also free of any livestock (Ali unpublished work 2016). The fence has been effective in keeping out all carnivores (lions, cheetahs, African wild dogs) except leopards (see part 's' below, Ali unpublished work 2016). The enclosed hirola population nearly doubled in three years, increasing to 86 animals by 2015 ($\lambda > 1.0$); this is in contrast with hirola declines ($\lambda < 1.0$) within two unprotected areas, one of which also had high range quality (Ali unpublished work 2016). The hirola predator-proof sanctuary represents one strategy adopted within the Hirola Recovery Plan, which aims to address causes of hirola population decline and to improve relations with local communities (Ali unpublished work 2016).

r) *What issues have been encountered when ungulate populations are held within fences?*

- **Fencing restricted animals' ability to move in response to environmental conditions, which led to mass die-offs in some cases.**
    - A reintroduced population of Arabian oryx in Mahazat as-Sayd Protected Area, Saudi Arabia (2,244 km^2 fenced area) experienced mass die-offs during the summer months of years with little rainfall (Islam et al. 2010a). Fencing surrounding the reserve appeared to restrict the oryx's natural long-distance movements in response to rainfall and subsequent forage availability (Islam et al. 2010a).
    - Fencing around Mahazat as-Sayd Protected Area also likely restricted movement of reintroduced sand gazelles, which typically move long distances in response to stressful conditions and forage availability (Islam et al. 2011a).
- **Inter- and intra-antagonistic interactions were noted within fenced populations.**
    - Scimitar-horned oryx reintroduced to a 2,400 ha fenced area within Bou Hedma National Park in Tunisia regularly fought with Addax that were also introduced to the Park, which had potentially severe consequences on the Addax population (Gordon and Gill 1993). Under natural conditions, the two species remain largely separated across the landscape and rarely interact

(Gordon and Gill 1993). Therefore, restriction within the fenced are likely increased their interactions and subsequent fighting (Gordon and Gill 1993).
    - Adult male Lichtenstein's hartebeest were aggressive and had to be removed from the enclosure to prevent fighting with subadult bulls (>1 year old) (Clegg et al. 2013).
    - Aggression between hirola males held within a 25 km^2 fence has caused the death of at least two males (Ali unpublished work 2016).
- **Fenced populations have needed to be managed for carrying capacity.**
    - A reintroduced population of Arabian oryx to Mahazat as-Sayd Protection Area in Saudi Arabia (2,444 km^2 fenced area) increased from 9 in 1988 to 613 in 2006, but declined sharply thereafter to 324 animals in 2008 (Islam et al. 2010b). Managers decided to remove oryx (and sand gazelle) from the area to prevent additional mortalities resulting from the lack of vegetation (Islam et al. 2010b).
    - It is acknowledged that as a fenced population, reintroduced Arabian oryx to the Dubai Desert Conservation Reserve will need to be managed according to an established carrying capacity (Simkins 2008).
    - The rapid expansion of a hirola population held within a 25km^2 enclosure (from 48 to 86 animals total) necessitated the expansion of the enclosed area (Ali unpublished work 2016).
- **Predators that breached the fence posed risks for animals and/or humans.**
    - Lions and leopards occasionally penetrated a fence surrounding a breeding nucleus of Lichtenstein's hartebeest in Zimbabwe, and killed both juvenile and adult hartebeest (Clegg et al. 2013). While trying to remove a lion from the enclosure, a wildlife manager was mauled (Clegg et al. 2013).
    - A predator-proof fence constructed to protect hirola has not been completely effective against leopards, which have occasionally jumped over the fence and killed hirola; however, this is not considered to be a major issue overall (Ali unpublished work 2016).
- **Diseases have broken out in fenced populations**, which can be especially catastrophic if all captive animals are kept within one enclosure (see health section above). But note: Other cases have had no problem with disease to date (e.g. hirola predator-proof fencing, Ali unpublished work 2016).
- **Predators may learn to use fences to their advantage when hunting, putting additional pressure on prey species** (VanDyk and Slotow 2003):
    - Wild dogs (*Lycaon pictus*) that were reintroduced to a 500km^2 fenced area in South Africa chased prey to fence lines and killed them while the prey was stunned by the electrified fence or confused (VanDyk and Slotow 2003). Doing so allowed them to catch larger prey along the fence line than they were able to away from the fence (VanDyk and Slotow 2003). Though in this case the wild dogs were held within the fence, predators outside the fence could presumably learn similar habits.
- **Social and financial issues have posed challenges to fencing programs.**
    - Locals losing grazing lands/rights as a result of the fence, political unrest along the Kenya-Somalia border, and inadequate funding and support from the international community have all been major challenges in hirola conservation (Ali unpublished work 2016).

Fencing may more negatively impact species with demanding habitat requirements. For example, reintroductions of dorcas gazelles to fenced areas has been largely successful, as compared to dama gazelles, which have done relatively poorly (Stanley-Price unpublished work). This contrast highlights potential challenges in reintroducing a social species with relatively more demanding habitat requirements (dama gazelle) (Stanley-Price unpublished work).

Appendix 3 References

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