A FEASIBILITY ASSESSMENT OF WOLF REINTRODUCTION TO THE CAPE BRETON HIGHLANDS

By

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Abstract

This thesis reports an assessment of the feasibility of establishing a viable population of Wolves (*Canis sp.)* on Cape Breton Island, Nova Scotia as a means of determining if Wolves can limit Moose (*Alces alces*) abundance in Cape Breton Highlands National Park (CBHNP) to densities desired by Parks Canada ecologists (e.g., 0.5 Moose/km²). As apex carnivores, Wolves are effective in preventing ungulate populations from reaching hyperabundant levels. It is uncertain which *Canis* species was present in the region prior to pre-European settlement and therefore, I used ArcGIS and Marxan to estimate the amount of available habitat for both Gray (*Canis lupus*) and Eastern Wolf (*C. lycaon)* to form territorial packs on Cape Breton Island. The population viability analysis programme *VORTEX* was used to predict the population size of a viable population, as well as the size needed to limit Moose (*A. a. andersoni*) abundance to levels less likely to impact vegetation. *VORTEX* simulated the population by evaluating the annual life cycle and tracking mate selection, reproduction, mortality, increment of age by one year of individuals, as well as migration among populations, removals, and supplementation. Based on the *VORTEX* model simulations of the long-term viability of both Wolf species, the optimal and suboptimal habitat within the National Park and adjacent highland areas could support 30 *C. lupus* (16 inside the Park and 14 outside of the Park), or 33 (17 inside the Park, and 16 outside of the Park) *C. lycaon*, respectfully. Results identified several factors important to the long-term viability of Wolf populations: 1) the percentage of adult females breeding; 2) carrying capacity; and: 3) mortality rates. If the percentage of female breeders (*C. lupus*) remains 55% or higher, and Wolves are not subject to immediate and long-term anthropogenic mortality risk (modelled as 30% inn mortality pups, 10% in adults), the population maintains a carrying capacity of n = 36 with a low probability of extinction (<0.25). However, based on mainly negative public attitudes to Eastern Coyote in the region, it is presumed that mortality rates will be high outside of the National Park; a Park-only population size of 16 Gray Wolves would not be viable, nor reduce Moose density to desired levels. Even if mortality rates outside

the Park were low, population models suggested Wolves may not reduce Moose to desired levels. A static functional response model of Gray Wolves to changing Moose density suggested that a larger Wolf population than theoretically modelled would be required to reduce the local Moose population to desired densities. A preliminary deterministic modelling approach indicated that 30 Wolves might reduce the Moose population to desired densities when the Moose population has a growth rate of 0.1. In conclusion, the likelihood of reintroduced Wolves reducing Moose in CBHNP to desired levels depends on mortality rates outside of the Park is low because the Park itself is too small to contain a viable population of Wolves. Further work on societal attitudes to Wolves would be vital before any Wolf reintroduction program is considered.

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Table of Contents

List of Tables Page

Chapter 3

Chapter 4

List of Figures Page

Chapter 4

Chapter 1. General Introduction

Introduction

The staff of National Parks (NP) in Canada strive to maintain the ecological integrity of designated parks through ecosystem-based management, which includes maintaining and protecting representative biodiversity for that ecoregion (Charest *et al.* 2000; Herremans and Reid 2002; Waithaka 2008). Maintaining ecological integrity can be difficult to achieve in parks when high densities of ungulate species impact representative (e.g.*,* native to area) plant communities, and in extreme cases, may result in extirpation of some native biodiversity (Corbett 1995; Hobbs 1996; Peek 1997; Waithaka 2008; Smith *et al*. 2014). Several studies throughout the Atlantic Region National Parks (e.g.*,* Terra Nova NP, Gros Morne NP) have documented significant impact of Moose (*Alces alces*) herbivory on vegetation. The introduction of Moose to Newfoundland in 1878 and 1904, and the subsequent closure of public hunting seasons in Gros Morne NP Reserve (GMNPR; established in 1973), resulted in a rapid population increase that exceeded 4 Moose/km² in certain areas of the Park (Connor *et al.* 2000; McLaren *et al.* 2000). Empirical evidence has suggested this population influx altered native forest successional patterns, vegetative composition, and browse availability (Connor *et al.* 2000; McLaren *et al.* 2004; Forbes 2006; McLaren *et al.* 2009a). A similar problem exists in Terra Nova NP where Moose are directly affecting the health and integrity of the forest (Gosse *et al.* 2011; Parks Canada 2013).

In Cape Breton Highlands National Park (CBHNP), Moose (*A. a. andersoni)* from Elk Island NP, Alberta were introduced in 1947-1948, after the local Moose population had been functionally extirpated due to excessive hunting and habitat alteration (Bridgland *et al.* 2007; Smith 2014). The Moose population increased and by the 2000's there were more than 5000 individuals within northern Cape Breton Island (CBI). Densities within the National Park had exceeded 4 Moose/km² by 2004 (Smith *et al.* 2015). The increase was due to abundant food supply and low mortality rates from natural predators (Smith 2014;

Smith *et al*. 2015). The successional regrowth provided an increased food supply for the reintroduced ungulates following a Spruce Budworm (*Choristoneura fumiferana)* outbreak in the 1970's (MacLean 1988; Lockett 2001; Smith *et al.* 2010; Franklin 2013). Traditional predators of Moose in Cape Breton were Wolf (*Canis* sp.), Black Bear (*Ursus americanus*), and humans (e.g.*,* aboriginal people, settlers; Smith 1940; Lohr and Ballard 1996; Forbes *et al.* 2010). Black Bear have been present since Park establishment and are known predators on Moose calves (Franzmann and Schwartz 1986; Ballard and Larsen 1987; Ballard 1992; Kotchorek 2002), but their ability to limit Moose populations is not apparent (Ballard *et al.* 1990; Zager and Beecham 2006). Hunting typically was not allowed in National Parks, and, other than poaching, predation by people was minimal since Park establishment in 1936. In 2015, Parks Canada partnered with the Unama'ki Institute of Natural Resources in Eskasoni, N.S., initiating research on the effects of Moose removal on boreal forest recovery (Parks Canada 2016).

There is uncertainty about which Wolf species were present in Cape Breton, either the Gray (*Canis lupus*) or Eastern Wolf (*C. lycaon*; Wilson *et al.* 2000, Benson *et al.* 2012, Whitaker and Beazley 2017), but, notwithstanding, by roughly 1928 the large canid that could have limited Moose populations in Cape Breton was classified as extirpated (Obbard *et al.* 1987). To date, there is no solid evidence that suggests which canid would have occupied the landscape post-European settlement (McAlpine *et al*. 2016). It is also uncertain whether the maritime region had ever supported a minimum viable population size due to scant documentation of historical Wolf numbers (Lohr and Ballard 1996). Therefore, any analysis of landscape-level Wolf reintroduction feasibly might lend insight to whether a viable population of either species could have historically existed in the area or not.

Adult female Gray Wolves weigh approximately 18 to 55 kg and adult males weight 20 to 80 kg (Mech 1974). Eastern Wolves are smaller in size; females weigh approximately 24 kg and males 29 kg (COSEWIC 2015). Forbes and Theberge (1996) have stated that Eastern Wolves in Algonquin Park, Ontario are inefficient predators of resident Moose (*A. a. americana*) there due to

their relatively small body size, and therefore do not account for a major source of Moose mortality. However, a recent study by Benson *et al.* (2017) in Algonquin Park found that Eastern Wolves killed enough Moose that the kill rate is similar to the purported by the functional response curve of Gray Wolf kill rate associated with a changing Moose density (Messier 1994). Benson *et al*. (2017) noted that the higher than expected kill rate may be attributed to the Moose population being nutritionally stressed, and therefore vulnerable to predation. Because *A. a. andersoni* in CBHNP are a larger sub-species than *A. a. americana* in Algonquin Park, it is unlikely that Eastern Wolf will exhibit a higher kill rate than the larger Gray Wolf. The Gray Wolf is a very efficient predator of large ungulates such as Moose, and tend to occupy areas where the large ungulate prey is available (Bergerud *et. al* 1983; Messier 1994).

The increased Moose density in CBHNP has resulted in changes to forest community type and the amount of preferred browsing species (Basquill and Thompson 1997). The boreal forest section of CBHNP comprised 11% of the total Park area, but one third of this forest has been converted to grasslands (Smith *et al.* 2015). Recent research in CBHNP suggested that vegetation could re-establish if Moose were excluded. This conclusion was based on stem density and percent cover difference in two exclosures (Skyline and North Mountain), compared to adjacent control plots exposed to browse (CBHNP 2014). One exclosure showed high potential (e.g., increased stem density versus control sites) for vegetative growth, while the other exclosure showed minimal regeneration potential (CBHNP 2014). The lack of regeneration in the Skyline exclosure was due to replacement of woody vegetation by grass which compromised germination and survivorship of seedlings. The North Mountain exclosure showed high potential for regrowth because many small live trees were present (J. Bridgland pers. comm. 2017). The alteration of forest successional dynamics caused by overabundant Moose may persist for decades even after Moose are excluded from the ecosystem, or reduced to lower densities. These effects have been identified in similar geographical regions (Pastor *et al.* 1993; McLaren *et al.* 2009b).

Concerns over the impact of the species with high (i.e., ‘hyperabundant’) population density on Park ecosystems resulted in policy and practices that attempt to minimize loss of ecological integrity (Wagner 1995; Parks Canada 2001). Hyperabundant describes “a population of species, where numbers clearly exceed the upper range of natural variability that is characteristic of the ecosystem, and where there is a demonstrated impact on ecological integrity” (CPCCCG 2013). The present policy in CBHNP is to “employ active Moose density reduction measures, such as Moose removal and habitat manipulation, as warranted” (Smith *et al.* 2015). Although reduction may be the goal, it is the means used to reduce density that have become controversial. For example, the culling of native species in National Parks is problematic for those who consider such parks as sanctuaries from human activity (Slocombe 1993; Mast and Mast 2010). The reintroduction of predators is controversial because of the inevitable impact on other wildlife species (Smith *et al.* 2003; Ballard *et al.* 2003), and potential conflict with land use (e.g., game and livestock production) adjacent to a Park (USFWS 1990; Forbes and Theberge 1996; Paquet *et al.* 1999). The use of fire to alter vegetative communities also is a concern for infrastructure and trans-boundary management (Bond and Keeley 2005; Ryan *et al*. 2013).

Worldwide, the reduction of a population by culling a percentage of animals is commonly used to achieve a desired density (Bergerud *et al.* 1968; Bradford and Hobbs 2008; Millspaugh *et al*. 2008; Wright *et al*. 2012; Simard *et al*. 2013). The application of culling in Canadian National Parks was relatively rare until the 1990’s when culls were conducted on hyperabundant populations of White-tailed Deer (*Odocoileus virginianus*) in Point Pelee NP, Ontario (Parks Canada 1996; Wright 1999; CBC 2015). In Newfoundland National Parks, Moose were recently reduced in Gros Morne and Terra Nova using regulated culls (Canadian Press 2012; CBC 2013; Parks Canada 2015). Plans are in place for the continued expansion of the cull in Terra Nova NP (CBC 2016).

In CBHNP, the reduction of Moose densities to <0.5 Moose/km^2 was considered an appropriate threshold that would significantly reduce the impact of selective browsing on Park vegetation (Smith *et al.* 2015; Parks Canada 2016).

The recent population density was reported at 1.9 Moose km² (Smith *et al.* 2015). Densities greater than 1.9 Moose km² have been shown to impact browse availability (Andreozzi *et al*. 2014; Persson *et al.* 2005; Parks Canada 2016). A pilot project to reduce the Moose population by 90 % (approximately 40 individuals) in a 20 km² area on the plateau at North Mountain, was implemented in November 2015 (Smith *et al*. 2015). A total of 37 Moose were removed from the study area with the help of First Nations harvesters (Parks Canada 2016). The population was monitored after the first cull and a second cull was undertaken in autumn 2016 with approximately 50 animals removed (J. Bridgland pers. comm. 2017).

A cull was initiated by Parks Canada to reduce the Moose densities because there was less stakeholder support for alternative methods. Other methods were supported but “lethal removal” was overwhelmingly (16:1) rated as the most feasible or preferred option (Smith *et al*. 2015). As well, the Federal Government recognized access to Moose for Cape Breton First Nations and because CBHNP is federal land, accommodation for this right was granted, in part, using CBHNP (Province of NS 2011; Smith *et al.* 2015). The other management possibilities considered prior to the cull were experimental translocations, confinement, reduction by fertility and birth control, herd driving, and increased levels of natural predation through Wolf reintroduction. These options have been put forward as a set of alternative strategies for the management of hyperabundant species in other parks with similar problems. However, there is limited information on their efficacy, and although it is not the intention for CBHNP to alter the present culling option, there is value in assessing their efficacy and thereby allow the use of culling to be assessed within the larger set of available tools, and within the context of due diligence and effective governance. Of the nine strategies examined “predator reintroduction” was the only option besides lethal removal which was identified as a first-choice selection by participants during the Moose hyperabundant workshop in 2014 (Smith *et al.* 2015).

The reintroduction of Wolves to other areas is one tool that has had considerable success in reducing the impact of hyperabundant ungulates on vegetation in other areas. As efficient predators, Wolves can alter the abundance and distribution patterns of local prey populations (Mech and Karns 1977; Fritts and Mech 1981; Dale *et al.* 1994; Schmidt and Mech 1997). This activity results in lower browsing impact on desirable forage species due in part to changing ungulate feeding and herding behavior (Messier 1994; McLaren and Peterson 1994; Frank 2008; Licht *et al*. 2010; Ripple and Beschta 2012). In Yellowstone NP, USA, for example, reintroduction of Gray Wolf in 1995 resulted in a considerable Elk (*Cervus elaphus*) population decrease (>50%) by the mid-2000's (Smith *et al.* 2003; White and Garrott 2005). Despite the debate on whether Wolves are a keystone species (i.e., having the ability to influence lower trophic levels) or not (Noss *et al.* 1996), Wolves have been associated with changes in forest successional patterns (Ripple *et al.* 2001; Beschta and Ripple 2009; Ripple and Beschta 2012).

The efficacy of increasing natural predation rates on hyperabundant ungulates in the Atlantic Region National Parks has not been well studied. Although there has been a preliminary exploration of the ecological possibility for Wolf recovery in NS (Whitaker 2006), no study (at a scale appropriate to Cape Breton) has attempted to predict the viability for a hypothetically reintroduced Wolf population. Wolves have been extirpated from Newfoundland and the Maritimes for > 100 years (Goldman 1944; Forbes *et al.* 2010) and their prolonged absence from the landscape, and the perceived impacts by some stakeholders, reduces their reintroduction as a viable option (Lohr and Ballard 1996). As well, the landscape of Cape Breton has since been populated by a smaller canid, the Eastern Coyote (*Canis latrans x lycaon*), which is a hybrid between Western Coyote (*C. latrans*) and either Eastern Wolf (*C. lycaon*) or Gray Wolf (*C. lupus*; Rutledge *et al*. 2010; Benson and Patterson 2013). Eastern Coyote are not considered to be significant predators on adult Moose, or calves (Paquet 1992; Benson and Patterson 2013; Patterson *et al.* 2013) and, therefore, likely play a limited role in reducing Moose numbers. The potential for

introgression of Eastern Coyote genes with any introduced Wolf species could affect the success of a reintroduction program (Parker 1990).

There are additional reasons to consider Wolves as an option. National Parks strive to maintain representative natural ecosystems (Waithaka 2008), and as a native species, Wolves would be considered a component of the CBHNP ecosystem. Wolves are found to maintain ecological integrity in designated US National Parks and elsewhere in North America (Smith *et al.* 2003). Although other mammalian predators, e.g., Cougar (*Puma concolo*r) and Black Bear (*Ursus americanus*) are considered important components of healthy ecosystems and vegetative integrity (Soule *et al.* 2003; Ripple *et al.* 2014), Wolves are more effective in the regulation of indirect ecosystem effects on vegetation composition in forested regions (including boreal; Soule and Noss 1998; Pace *et al.* 1999; Sergio *et al.* 2008). In addition, the presence of a viable population of large canids would potentially negate the need for culling operations, which in 2015 cost approximately $300,000 (Parks Canada 2016). Notwithstanding the relative merits of economic, social, or biological reasons for a Wolf reintroduction, all of which would be relevant in any cost-benefit analysis, the initial information that is required is whether a reintroduction of Wolves is even likely to successfully reduce Moose density to desired levels in CBHNP.

Study area

CBI has a total land area of 10,416 km², excluding the inland Bras d'Or Lake system, and is located on the east coast of Nova Scotia (NS), Canada (46.25° N, 60.85° W – Figure 3.1). The island is approximately 140 km wide (east to west) by 175 km long (south to north). It is bordered by the Atlantic Ocean, and is only connected to mainland NS via the Canso Causeway, an artificial, rock-filled causeway. The island is divided into four counties: Victoria (2,870 km²), Inverness (3,830 km²), Richmond (1,245 km²), and Cape Breton (2,471 km²) with human populations of 7,115, 17,946, 9,293, and 101,619, respectively (Statistics Canada 2011). The most populated county (Cape Breton) has an average human density of 41 people/km² (Statistics Canada 2011). The Bras

d'Or Lake, located in the center of the island, has a surface area of approximately 1,100 km².

Cape Breton Highlands National Park (CBHNP; 46.73° N, 60.65° W) encompasses approximately 950 km² of CBI, facilitating the protection of the Maritime Acadian Highlands Natural Region. The shoreline consists of steep cliffs and deep river gorges that carve into an elevated highland plateau. The Park is characterized by three distinct land regions: Boreal Forest, Acadian Forest, and Taiga. The Boreal Forest covers approximately one-third of the Park, dominating elevated areas on the plateau. Representative boreal tree species include Balsam Fir (*Abies balsamea*), White Spruce (*Picea glauca*) White Birch (*Betula papyrifera*), and Mountain Ash (*Sorbus americana*). The Acadian Forest, occupying low-lying coastal fringes and deep, sheltered valleys covers just under one-third of the Park (Rowe 1972; Wein and Moore 1979). Associated Acadian tree species include Sugar Maple (*Acer saccharum*), Yellow Birch (*B. alleghaniensis*), American Beech (*Fagus grandifolia*), and Eastern Hemlock (*Tsuga canadensis*). The Taiga forest region is predominately characterized by krumholtz Black Spruce (*P. mariana*) trees and elevated boggy wetlands (Smith *et al.* 2015).

Project goals and thesis structure

The goal of this project is to assess the long-term viability of reintroduced Wolves (*Canis lupus or C. lycaon)* on CBI and present the feasibility of Wolf predation as a means of reducing Moose populations to acceptable levels (approximately 0.5 Moose/km^2 or 1 Moose/2 km²; Parks Canada 2015). The chapters are organized in sequence such that information in early chapters is then applied to following chapters. Because it is still uncertain which species of Wolf was historically present in the maritime region (McAlpine *et al.* 2016, Whitaker and Beazley 2017) the feasibility of reintroducing both species were explored. The carrying capacity is determined for both species (Chapter 3) and the theoretical population viability of both species is modelled based on their respective life history traits (Chapter 4).

Chapter 1. General Introduction: A general introduction on Parks Canada Policy and the issue of hyperabundant Moose in Cape Breton Highlands.

Chapter 2. Review of parameters used in Wolf reintroduction literature: This chapter determines a list of parameters relevant to the feasibility of Wolf reintroduction. The rationale for the selection of parameters and methods was used to assess the ecological feasibility of Wolf (*Canis sp.*) reintroduction to CBI.

Chapter 3. Systematic habitat modelling and estimation of carrying capacity for Wolves (*Canis lupus* and *C. lycaon*) on CBI: This chapter determines if CBI supports critical habitat for Wolves by analyzing the biophysical characteristics of the study area, and anthropogenic-related risk. I used ArcGIS (ESRI 2010) and Marxan (systematic conservation planning software) to assess this objective. A carrying capacity was determined based on the amount of optimal and suboptimal habitat, and was used as a modelling input to *VORTEX* in Chapter 4.

Chapter 4. Model simulation of viability of re-introduced Wolves (*Canis lupus* and *C. lycaon*) and subsequent impacts on Moose population in CBI: This chapter determines the Wolf population size, or requirements for a viable Wolf population size to maintain long-term viability in CBHNP via population viability analysis (PVA-*VORTEX*), as well as the Wolf population size needed to limit Moose to acceptable densities (e.g., 0.5 Moose/km^2).

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Chapter 2. Review of parameters used in Wolf reintroduction literature

Introduction

The number of parameters used in recovery analysis is extensive because they include both ecological and sociological values (Fritts and Carbyn 1995; Mladenoff *et al.* 1995, 1997; Noss *et al.* 1996; Soule and Noss 1998; Nie 2001; Carroll *et al*. 2003; Estes *et al.* 2011). There are many immediate threats that impede population re-establishment. Mortality risk from humans and habitat loss caused by land development are arguably the most critical (Noss *et al.* 1996). Without cooperation among government and non-government organizations (NGOs) at regional and cross-boundary scales, any attempt to recover extirpated large carnivores, such as the Wolf (*Canis sp*.), is practically impossible (Phillips and Parker 1988; Lohr and Ballard 1996; Rutledge *et al.* 2010). Conflicting policy mandates across multiple jurisdictions (e.g., Federal National Park and Provincial Government) make it difficult for managers to establish and manage Wolf populations that require large territories and wide dispersal ranges, whether the species is reintroduced or in the process of natural re-colonization (Forbes and Theberge 1996; Rutledge *et al.* 2010).

From early Wolf recovery programs and reintroduction assessments to contemporary ecological studies, anthropogenic disturbance has been the most pervasive factor affecting the welfare of canid populations (Carley 1975; Carley and Mechler 1983**;** Berger 1999; Mech 2003; Bangs *et al.* 2001, 2005; Foreman 2006). Large wilderness areas with extensive connectivity and abundant prey species are typically considered ecologically feasible for Wolf reintroduction if persecution by humans is not expected (Fuller *et al.* 1992; Soule and Noss 1998; Mech 1995; Smith *et al.* 2003). Although suitable areas have been identified, political opposition typically halts the implementation of proposed reintroductions (Parker 1987; Bennett 1994; Mladenoff *et al.* 1995; Harrison and Chapman 1997,1998; Mladenoff and Sickley 1998; Paquet *et al.* 1999)

Numerous proposals and environmental assessments completed in former Wolf range since the 1980’s have explored the feasibility of Wolf reintroduction. Typically, the parameters considered in early reintroduction research were based

on predation rates from places Wolves already occupied, Wolf ecology, and the biophysical and anthropogenic characteristics of the study area (Carley and Mechler 1983; Fritts 1993; Mech 1995). Early ecological studies have often made predictions on the biological success of Wolf reintroduction and the impact Wolves may have on the ecosystem and other animals (Bednarz 1989; Parker 1990). These ecological predictions associated with Wolf reintroduction were often subject to conjecture because of the lack of post-monitoring data (Bednarz 1989; Fritts 1990; Mech 1995). Baseline data on ecosystem function preceding a species extirpation are usually not available, therefore making predictions of post ecosystem function prior to any reintroduction is risky (Polak and Saltz 2011; Converse *et al.* 2013).

Before the reintroduction to Yellowstone National Park (YNP) in 1995, no studies had comprehensively tested the response of plant communities to changing ungulate demographics, and feeding behavior(s) at such a large scale. Researchers could not be certain what effects Wolves would have on prey and non-prey species until they were released and monitored (USFWS 1990). The successful reintroduction to YNP is considered one of the most salient acts of wildlife conservation in the 20th century (Smith *et al.* 2003; Smith and Ferguson 2012) and has facilitated a test of early research hypotheses, predictions of environmental impact, and other uncertainties. The YNP reintroduction became a benchmark for ensuing studies attempting to accurately forecast reintroduction outcomes.

The objective of this Chapter is to develop a list of parameters relevant to the feasibility of Wolf reintroduction, based on the review of available studies related to actual Wolf reintroduction and Wolf reintroduction planning. The parameters and methods will be used to assess the ecological feasibility of Wolf (Canis sp.) reintroduction to Cape Breton Island (CBI), which is discussed in subsequent Chapters. I will examine the parameters that other studies have used and apply the most typical and relevant ones that are available for use in my study on CBI.

Methods

Literature search and selection

I collected available North American (NA) and European (EUR) literature on Wolf (Gray Wolf, *Canis lupus*; Mexican Wolf, *C. l. baileyi;* Red Wolf, *C. rufus*) reintroduction plans (proposed-implemented (PI), proposed-implemented-failed (PIF)), reintroduction feasibility assessments (RFA), and recovery programs (RPR). I also collected Wolf-habitat use and selection studies (WHUS) and habitat modeling studies (MS). I did not specifically search for population viability assessments, however some of the reintroduction assessments included population viability assessment as part of their analysis (e.g., Carroll *et al.* 2006). I performed a literature review using the University of New Brunswick search engine (UNB WorldCat) to identify all applicable literature. I searched with the following keywords: (*Wolf, *wolves) AND (reintroduction, or recovery, or habitat, or feasibility). A total of 53 peer-reviewed studies relevant to Wolf reintroduction were reviewed. I am confident that the amount of studies reviewed was sufficient to meet the objective of this Chapter; the Wolf reintroductions are well published and there were few unique parameters among reviews.

Parameter selection, use and analysis

A total set of 29 parameters related to Wolf reintroduction planning were identified in the literature and categorized as: (i) biophysical; (ii) infrastructure; (iii) biological; (iv) population-related; and (v) sociological. The level of individual parameter use (quantification degree) among studies was tallied into one of five categories: (i) parameter quantified; (ii) parameter quantified with poor data; (iii) parameter considered, but not quantified; (iv) parameter not used, due to data limitations; and (v) parameter not identified in the study. A master chart (see Appendix 1) was used to sort parameters into respective categories, and to categorize their degree of quantification. Geographic habitat suitability studies for Wolves (MS, WHUS; e.g., Harrison & Chapman 1997, 1998) and studies that included population viability assessments (e.g., Carroll *et al.* 2004, 2006) were included in the master chart with all reintroduction and recovery plans (PI; PIF,

RFA, RPR). I quantified the relative importance of parameters among the 53 studies by calculating the mode frequency of use. A discussion of major lessons learned from implemented reintroductions was completed.

Results

Eight biophysical (land cover, prey base, water availability, climate, topography, hydrology, winter severity, and land area) and five infrastructure-related parameters (human density, road density, land ownership, land use, and livestock abundance) were identified. The biological parameters identified and assessed among the studies were pack territory size, native to area, hybridization risk, effects on scavengers, disease, intraspecific competition. Specific population related parameters included survival rates, pack size, Wolf density, sex-age ratio, natality, fecundity, mortality, and dispersal. These parameters were more often quantified when studies began to analyze the viability of the reintroduced species (e.g., Carroll *et al.* 2006) and the ecological impacts of reintroduction (e.g., USFWS 1987). Sociological parameters (public attitude and public behavior) were also identified, but seldom quantified due to the lack of funding for further examination (Appendix 1).

The amount of quantification of parameters varied among studies. Some parameters were quantified (e.g., road density), while others received limited quantification (e.g., winter severity), when expressed as a percentage of all studies (Table 2.1).

Parameters that were considered were combined in the same chart with the numerically quantified parameters for each individual study (Table 2.2), because some parameters held high importance among all the studies, even if they were not numerically quantified. Two of the Wolf reintroduction parameters (human attitude and human behavior) were placed in the ‘considered’ category because they were too difficult to quantify in most studies. The average use of these two sociological parameters (36%) was 24% higher than their numerical quantification usage by all other studies (12%). The combined percentage of

studies either numerically quantifying the parameter or considering the parameter fell in the 41-60% range (Table 2.2).

Habitat suitability studies for Wolves usually include a landscape analysis of mortality risk for Wolves based on human disturbance. The most commonly used biophysical and infrastructure-related parameters (quantified and considered by >80% of all studies) used to measure mortality risk for Wolves were land cover (biophysical), road and human density (infrastructure; Table 2.2). Land cover was split into two categories, urban and agricultural areas, which are known surrogates for identifying mortality risk in Wolves. All land cover used in subsequent analyses was either agricultural or urban areas. These parameters were used to scale the mortality risk (risk weightings, addressed in Chapter 3) for Wolves, and determine the amount of optimal and suboptimal habitat (Chapter 3). These were the only downloadable GIS data sets for CBI. Parameters quantified and considered by >61-80% of the time throughout the 53 studies were pack territory, historical occurrence, land use, and prey base. These parameters are important; however, no useable data were available for the study area, and therefore were not included in subsequent analysis. Local snow depth data was also not available. Pack territory sizes were used to determine carrying capacity for Wolves on CBI, however they were not used in calculation of the suitability of habitat, nor were any other parameters identified in the chapter.

Discussion

Objectives of reviewed studies

The objective(s) of the reviewed studies were to recover (through reintroduction or natural recolonization) populations of endangered and threatened Wolves in areas of their former range (n = 9; e.g., USFWS 1978; 1982; 1987), assess the ecological feasibility of reintroduction (n = 5; e.g., Bennett 1994; Paquet *et al.* 1999; Ratti *et al.* 1999), assess the landscape for suitable habitat (n = 39; e.g., Mladenoff *et al.* 1995; Gehring and Potter 2005), and assess the population viability of recovering Wolf populations (e.g., Carroll *et al.* 2004, 2006). Studies with such objectives commonly used parameters in

assessing the feasibility of a Wolf reintroduction. Several studies (not reviewed due to lack of quantifiable parameters) have further explained how Wolf reintroduction can be used as a tool for restoring biodiversity and maintaining ecological integrity (Griffith *et al.* 1989; Licht *et al.* 2010; Beschta and Ripple 2009; Lipsey *et al.* 2009); however, they did not specifically list parameters relevant to Wolf reintroduction.

Lessons learned from previous reintroductions

There have been five Wolf reintroduction attempts worldwide in the past 35 years that have physically released and monitored individual animals and packs (parameters used in each study are given in Appendix 1). They include two Red Wolf reintroductions or experimental re-establishments; the Land between the Lakes (LBL) recreation area in western Kentucky and Tennessee in 1983 (Phillips and Parker 1988), and eastern Tennessee's Great Smoky Mountains National Wildlife Park (GSMNWP) in 1991 (Parker 1990). A third Red Wolf reintroduction to North Carolina's Alligator River National Wildlife Refuge (ARNWR) was initiated in 1987 (Henry and Lucash 2000; USFWS 1987). Following the Red Wolf reintroductions, there were two Gray Wolf reintroductions in Arizona and New Mexico in 1998 (USFWS 2014) and Yellowstone Park and central Idaho in 1995 (Smith and Ferguson 2012).

All Red Wolf reintroductions were initiated by the US Fish and Wildlife Service (USFWS 1987). Due to opposition from citizens, the reintroduction of Red Wolves to the LBL was abandoned in the late 1980's (Fritts 1990). Although the 690 km² area was considered biologically suitable, the reintroduction was said to be interfering with the lives of citizens because of the impact of livestock depredation in the adjacent area and obstruction of local development projects (Fritts 1990). A total of 37 Red Wolves were released into the GSMNWP from 1991 - 1998, and 26 of these individuals were reportedly killed by humans outside of the Park. Red Wolves in GSMNWP could not establish home ranges within the 2,070 km² Park, likely due to limited legal protection outside Park boundaries. The low Wolf-pup survival resulted in the cessation of the plan in

1998 (Henry and Lucash 2000). The reintroduction to the ARNWR in North Carolina was still ongoing as of 2016, and there has been much debate regarding the long-term viability of the population, as the population is still considered “biologically unrecovered”. The area was initially considered feasible for a reintroduction because of the abundant prey, sparse human settlement, and low Coyote (*Canis latrans*) population, minimizing the potential of hybridization between both species. A review of the reintroduction from 1987 – 1993 suggested that successful re-establishment ultimately depended on the length of the acclimation period, the size of the recovery area that received full legal protection, the resistance of Wolves to human persecution, and the formation of social groups within the population (Phillips 1994). Several other studies concluded that mortality associated with dispersal was a major concern, since much of the land adjacent to the Park was under private and state ownership. In addition, the recent range expansion of the Eastern Coyote has caused hybridization to occur between species. This introgression, along with human resistance to recovery has resulted in the failure of this reintroduction (Phillips *et al.* 2003).

The Mexican Wolf has been monitored extensively by the US Fish and Wildlife Service since its reintroduction in 1998, and represents the most genetically distinct subspecies of North American Gray Wolves (Parsons and Nicholopoulos 1995; Vonholt *et al.* 2011). Extensive connectivity within the reintroduction management area (Arizona and Mexico) provided for dispersal opportunity between populations, causing the population to peak at a total of 114 Wolves by 2014 (USFWS 2015). Although growth occurred throughout the initial years of the reintroduction, the population has declined since 2014 due to low pup survival. The decreased pup survival is generally attributed to human-induced mortality and increased resistance to recovery efforts from ranchers and livestock producers. Genetic threats are also a major concern because the entire reintroduced population, as well as all the animals in captivity, are descendants from seven wild founders (Hedrick and Fredrickson 2008).

The 31 Canadian Gray Wolves introduced in Yellowstone have successfully re-established for several reasons. Abundant prey, large dispersal ranges, legal protection, and plenty of space provided adequate habitat conditions for the population to recover and establish multiple territories within the greater Yellowstone area, despite the ongoing opposition from livestock and cattle farmers (Smith *et al.* 2003).

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Table 2.1. Proportion of parameters quantified in 53 studies, arranged from most to least used in each study.

Parameter Category	Percentage of all studies (n=53) that numerically quantified the parameter				
	81 - 100	**61 - 80**	**41 - 60**	**21 - 40**	**0 - 20**
Sociological					Public behavior
					Public attitude
Population Data			Wolf densities	Pack Size	Mortality
				Dispersal	Fecundity
					Natality
					Sex/age ratio
					Survival rates
Biological		Historical occurrence	Pack territory		Interspecific competition
					Disease
					Effects on scavengers
					Hybridization
Infrastructure	Road density	Human density		Land ownership	
				Land use	
				Livestock abundance	
Biophysical		Land cover	Land area	Topography	Winter severity
		Prey base			Hydrology
					Climate
					Water availability

Table 2.2. Proportion of parameters quantified and qualitatively considered in 53 studies, arranged from most to least used in each study.

Parameter Category	Percentage of all studies (n=53) that numerically quantified, and qualitatively considered the parameter				
	81 - 100	**61 - 80**	**41 - 60**	**21 - 40**	**0 - 20**
Sociological			Public behavior		
			Public attitude		
Population Data			Wolf densities	Survival rates	Natality
			Dispersal	Pack Size	Sex/age ratio
				Mortality	
				Fecundity	
Biological		Pack territory		Hybridization	Interspecific competition
		Historical occurrence			Disease
					Effects on scavengers
Infrastructure	Road density	Land use	Land ownership		
	Human density		Livestock abundance		
Biophysical	Land cover	Prey base	Topography	Climate	Winter severity
			Land area	Water availability	Hydrology

Chapter 3. Systematic habitat modelling and estimation of carrying capacity for Wolves (*Canis lupus* and *C. lycaon*) on Cape Breton Island.

Introduction

Apex carnivores such as Wolves (*Canis sp.)* are habitat generalists, but are nevertheless vulnerable to displacement when exposed to human activity (Mech 1995; Noss *et al*. 1996). Wolves persist in remote areas where they are not significantly persecuted by humans (Mladenoff *et al.* 1995; Treves and Karanth 2003; Benson *et al.* 2017). Biophysical attributes such as agriculture, urban development, human and road density are commonly used as surrogates for identifying the effect of anthropogenic disturbance on Wolf viability and are systematically modelled to predict the suitability of an area for Wolf reintroduction (Noss *et al.* 1996; Carroll 2000; Carroll *et al.* 2003; Chapter 2, Appendix 1). A review of available North American Wolf reintroduction and habitat modelling literature was completed in Chapter 2 to identify the most pertinent variables to indicate suitability of reintroduction sites.

Systematic conservation planning involves the management of entire landscapes wherein areas are allocated for production and protection of biodiversity representative to that region (Mace *et al.* 2000; Margules and Pressey 2000). MARXAN software can be helpful when conflicts of interest arise between biodiversity conservation and human needs (PacMara 2017). As apex carnivores, Wolves are often perceived as direct competitors with humans for resources (Soule *et al*. 2005; Estes *et al*. 2011). Wolves require areas with large ungulate prey; however, these areas must have low human associated mortality risk to ensure the long-term persistence of the population (Fritts and Carbyn 1995). This situation makes their conservation difficult and perhaps not feasible on most human-dominated landscapes.

MARXAN systematically attempts to achieve a minimum representation of conservation or biological features within a reserve network for the lowest possible cost, as specified by the user to isolate it from threat (Possingham *et al.* 2002). The program defines cost by using simulated annealing algorithms and mathematical optimization to find multiple alternative solutions to a variety of

spatial prioritization problems. Simulated annealing provides answers to a wide range of conservation problems (Ardron *et al*. 2008). This algorithm is relatively simple, fast, and adaptable to changes in the type and context of the conservation problem (Ball *et al.* 2009, Ban *et al.* 2010).

Methods

The objective of this Chapter was to use MARXAN to estimate the maximum amount of area Wolves could occupy on Cape Breton Island (CBI) where Moose (*Alces alces*) relative abundance was high, under the constraint of only including land representative of low (0 Risk Weighting) to moderate (0.50 Risk Weighting) human associated mortality risk (Table 3.1).

This tool was freely available at (http://marxan.net) and allowed for linkage with ArcGIS and ArcMap. Suitable and non-habitat sites were identified in the study area. This was a necessary yet essential prelude to approximating the potential carrying capacity of Wolves on suitable and less than optimal reintroduction sites of CBI.

Planning units

The study area was systematically proportioned into 1-km^2 hexagonal planning units, based on the resolution of local data and peer-reviewed studies that used similar spatial methods to assess the availability of favorable habitat (Carroll *et al*. 2001; Gehring & Potter 2005; Larsen and Ripple 2006). The 1-km^2 resolution provided a fine scale identification of optimal and suboptimal habitat sites. A spatial GIS shapefile feature class layer, excluding all inland fresh and salt water bodies > 0.2 km^2, served as the base planning unit layer for subsequent MARXAN analysis. Wolves generally avoid water bodies (Norris *et al*. 2002) and pursuing ungulates on frozen water bodies (Paquet 1992). A systematic approach to identifying sites allows managers to prioritize actions and determine the cost associated with conserving area for Wolves (Rondinini and Boitani 2007; H. Possingham pers. comm. 2017).

Local data and habitat suitability

Four data-sets (identified as pertinent indicators of Wolf reintroduction suitability – Chapter 2, pg. 20) for CBI were downloaded from the province of Nova Scotia's open data portal and forest inventory catalog (NS GIS 2015). These were also the only available land coverage sets for CBI, they include: urban and agriculture areas (land cover-biophysical), as well as human and road densities (socio-economic). Urban and agriculture land coverage were first extracted from the land cover data-set. The biophysical shapefiles were overlaid on the planning unit layer to determine how many planning units contained either urban or agriculture variables. In this analysis, the source layer was identified as the planning unit layer, and the target layer identified as the land coverage variable. The selection method for the land coverage variable was 'intersect the source layer (the planning unit)'. This process allowed the variable to be intersected by the planning unit layer and was repeated for both urban and agriculture land coverage.

Road coverage data for CBI were derived from Digital Mapping Technologies Incorporated's (DMTI) real time location intelligence geospatial database (DMTI Spatial 2016). All roads included: expressways, principle highways, secondary highways, proposed highways, major roads, local roads, proposed roads, and trails were merged as one feature class. Road densities for each planning unit were calculated using Kernel Density and Zonal Statistics as Table tools in Arc-GIS Desktop ver.10.5. The search radius and output cell size of 250 square meters were used for the kernel density tool. Search radius is referenced to the center of each raster cell to assure that roads outside the range of 250 meters would not be included in the density calculations. A 250-m square resolution provided a finer scale resolution of the mean road density across the landscape, compared to other studies which have used 1 km^2 or greater resolution. The zonal statistics as a table tool is used to summarize values within raster data-sets and report results in tables. It was used to summarize the mean road density value of each raster cell within a planning unit and report the results

to an output text file. This text table was then joined to the planning unit shapefile by way of the planning unit unique identifier field (PU_ID).

Human population data were downloaded from the 2011 Canadian census profile on the Statistics Canada website (Statistics Canada 2015). Cape Breton Island census data were extracted from the comprehensive area file within the provincial database. Census data are divided into standard geographic zones, called dissemination areas; they are essentially the smallest geographic units for which census data are collected (Statistics Canada 2015). The average human population density in each dissemination unit was calculated by dividing the total number of people within each dissemination unit (total 273) by the total land area (in km^2) within the unit's boundary (geographic zone layer). The planning unit layer was then intersected with this geographic zone layer shapefile to compute the population density in each unit by way of the PU_ID field.

To scale the extent of human-associated mortality risk to reintroduced Wolves on CBI, each planning unit was assigned a risk weighting based on the summed value for each risk indicator variable. The weighting for each individual variable was between 0 and 1 (H. Possingham, pers. comm. 2017). The lowest and highest weighting for a planning unit was 0 and 4, respectively (Table 3.1). Planning units that intersected either of the land coverage variables, urban or agriculture, were assigned a value of 1 and those not intersecting were assigned a value of 0. Values were connected to the unique identifier field (PU_ID) in the planning unit layer. Variables road and human density were measured at the planning unit scale and assigned weightings based on their respective thresholds; thresholds were determined by review of the 53 studies in Appendix 1 (Table 3.1). The values were linked together based on the unique identifier PU_ID field in the planning unit layer. A risk-layer map, which served as the input cost layer to the MARXAN program, was generated in ArcGIS (Figure 3.2). All planning units with values of 0 were assumed to be highly suitable habitat (optimal – low risk of mortality, due to anthropogenic causes), 0.5 (marginal suitability – moderate risk), and ≥1 as non-habitat (high risk; Table 3.1). If a planning unit was weighted at 1 due to a combination of two of the marginal

socioeconomic variable weights, it was identified as being non-habitat (low suitability).

Territory size and carrying capacity

Planning units with high relative Moose abundance and low or low-moderate human induced risk (Table 3.1) were assumed to be adequate areas for Wolf packs to establish territories. A catch per unit effort (CPUE) analysis or “hunter success rate” was applied in each Moose management unit (Figure 3.1) employed by the Nova Scotia Department of Natural Resources (NSDNR) as an indicator of the Moose relative abundance in each unit. Moose management unit boundaries (5) were downloaded from the NSDNR ArcGIS server. To calculate the CPUE in each unit, the total number of Moose permits issued were pro-rated by the average number of animals harvested in the past 5 years (2010 – 2016; NSDNR 2016). A CPUE in CBHNP was estimated by analyzing hunter harvest success from the 2015 season Moose cull.

The zones were converted to ArcGIS shapefiles and populated with a field containing CPUE %’s as an indicator of relative Moose density (Table 3.2). The area of CBI outside of these zones was labeled as a NULL area (zone 6), and did not contain a CPUE. There were no Moose demographic data available for this area (J Bridgland, pers. comm. 2017).

The average Wolf pack home range (territory) size was estimated from other studies for *Canis lupus* (Table 3.3; approximately 287 km²) and *C. lycaon* (Table 3.4; approximately 225 km²). Because planning units are 1 km² in size, it was assumed that a road or small town could be occupied within this 1 km² area and affect the viability of the population. Therefore, areas representative of pack home range had to be spatially connected (< 1km distance between planning units). To estimate the number of pack territories and carrying capacity (number of Wolves based on amount of available habitat), home range size was overlaid on connected land areas classified as highly suitable (optimal = low risk ‘0’) within the National Park. Home range size was also overlaid on a combination of highly suitable (low risk ‘0’) and marginally suitable (moderate risk ‘0.5’; Figure

3.2) areas in all Moose management units (Figure 3.1). Areas including both highly and marginally suitable planning units outside of the National Park were classified as ‘suboptimal’ habitat.

MARXAN was used to identify lower and moderate risk PU’s (0 and 0.5) within each Moose management unit and the National Park, and if there was enough connectivity to support the establishment of one or more territorial home ranges. MARXAN estimated the maximum amount of available area in each unit, under the constraint of only including land representative of low to moderate human-associated mortality risk (Table 3.1) The ArcGIS polygon measurement tool was used to measure the connected size (in km^2) of habitat patches identified in the analysis. Finally, the number of pack territories and the carrying capacity were estimated for all fragmented habitat that was highly and marginally suitable (0 and 0.5 risk weightings), outside of the National Park including Moose management zones 3 and 4. It was assumed that no packs could be sustained by areas classified as non-habitat (risk weighting of 1 or higher).

Results

Habitat suitability

Areas of CBI identified as highly suitable (optimal), marginal, and non-habitat for both *Canis lupus* and *C. lycaon* are shown in Figure 3.2, with risk ratings of 0 (low), 0.5 (moderate) and ≥1 (high) respectively (Table 3.1). Much of the optimal (low risk) habitat is located within the National Park and in the adjacent southeast area. A small portion of suitable habitat is identified for Richmond County. Cape Breton County and southern Inverness County are identified as high risk areas for Wolves, due to high human and road population density (Figure 3.2). There is no difference in habitat use between *Canis lupus* and *C. lycaon*; both Wolf species can occupy area, where they are not persecuted by humans.

Wolf pack territories and carrying capacity

All spatially connected low risk (0; optimal) and low to moderate risk (0.5; suboptimal) land areas on CBI that is assumed to facilitate the establishment of territorial home ranges is identified in (Figure 3.3). MARXAN identified approximately 750 km² of connected low risk (0) habitat within the National Park (Figure 3.3) and 770 km² of low (0) to moderate risk [0.5; suboptimal – combined low (0) and 0.5 (moderate) risk weighting] habitat within Moose management zone 3. However, when a BLM (boundary length modifier – default setting) function was applied to ensure connectivity among planning units (<1 km² - one planning unit), MARXAN identified just 470 km² of the habitat classified as suboptimal within zone 3 (shown in Figure 3.3) as spatially connected. MARXAN also identified an additional 250 km² of connected suboptimal habitat within zone 4.

This 470 km² area was assumed suboptimal for Wolves to form territorial home ranges because mortality rates are expected to be higher in these zones as they include both low (0) and moderate risk (<0.5) planning units and because they currently do not receive the same legal protection from hunting and trapping as in the National Park. Subsequent *VORTEX* analysis (Chapter 4) allowed for modeling of uncertainty in the expected mortality rates (Table 3.5) of Wolves in this suboptimal habitat (outside of the Park). There is assumed no added human-induced mortality for Wolves within the National Park (Table 3.5).

Since there is apparent variation in the home ranges size of Wolf packs in large prey systems, it was assumed that the average Gray Wolf pack home range (287 km²) could vary +/- 40 km² (Messier 1985) and the average Eastern Wolf pack home range (225 km²) varied within +/- 69 km² (Mills *et al.* 2006). The possible number of packs, and carrying capacity of Wolves was estimated under such variation for both species (Table 3.5 & 3.6) in the National Park and the low-moderate risk (sub-optimal) connected habitat outside of the National Park (identified in Figure 3.3). Estimates for carrying capacity were based on the average pack size of 5.9 (*Canis lupus*) and 5.0 (*C. lycaon*). There is no overall

increase in mortality for Wolves inside the National Park because human activity is low and Wolves receive full legal protection.

Due to the fragmentation (> 1 km² between planning units and < the average area required for a pack to form) of suitable habitat patches and distribution of solitary planning units within Moose management zones 1, 2, 3, 5, and 6 (Figure 3.4), there is not enough spatially connected optimal or suboptimal habitat to allow Wolf packs to establish home ranges without great uncertainty in probable mortality rates. Although some of these zones (e.g., 2, 3, & 5) have a Moose relative abundance > 70%, we cannot be certain if packs will establish. There is an absence of Moose in zone 6 (Table 3.2), and to date there is not a clear explanation to why this is the case (Beazley et al. 2008); other prey items such as White-Tailed Deer (*Odocoileus virginianus*) and smaller mammals (e.g., Beaver; *Castor canadensis*) may become prey within these zones.

The mean number of pack territories and carrying capacity was also estimated for both Wolf species in the fragmented suboptimal habitat (Table 3.7 & 3.8; Figure 3.4) outside of the optimal and suboptimal connected Wolf habitat (Figure 3.3). An average of nine packs could establish for *Canis lupus*, and 12 for *C. lycaon*, but it is unlikely that this number of packs could establish in fragmented habitat surrounded by high risk areas.

Discussion

A Wolf reintroduction is more viable if Wolves are to receive protection from harvesting in areas of Cape Breton Highlands and CBHNP that comprise optimal and suboptimal habitat. If a reintroduction is going to be successful, hunting, and trapping of Canids would need to be prohibited in optimal areas as well as adjacent lands were Wolves could establish territories. Such regulation can be very difficult to achieve on a small land base with several jurisdictions that have different regulations regarding the legal status of the species (Forbes and Theberge 1996, Theberge and Theberge 2004). The amount of low risk area (optimal) identified within the National Park (750 km²) and low – moderate risk (suboptimal) area (720 km²) adjacent to the Park can support several Wolf packs, however if individual Wolves move outside of these areas they will be less likely

to establish packs and persist on the landscape, due to human-associated mortality risk and fragmented habitat patches (Rutledge *et al.* 2010). The suitable area is located mainly in the northeast portion of CBI (Figure 3.3). Because hunting and trapping are allowed in most of CBI outside of the National Park, and protected species can be killed by incidental harvest, the mortality risk will be substantially higher than expected inside the National Park.

Estimating the total area of optimal and suboptimal habitat (1470 km²) was a necessary first step in estimating the potential carrying capacity for the species based on the approximate number of Wolf pack territories that could occupy the island. Previous research (Table 3.3 & 3.4) has estimated the territorial home range size for Wolves using the minimum convex polygon method using 95% of the closest locations of radio-tagged pack members (Bekoff and Mech 1984; Messier 1985). It is assumed that Wolves on CBI will have similarly sized home ranges to those observed when (i) the MCP method was employed, (ii) Moose is the main prey base, and (iii) the land is mainly forested (typically, boreal). However, there are many factors that influence territory size, e.g., the number of Wolves in the pack, inter-specific strife, and prey availability (Packard and Mech 1980). The potential carrying capacity identified from these analyses provided a baseline estimate of population size for subsequent population viability analysis scenarios in Chapter 4. Optimal spatially connected areas, and suboptimal spatially connected areas (Figure 3.3) can hold approximately 30 individuals (16 inside the Park and 14 outside of the Park, respectively or about 5 packs) of *C. lupus* or 33 individuals (17 inside the Park, and 16 outside of the Park, respectively, or 6 packs) of *C. lycaon*. In combination with the suboptimal (low and moderate risk planning units) fragmented area (Figure 3.4), which could contain 55 *C. lupus* or 57 *C. lycaon* individuals, it is feasible that CBI could support 85 individual (14 packs) of *C. lupus* or 90 individuals (18 packs) of *C. lycaon*. However, survival of Wolves in the large suboptimal fragmented area (where human and road densities are high), and the suboptimal connected areas adjacent to the Park is likely to be low (due to anthropogenic risk), and the goal for reducing Moose densities to desired levels would occur only in CBHNP, an

area that can hold approximately 16 Wolves. Furthermore, if Wolves consume prey such as White-tailed Deer in areas where Moose abundance is low, their ability to control Moose in the National Park and the closely surrounding Cape Breton Highlands Ecodistrict (Neily *et al.* 2005) may be limited.

Positive human attitudes towards carnivores play a critical role toward the success of Wolf reintroduction. Sponarski *et al.* 2015 completed a survey of human attitudes towards Eastern Coyote, a significantly smaller canid, among three groups in CBHNP: residents, park staff, and visitors. Results indicated that residents were more fearful of and had less tolerance for Coyotes than the other parties. When people fear an animal, they are less willing to co-exist with it (Bergstrom 2017). Although sufficient habitat may be available to support a viable population of Wolves in CBHNP, residents and other interest groups should have positive attitudes toward carnivores, if not, conflict may occur and pose a challenge to long-term viability of the reintroduced species. The prolonged absence of Wolves in CBHNP, fear of Wolves ingrained in the European psyche, and a fatal Coyote attack on a hiker in 2009 (CBC 2009) is probably why residents are fearful of all canids. If people are willing to attend educational seminars and understand that co-existence is possible, they may reflect more positive attitudes, which in turn will benefit the success of a canid reintroduction (Soulé *et al.* 2005).

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Table 3.1. Risk weightings assigned to biophysical and socio-economic risk indicators on Cape Breton Island.

	Risk Weighting		
Variable	[1]0	[2]0.5	[3]1
Urban Area	no intersect	n/a	if intersect
Agriculture Area	no intersect	n/a	if intersect
Road Density (km/km²)	<= 0.37	> 0.37 & < 0.65	>= 0.65
Human Density (x/km²)	<= 1.55	> 1.55 & < 6.00	>= 6.00
Suitability	high	marginal	low

[1] low risk (No added human mortality to baseline mortality assumed in undisturbed systems)

[2] moderate risk (50% increase in mortality)

[3] high risk (100% increase in mortality)

Table 3.2. Area and Moose catch per unit effort (%) for Moose management units (zones) on Cape Breton Island. The gray highlighting indicates the prey base is likely unable to support Wolf pack formation.

Zone	CPUE (%)	Area (km²)
Park	93	952
1	68	2822
2	75	1100
3	76	1419
4	88	265
5	89	162
6	Not Available	3902

Table 3.3. Average territory, pack size, and pack size variation for *Canis lupus* in a large ungulate prey base system.

Location	Territory size (km²)	*Method	Pack size variation	Pack size	Prey System	Forest type	Reference
Isle Royale	272$\bar{x}$	Mohr 1947-MACP 95%	varied (1976-1986)	6.5$\bar{x}$ (1983-1986[2]E)	moose	boreal	Peterson and Page 1988
Quebec	85-325[1]r, 199$\bar{x}$ +/-16km	95% MCP, Mohr 1947	[3]x	5.6$\bar{x}$ +/- 0.44	moose, ˚WTD	mixedwood	Potvin 1988
Quebec	390$\bar{x}$ +/-40km²	95% MCP	3.7-5.7	5.5$\bar{x}$	moose	boreal & mixedwood	Messier 1985
	$\bar{x}$ = 287			$\bar{x}$ = 5.9			

*method used to estimate territory

[1] range

[2] wolves and moose at equilibrium

[3] no data

˚ white-tailed deer

*Note; Messier's territory estimate is in areas where there is high prey abundance

Table 3.4. Average territory, pack size, and pack size variation for *Canis lycaon* in a large ungulate prey base system.

Location	Territory size (km²)	*Method	Pack size variation	Pack size	Prey System	Forest Type	Reference
Ontario	216-252[1]r, 234$\bar{x}$	95% MCP	4.8 - 6.6	5.7$\bar{x}$	moose,˚WTD	boreal	Forbes and Theberge 1996
Ontario	214 $\bar{x}$	100% MCP	3.3 - 6	4.2$\bar{x}$ ± 0.4 ([2]SE)	moose, WTD	boreal	Patterson *et al.* 2004
Ontario	50-620r (228$\bar{x}$ +/-67km	95% MCP	[3]x	[3]x	moose, WTD	boreal & deciduous	Mills *et al* . 2006
	$\bar{x}$ = 225			$\bar{x}$ = 4.95			

*method used to estimate territory

[1] range

[2] standard error

[3] no data

˚ white-tailed deer

Table 3.5. Literature-based estimate of potential minimum, mean and maximum number of pack territories (km², includes standard deviation (s²)) and carrying capacity (K) for *Canis lupus* on Cape Breton Island in connected low (optimal) and low–moderate (suboptimal) risk areas (Figure 3.3). The [1]percent increase in mortality will be added (modelled) to Wolf packs outside of the Park (Zones 3 and 4) in *VORTEX.* Pack territory and Wolf K are inputs to the VORTEX model. Pack territory size is expressed as a mean +/- 40 km².

Zone	**Approximate Home Range Area (km²)**	**Pack Territories (s²)**			**Wolves (K)**			**[1]Percent Increase in Mortality**
		247	*x̄ (287)*	*327*	Max	° x̄	Min	
[2]Park	750	3	2.6	2.3	17.7	15.3	13.6	0
[3]3	470	1.9	1.6	1.4	11.21	9.4	8.3	10 or 20
[3]4	250	1	0.9	0.8	5.9	5.3	4.7	10 or 20
1,2,5,6	0	-	-	-	-	-	-	-
Total (x̄) >			***5.1***			***30***		

[1]Subsequently modelled using VORTEX, [2]Land classified as 0 risk

[3]Land classified as 0 or 0.5 risk, °(x̄ Pack Territory Size)*(x̄ Pack size (5.9))

Table 3.6. Literature-based estimate of potential minimum, mean and maximum number of pack territories (km², includes standard deviation (s²)) and carrying capacity (K) for *Canis lycaon* on Cape Breton Island in connected (optimal) and low–moderate risk (suboptimal) risk areas (Figure 3.3). The [1]percent increase in mortality will be added (modelled) to Wolf packs outside of the Park (Zones 3 and 4) in *VORTEX.* Pack territory and Wolf K are inputs to the *VORTEX* model. Pack territory size is expressed as a mean +/- 69 km².

Zone	**Approximate Home Range Area (km²)**	**Pack Territories (s²)**			**Wolves (K)**			**[1]Percent Increase in Mortality**
		156	*x̄ (225)*	*294*	Max	° x̄	Min	
[2]Park	750	4.8	3.3	2.6	24	16.5	13	0
[3]3	470	3	2.1	1.6	15	10.5	8	10 or 20
[3]4	250	1.6	1.1	0.9	8	5.5	4.5	10 or 20
1,2,5,6	0	-	-	-	-	-	-	-
Total (x̄) >			***6.5***			***33***		

[1]Subsequently modelled using VORTEX, [2]Land classified as 0 risk

[3]Land classified as 0 or 0.5 risk, °(x̄ Pack Territory Size)*(x̄ Pack size (5.0))

Table 3.7. Literature-based estimate of minimum, mean, and maximum number of pack territories (km²) and carrying capacity (K) for *Canis lupus* on Cape Breton Island in fragmented suboptimal land area [(0 high suitability-low risk) + (0.5 marginal suitability-moderate risk)].

²Approximate Area (km²)	**# Pack Territories if x̄ (287)**	**Total # Wolves °x̄**
2600 if (100%)	9.1	54.7
1950 if (75%)	6.8	40.1
1300 if (50%)	4.6	27.1
650 if (25%)	2.3	13.6

°(x̄ Pack Territory Size)*(x̄ Pack size (5.9))

²Includes Moose management zones 1,2,3,5,6

Table 3.8. Literature-based estimate of minimum, mean, and maximum number of pack territories (km²) and carrying capacity (K) for *Canis lycaon* on Cape Breton Island in fragmented suboptimal land area [(0 high suitability-low risk) + (0.5 marginal suitability-moderate risk)].

²Approximate Available Area (km²)	**# Pack Territories if x̄ (225)**	**Total # Wolves °x̄**
2600 if (100%)	11.6	58
1950 if (75%)	8.7	43.5
1300 if (50%)	5.8	29
650 if (25%)	2.9	14.5

°(x̄ Pack Territory Size)*(x̄ Pack size (5.0))

²Includes Moose management zones 1,2,3,5,6

Figure 3.1. Spatial layout of Moose management units (zones) and Cape Breton Highlands National Park on Cape Breton Island. Solid line in the center of the map indicates the boundary between Zones 2 and 6.

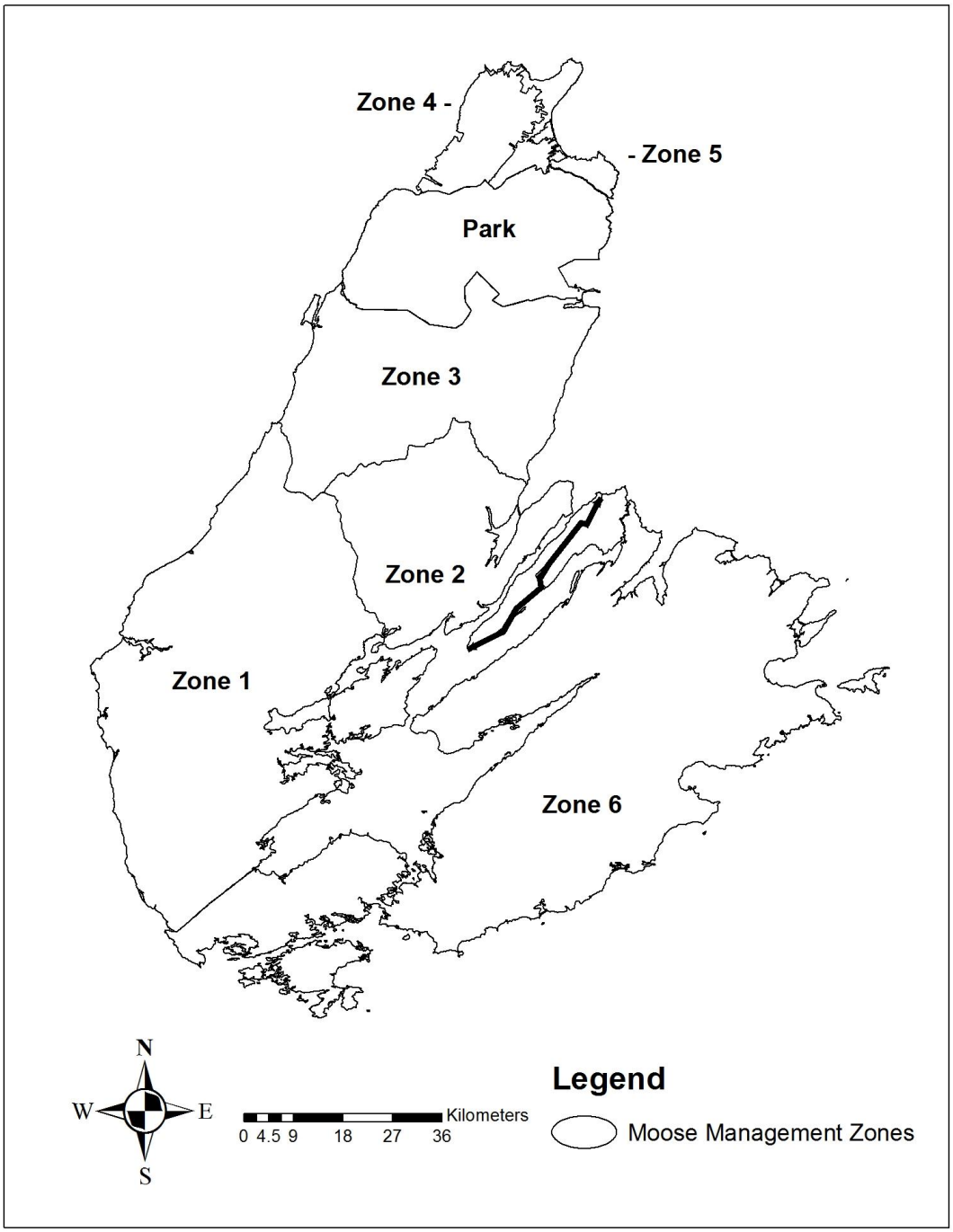


Figure 3.2. Human-induced risk weightings for wolves as a means of suitability for Wolf survival on individual 1-km² planning units on Cape Breton Island. See Table 3.1 for threshold ranges that define human-induced risk weightings.

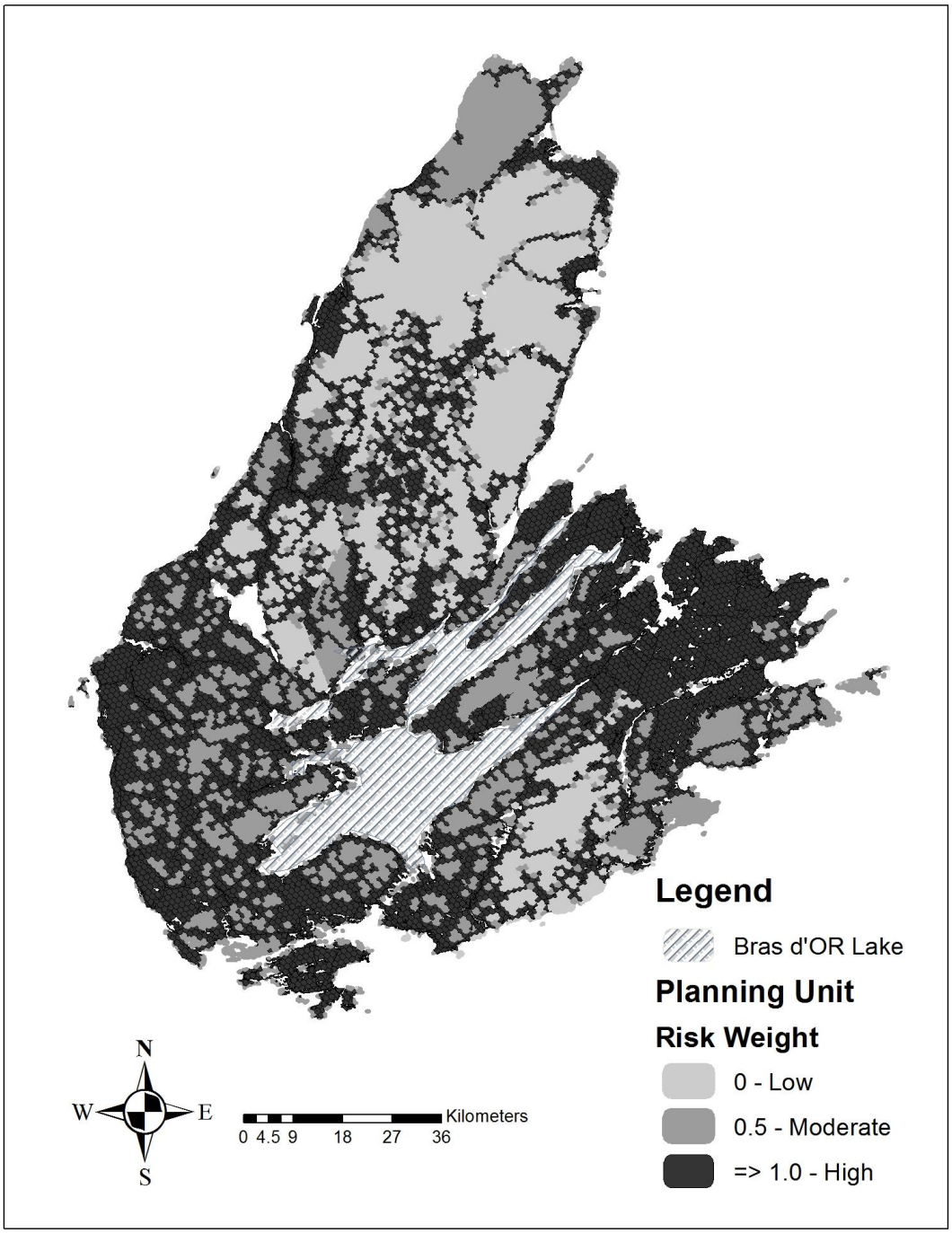


Figure 3.3. Location of connected optimal land area (0-high suitability-low risk) inside the National Park and connected suboptimal land area [(0) + (0.5 marginal suitability-moderate risk)] outside the Park on Cape Breton Island that is assumed adequate for Wolf packs to establish territorial home ranges. Optimal (low risk) habitat is the light grey area within the National Park and suboptimal (low and moderate risk) habitat is the dark gray area in Zones 2 and 3. White area within the zones are “Survival Uncertain Area” and do not fall into either of the classifications listed above. Solid line in the center indicates the boundary between Zones 2 and 6.

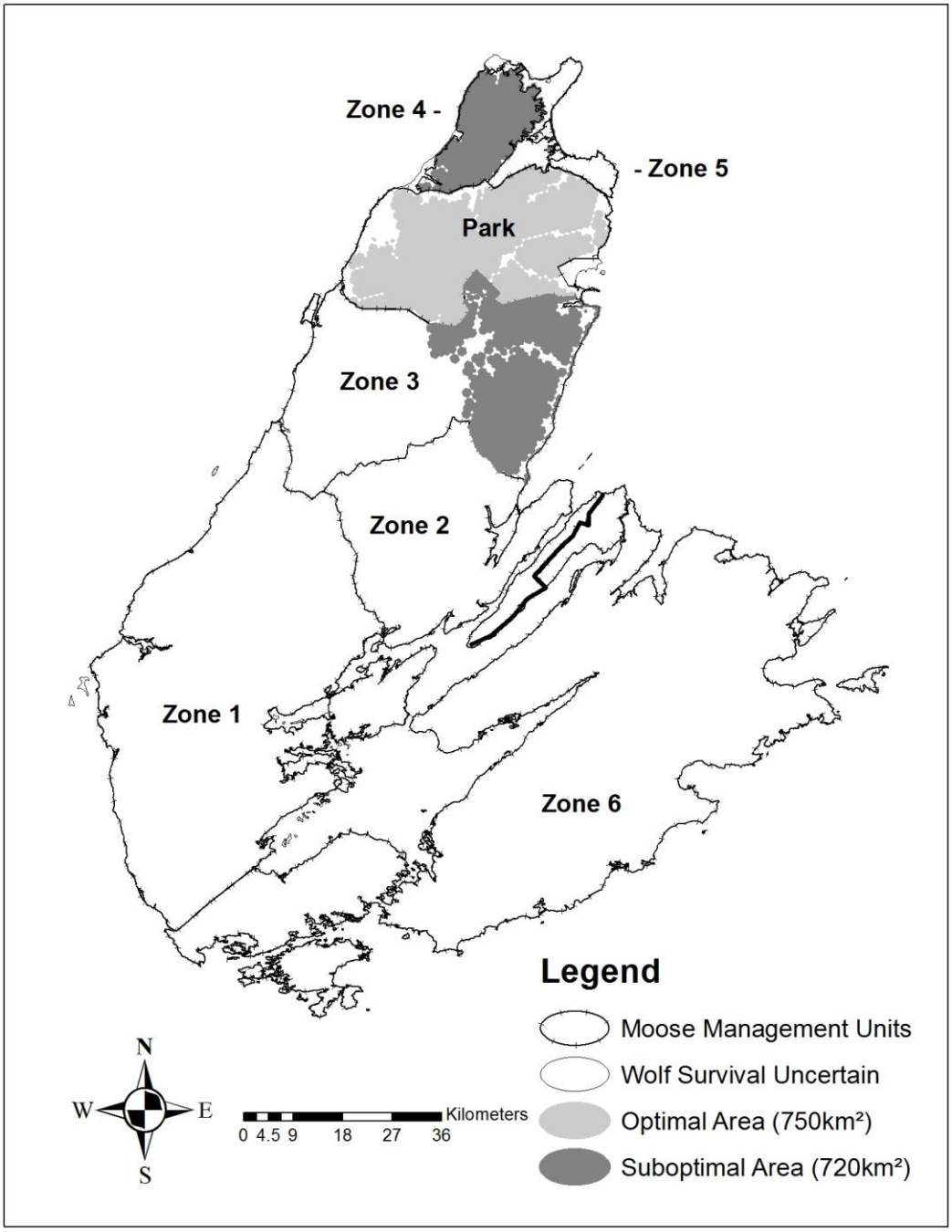
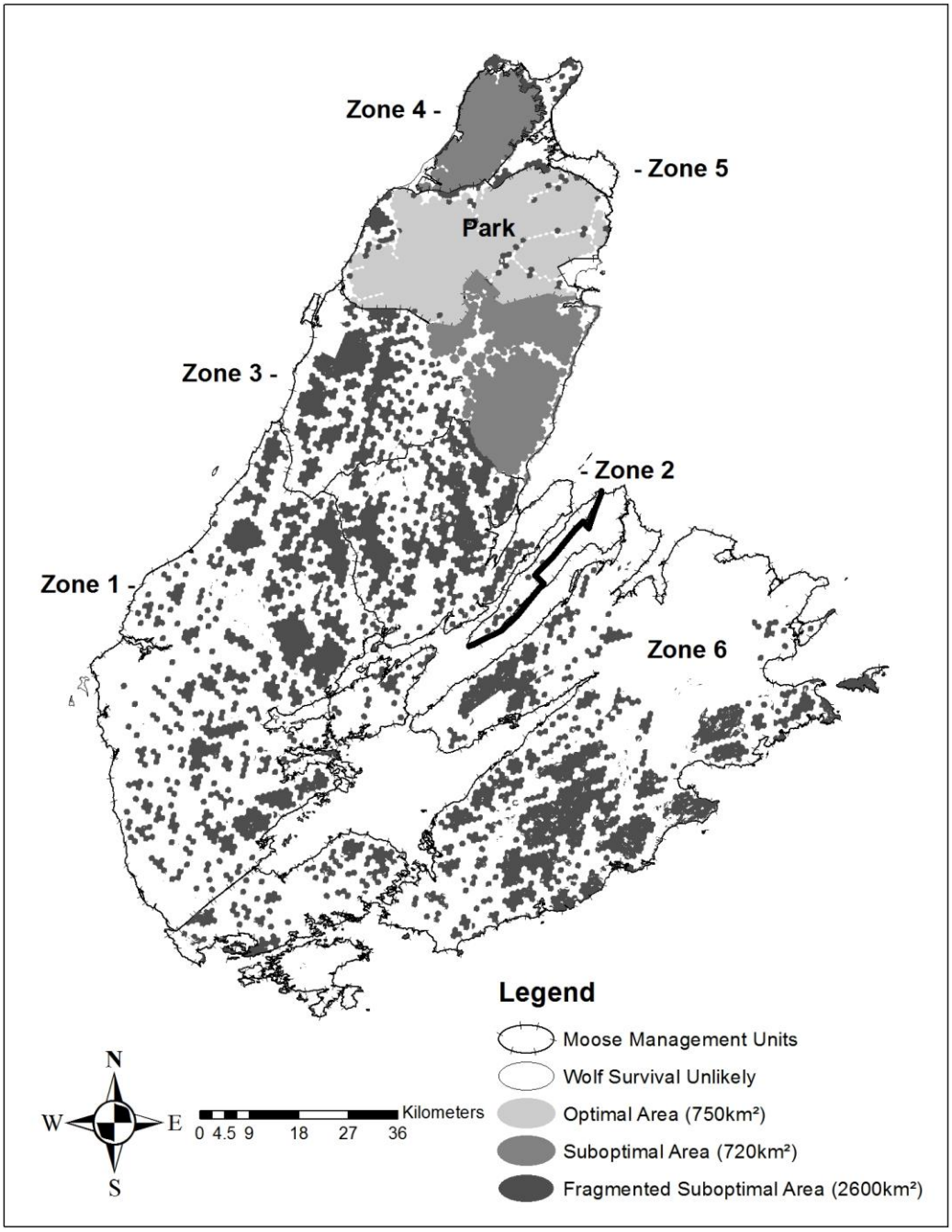


Figure 3.4. Location of fragmented suboptimal [(0-high suitability) + (0.5 marginal suitability)] land area on Cape Breton Island within Moose management zones 1-3, 5-6. Solid line in the center indicates the boundary between Zone 2 and 6.



Chapter 4. Model simulation of viability of re-introduced Wolves (*Canis lupus* and *C. lycaon*) and subsequent impacts on the Moose population on Cape Breton Island.

Introduction

Population viability models are useful in conservation planning as they allow managers to model stochastic threats to which many populations are inevitably subject, and foretell future population sizes under a variety of scenarios (Lacy *et al.* 2013). Population viability analysis (PVA) uses quantitative methods to determine the probability that a population will go extinct, and identifies threats to the population's survival (Akçakaya 2000). A viable Wolf *(Canis lupus* or *C. lycaon)* population requires an area of adequate size, abundant prey, and absence of persecution by humans. Prior to predicting the likelihood that a reintroduced population of Wolves would remain viable in the long term on Cape Breton Island (CBI) it is important to determine if there is sufficient habitat, as well as "optimal areas" through spatial modelling analysis. Optimal habitat areas are locations where Wolves are less likely to be exploited by humans (Mladenoff *et al.* 1995; Leonard *et al*. 2005; Carroll *et al.* 2006). In the previous Chapter, a habitat analysis provided baseline estimate of the hypothetical Wolf carrying capacity on CBI based on an average Wolf pack and territory size. PVA and landscape-level habitat analysis are often used in conjunction to assess the feasibility of a species reintroduction by linking GIS with spatial PVA-modelling procedures (Carroll *et al.* 2003; H Possingham, pers. comm. 2017). The *VORTEX* PVA model was employed to model the long-term viability (over a 100-year time horizon) of the hypothetically reintroduced population of Wolves on CBI. *VORTEX* has been previously applied to model the extinction probability of several threatened Wolf populations worldwide (Ewins *et al.* 2000; Theberge and Theberge 2004; Carroll *et al.* 2014; Vonholdt *et al.* 2008; Bruford 2015).

The intent of this study is to provide managers with options on how Wolf reintroduction may be beneficial to restoring forest ecosystems that have been significantly affected by overabundant ungulates. The feasibility of using the Wolf as a management option to control a hyperabundant Moose (*Alces alces*)

population was explored. If establishment of a viable Wolf population is predicted to be non-achievable, the initial goal of the reintroduction may be unattainable. Candidate sites for Wolf reintroductions are often areas where they have been extirpated, however their absence from the landscape for a long time reduces public acceptance of their reintroduction (Lohr and Ballard 1996). Medium-sized predators, such as the Eastern Coyote, often attempt to fill the vacant niche when Wolves are absent. However, Eastern Coyotes exert less predation pressures on ungulates than Wolves, and therefore cannot fully fill the ecological role of top predators (Messier *et al.* 1986; Benson *et al.* 2017).

Areas where the absence (extirpation) of Wolves has resulted in unanticipated ecological impacts, loss of biodiversity, and ungulates negatively affecting the welfare of other populations warrants consideration for reintroduction (Estes *et al.* 2011). When exploring the feasibility of an area for reintroduction of top carnivores such as Wolves, conservation managers are often faced with questions: Will the population be viable? What are the limiting factors to establishing a viable population? Will the objective of the reintroduction be attained? To best answer these questions, a series of *VORTEX* simulations were generated to identify which inputs (demographic and biophysical) affected the probability that the population would persist, and under what circumstances population viability can be achieved. If the population has a high probability of going extinct post-reintroduction, the reintroduction may not be feasible, and alternative methods should be explored to meet the primary goal (i.e., reduce Moose densities). I also determined whether the viable population size was large enough to limit Moose abundance to the desired density of 0.5 Moose/km^2 (Smith *et al*. 2015) within the Cape Breton Highlands National Park (CBHNP) on CBI.

Methods

Life history data

A literature review was conducted to identify input parameters for *VORTEX*. Life history parameters, including reproductive system type (e.g., polygamous, monogamous), age of first reproduction, maximum age of

reproduction, life span, number of litters, number of progeny per litter, sex ratio, density dependent reproduction, percent adult males and females in the breeding pool, and mortality rates were extrapolated from peer-reviewed biological and demographic Wolf studies for Gray (*Canis lupus*) and Eastern Wolf (*C. lycaon*; Table 4.1, 4.2).

Parameter values were based on studies in a similar geographical area (forested and flat topography), when available. The baseline mortality rates inside the Park were those expected in undisturbed systems, where Wolves would not be exposed to any direct risk from human-associated factors (Chapter 2). The carrying capacity for both species was based on the habitat area requirements to establish territorial home ranges and pack size (see Tables 3.5, 3.6 in Chapter 3).

The *VORTEX* model

The *VORTEX* PVA version 10.2.6 was used for all simulations. *VORTEX* is a Monte Carlo computer simulation model of the extinction process that evaluates the effects of deterministic forces, as well as demographic, environmental, and genetic stochastic events that affect populations of wildlife (Lacy and Pollak 2016; Lacy *et al*. 2013). *VORTEX* models extinction vortices that threaten the persistence of small populations, and allows manipulation and tracking of quantified demographics (e.g., founder population size, extinction probability, effective immigrants), and genetics (heterozygosity, allelic diversity, and inbreeding depression; Carroll *et al.* 2014; Bruford 2015). The model also allows the user to specify the pedigree of the founder population and account for genetic differentiation between populations

VORTEX has been used to simulate population dynamics of a range of mammalian carnivore species (Lee *et al.* 2001; Haines *et al.* 2005; Faust *et al.* 2016). *VORTEX* simulates population dynamics by evaluating the annual life cycle of sexually reproducing organisms by tracking mate selection, reproduction, mortality, age increment of individuals, as well as migration among populations, removals, and supplementation. For a more detailed description of PVA using

VORTEX, and how it accommodates stochastic processes see Lacy and Pollak's (2016) *VORTEX* 10 User's manual.

Creating a Wolf population viability model for Cape Breton Island

VORTEX 10.2.6 was used to create a two-population model, in which one population was assigned 'Inside the Park' and the other 'Outside the Park'. The model was built on the assumption that the habitat could support a fixed number of reproducing packs on CBI. The number of packs in both populations varied depending on the mean home range size, including the standard deviation of home range size and the availability of habitat inside and outside of the Park (Figure 3.3 in Chapter 3). The model was more realistic than a typical density dependent function because it is fitted to the actual amount of optimal and suboptimal habitat identified on CBI (Chapter 3); hence, reproduction was limited by available space and Wolf pack dynamics. The model was calibrated to limit the number of females that could breed, based on the number of packs supported by the habitat, rather than a fixed probability of each female breeding each year (R. Lacy, pers. comm. 2017).

To build this model, population state variables were created to provide a description of the characteristics of the population. Each pack was assigned one reproductive unit (Packard and Mech 1980). To determine how many packs can form inside and outside of the Park, a population state variable "PSI" – label: PACKS was created. The initialization function within this variable represented the maximum number of packs that could form (specified by the user). The initialization function defined the starting value for each population in the simulation. Population state variables ("PS2" – label: AVAILF and "PS3" – label: BRF) were also used to tally the number of adult females that are breeders (BRF), as well as the number of available breeding females that are not breeders (AVAILF).

Population state variables were then combined with individual state variables to track the properties of individual Wolves. Individual state variables are commonly used to modify properties such as mortality rates, dispersal rates,

and breeding rates of individual animals. The first individual state variable created was labelled “IS1” – label: BRFEM (breeding female). The initialization function for this variable: ((=(S=’F’) * (A>1) * (A<11) * (RAND<(PACKS/F))) specified that at the start of the simulation if the Wolf is a female, and age is > 1, they have a 50:50 chance of being initially assigned as a breeding female. If they became a breeder (BRFEM = 1) they create PACKS number of breeders, when space was available. All other sub-adult females, and non-breeding adult females were assigned BRFEM = 0. The second individual state variable was called “IS2” – label: AVAILFEM. This variable represented the adult females that were not breeders. The initialization function was: ((S=’F’) * (A>1) * (A<11) * (BRFEM=0)).

The transition function for individual state variable BRFEM was: (=ISI * (A<11) + (ISI=0) *(S=’F’) * (A<11) * (A>1) * (RAND<((PACKS-BRF) / (MAX (1; AVAILF)))). Transition functions are used to track changes in the population for each year of the simulation. This specified that if a Wolf is a breeding female, the individual remains a breeding female until post-reproductive status (age 11 *C. lupus*, 10 *C. lycaon*; Mech 1988; Theberge *et al.* 2006). This function also calculated the number of empty spaces available for Wolves to disperse into and form new packs by inventorying the number of current packs. If space became available (due to a female reaching the age of reproductive senescence or a mortality event occurred and the female expired) the function picked a new female breeder(s) from the AVAILF pool to form a pack(s). The model was set so that Wolves would preferentially form packs inside the Park first (due to high Moose relative abundance) if there was available space both outside and inside the Park. This typically occurs when there is a low proportion of females that breed in any given year. The transition function for the individual state variable AVAILFEM remained the same as the initialization function because there is no change in the *status* of AVAILFEM through time.

The model specified that individual Wolves would migrate outside of the Park if all available habitat become occupied; dispersal at a 10% rate from Park to outside when the population exceeded 90% of maximum K. Wolves would migrate into the Park if the Park population dropped below 90% K. The dispersal

modifier prevented female breeding Wolves from dispersing into or out of the Park. Both packs inside and outside the Park had the same state variables assigned, and this allowed the pack size to fluctuate dependent on carrying capacity. This model represents that the habitat is limiting in terms of how many packs can be supported and that only a specified number of females can breed (R. Lacy, pers. comm. 2017). The two-population packs model was employed with demographic data for both *Canis lupus* (Table 4.1) and *C. lycaon* (Table 4.2). A detailed description of how demographic parameters (species description, dispersal, reproductive system, reproductive rates, mortality rates, and mate monopolization) are entered in the scenario settings is available in the *VORTEX* 10 User's Manual (Lacy and Pollak 2016).

Scenarios

Six scenarios were simulated with the two-population model. The scenarios are parameterized in the same manner as the baseline. However, demographic inputs are varied to test their sensitivity with respect to the population's probability of extinction (Pr. Extinction). The probability of extinction was reported for the entire population for each scenario. All scenarios were reported for *Canis lupus* and C. *lycaon*. Scenarios 1 and 5 can be considered as management scenarios, where humans can directly control the number of Wolves in the population. The others are considered for sensitivity analysis, which tests the model sensitivity to certain parameters or environmental condition.

Baseline Simulation: (Table 4.3 & 4.4). The baseline scenario was populated with standard life history (demographic) parameters from the literature (Tables 4.1 & 4.2). Mortality rates for the population inside the Park were as specified in Tables 1 and 2 (undisturbed system). The population outside of the Park had a 10% increase in mortality (adults and pups) due to human-induced mortality risk in suboptimal habitat (Table 3.5 & 3.6 in Chapter 3). The default value for lethal equivalents was 6.29; Wolves are inevitably subject to inbreeding in small

populations (Peterson *et al.* 1998, Peterson 1999), therefore, it was appropriate to leave the lethal equivalents value at the default setting. Carrying capacity for both 'populations' was based on the number of Wolves [*C. lupus* – 16 individuals in the Park (3 packs), 14 individuals outside of the Park (2 packs); *C. lycaon* – 17 individuals in the Park (3 packs)], 16 individuals outside of the Park (3 packs) that could be supported by the average pack territory size (Table 3.5 and 3.6 - Chapter 3) inside (N=3 for C. *lupus and C. lycaon*) and outside of the Park (N=2 for C. *lupus* and N=3 for C. *lycaon*) on optimal and suboptimal habitat. The habitat could support 6.5 *C. lycaon* packs, however I parameterized the model with 6 packs for the baseline scenario. The initial population size inside and outside the Park was 5 packs each, for both species. Proportional values for the age class distribution were used to accommodate variation in pack structure. This baseline scenario involved one simulation for each species.

<u>Scenario 1: Influence of Changing the Initial Population Size:</u> The initial population size (inside and outside of the Park) was increased to 7 founder animals (simulation 1), and then 9 founder animals (simulation 2).

<u>Scenario 2: Influence of Changing % Adult Females Breeding:</u> For *C. lupus* the proportion of breeding females was increased from baseline of 35 to 55% in simulation 1 and to 75% in simulation 2. For *C. lycaon* the baseline proportion of 57% of females breeding was decreased to 35% in simulation 1 and increased to 75% in simulation 2.

<u>Scenario 3: Influence of Changing Mortality Rates:</u> Note the baseline scenario: (Baseline in Park = 30% pups, 10% adults, Baseline outside Park = 40% pups, 20% adults). For simulation 1 the mortality rates outside the Park had a 10% increase for pups and adults and then a 20% increase for simulation 2. These simulations were done to test the increased human induced mortality rates expected outside of the Park (Tables 3.5 & 3.6 - Chapter 3). Simulation 3 included a 15% decrease in the mortality rate for pups and 5% for adults in and

outside of the Park. Simulation 4 included a 15% decrease in the mortality rate for pups and 5% for adults inside the Park and a 25% decrease in the mortality rate for pups and 15% for adults outside the Park. A fifth simulation was done for *C. lycaon* where mortality inside the Park increased by 5% for pups and adults and mortality rates outside the Park was set as the baseline. This added mortality represents a likely scenario of C. *lycaon* Wolves leaving the Park in search of White-Tailed Deer (Forbes and Theberge 1996). Simulation 2 was not reported for *C. lupus* because the population had a Pr. Extinction of 100% in Simulation 1.

Scenario 4: Influence of Varying Carrying Capacity: In this scenario, I am assuming more packs can be supported inside and outside of the Park. This scenario was tested by varying the initialization function in the state variables (# packs) and K for both populations inside and outside the Park. The baseline model for (C. *lupus*; 3 packs inside the Park, 2 packs outside the Park K = 30) was tested with an additional 1 pack inside the Park (K =36; Simulation 1), 2 packs inside the Park, 1 outside the Park (K =48; Simulation 2), and 3 packs inside the Park, 2 outside the Park (K =60; Simulation 3). The baseline model for (C. *lycaon*; 3 packs inside the Park, 3 packs outside the Park K = 33) was tested with an additional 1 pack inside the Park (K =38; Simulation 1), 2 packs inside the Park, 1 pack outside the Park (K =48; Simulation 2), and 3 packs inside the Park, 2 outside the Park (K =58; Simulation 3).

Scenario 5: Influence of Supplementation: Three Wolves (2 females, 1 male) were added the second year after reintroduction, and then 2 females, 1 male were added every fifth year for the next 20 years (Simulation 1). Three Wolves (2 females, 1 male) were added the second year after reintroduction, and then 2 females, 1 male were added every fifth year for the next 50 years (Simulation 2). Six Wolves (4 females, 2 males) were added the second year after reintroduction, and then 4 females, 2 males were added every fifth year for the next 20 years (Simulation 3). Six Wolves (4 females, 2 males) were added the second year after reintroduction, and then 4 females, 2 males were added every

fifth year for the next 50 years (Simulation 4). All supplementations were released within the National Park.

Scenario 6 (*Canis lupus* Only): Requirements of a Viable Population: After examining the results of the previous scenarios and simulations, and identifying which inputs most affect the population's probability of extinction, several "optimal simulation(s)" were created to reflect what is required for a viable population on CBI. This was modelled for *C. lupus* only because it is not agreed upon in the literature if Eastern Wolves are effective predators of Moose. Eastern Wolves are also likely to occupy areas where White-Tailed Deer are abundant, and are therefore unlikely to use Moose as their main prey (Scenario 3 – Simulation 5; Forbes and Therberge 1996). Several critical model inputs (% of females in breeding pool, mortality rates, and carrying capacity) were modified to ensure the population had a low probability of extinction over a 100-year time horizon. These inputs were the most sensitive to changes in the probability of extinction (See Results – *Canis lupus*). The results of this scenario were expected to be substantially different from the baseline. This scenario was employed for *C. lupus*, because we were interested in whether the viable population size was large enough to reduce the densities of a hyperabundant Moose population, as *C. lupus* is the more efficient predator of Moose (Fuller 1989).

Assessing the impact on Moose

An estimate of the current Moose density within the National Park was required in order to determine if a viable Wolf population could limit Moose densities to 0.5 Moose/km^2 (Table 4.5). After determining the current density (2.3 Moose/km^2), the Wolf kill rate (per 100 Wolf days) for Gray Wolves was estimated by inputting the Moose density for the National Park into a functional response curve (Figure 4.1; Messier 1994). Messier (1994) estimated the functional response in an area where Gray Wolves feed primarily on Moose. By determining the kill rate/100-Wolf days, the number of Moose killed / year / Wolf

was extrapolated. This analysis can be considered a ‘yearlong harvest’. The number of Moose killed was then determined for the average Wolf population size over a 50-year time horizon for simulations 1 and 3 (Table 4.14 – Chapter 4) and simulations 4 and 6 (Table 4.15): Scenario 6 - Requirements of a viable population to predict the subsequent reduction in Moose population size under several scenarios reflective of a viable population. Because a functional response curve of Wolves to changing Moose density was only available for Gray Wolves, the number of Moose that could be killed by Eastern Wolf could not be estimated.

However, because this model just provides a static prediction, a simple deterministic model was then used to estimate the impact a hypothetical Wolf population inside the National Park would have on the Moose population, as well as a population that occupies area both inside and outside of the National Park. This was reported for a ten-year period (Table 4.9 & 4.10). The model was built in Microsoft excel and was tested with 16 Wolves in optimal habitat (inside the National Park), and 30 Wolves in optimal and suboptimal habitat (inside and outside the National Park). The number of Wolves were derived from the baseline scenario (Table 4.3 – Chapter 4). The Moose population had a growth of 0.1 per year, and the Wolf kill rate was defined according to Messier (1994).

Results

Two-population viability model (*C. lupus*)

Scenarios

The following scenarios suggest that the Wolf population is most sensitive to changes in the average percentage of adult females that breed each year, mortality rates, and carrying capacity. Catastrophes caused variations in the population size when they occurred, but were not a limiting factor for extinction. Results are displayed for the Wolf population in all scenarios after the baseline scenario. The “two-population” factor in the model accounts for tracking the survival of individuals who emigrate outside of Park boundaries, and therefore, it is more appropriate to examine the results for both populations inside and

outside of the Park (meta-population). The meta-population represents the average population size over a 100-year time horizon in *VORTEX*.

Baseline scenario (Simulation 1)

The population is very unstable for several reasons. There is a low number of reproductive females that breed in any given year (35% - Mech *et al*. 2016) and mortality rates outside the Park are high (Table 4.3), causing the population outside of the Park to become extinct (e.g., with no reproductive unit) by year 25 since introduction. The population fluctuates and sometimes reaches a high of 20 animals in the first 10 years, but has a 100% probability of extinction over 100 years (Figure 4.2). The percentage of females breeding is a subjective value; it can vary below or above the set value of the baseline simulation depending on environmental and biological conditions.

Scenario 1 (Increase initial population size)

Increasing the initial population size from 10 to 18 individuals had almost no effect (0.02% decrease) in the probability of extinction (Figure 4.3).

Scenario 2 (Increase % adult females breeding)

Increasing the percentage of adult females that breed in any given year had the most significant effect on reducing the population's probability of extinction; Pr. Extinction = 0.33, when 75% of adult females were breeding (Figure 4.4).

Scenario 3 (Vary mortality rates)

Decreasing the mortality rates had the second most significant effect on reducing the population's probability of extinction; reducing Pr. Extinction to 0.71 for the simulation with the lowest mortality. However, when the mortality rates were increased by just 10%, the population had a 100% probability of going extinct by year 35 (Figure 4.5).

Scenario 4 (Vary carrying capacity)

Increasing the carrying capacity helped to offset the probability of extinction, however the population had to be at least doubled to have a 33% reduction in the probability of extinction. It is probably not realistic that CBI can support twice as many animals as estimated. Adding one pack only marginally reduced the probability of extinction; however, when 3 packs were added the probability of extinction was reduced by 25% (Figure 4.6). By changing carrying capacity (K), the model allowed more females to breed. If the population got larger, only the specified number of females could continue to breed. When the population decreased, the pack size number declined but the same number of female breeders were retained.

Scenario 5 (Supplementation)

Supplementation does not appear to offset the long-term probability of extinction. Increasing the number of Wolves supplemented every 5 years (3 vs 6) only marginally reduced the probability of extinction (0.95 vs 0.92; 0.70 vs 0.66; Figure 4.7). Continual supplementation to the baseline over 50 years is required to ensure the population does not have a probability of extinction > 65%. This is highly optimistic in a management setting.

Two-population viability model (*C. lycaon*)

Scenarios

Subsequent modifications of the baseline scenario (see Simulations *IN* Scenarios 1-5) resulted in a more pronounced offset of the probability of extinction for *C. lycaon*, for example, when the mortality rates are decreased (mortality, Simulation #4), the Pr. Extinction = 0.22, versus 0.71 for *C. lupus* in the same mortality reduction simulation. This result is mainly attributable to the population having a much higher initial assigned percentage of adult females that can breed in any given year.

Baseline scenario (Simulation 1)

The baseline scenario for *C. lycaon* included 57% of females breeding and had a higher carrying capacity than the population of *C. lupus* (33 vs 30 Wolves), therefore the probability of the Wolf population going extinct was 20% lower (Pr. Extinction = 0.80; Figure 4.8). However, the maximum number of progeny per litter was smaller (8 vs. 11), and the percent of adult males in the breeding pool was lower (80 vs. 100).

Scenario 1 (Increase initial population size)

Increasing the initial population size from 10 to 18 individuals had a 0.32% decrease in the probability of extinction (Figure 4.9).

Scenario 2 (Increase % adult females breeding)

To increase the percentage of adult females that breed in any given year caused the population's probability of extinction; Pr. Extinction = 0.42, when 75% of adult females were breeding (Figure 4.10).

Scenario 3 (Vary mortality rates)

Simulation 5 (increase the mortality of Park Wolves) represents the probability of extinction when Park Wolves leave the protected areas to search for Deer. Just a 5% increase in mortality for both adults and pup Park Wolves resulted in almost immediate extinction of this population (Pr. Extinction = 0.96). It does not appear that a small population of Wolves can recover from human-caused mortality when the Wolves leave the Park.

Scenario 4 (Vary carrying capacity)

Increasing the carrying capacity to 58 animals resulted in lowering the probability of extinction to only 13%. When the carrying capacity was increased by just 5 animals, the probability of extinction was reduced by 26% (Figure 4.12).

Scenario 5 (Supplementation) When the population was supplemented with 5 Wolves every 5 years for 51 years after an initial supplementation at year 2, the probability of extinction was reduced to 0.43%; when the population was supplemented for only 21 years, the probability of extinction was 0.66%.

Scenarios: Requirements for a viable population for *Canis lupus*

Several scenarios (6) were generated after examining the results from scenarios 1-5 for *C. lupus* (see above). The inputs that had the most effect of changing the probability of extinction were: (i) percentage of adult females breeding (increasing Pr. Extinction), (ii) carrying capacity (increasing), and (iii) mortality rates (decreasing outside the Park, whereas increasing in the Park). These scenarios combined settings for the inputs identified as having the most effect on influencing the extinction risk to simulate a population of *C. lupus* that have a much lower probability of extinction than the baseline scenario. The main reason the baseline scenario has a Pr. Extinction = 1 is due to the low percentage of adult female breeders in the breeding pool.

In the first three simulations (1-3), the percentage of female breeders ranged from 45 to 55%, the mortality rate outside the Park was lowered to match the baseline mortality inside the Park, and K was increased by 6 (1 pack), N = 36 (Table 4.6, Figure 4.14). Because uncertainly exists in the estimated percentage of adult females that breed in any given year, the variable was modified in each set of simulations. I consider the following simulations (1-6) to be reflective of a reintroduced population of Wolves to CBI that receives 100% legal protection from humans.

When 55 % of the adult females were breeding in any given year (Simulation 3-Figure 4.14), and the carrying capacity was set at 36 Wolves, the probability of the population going extinct was low (Pr. Extinction = 0.21). The set mortality rates in these simulations were assumed to reflect a population that is not harassed, shot, or trapped by humans.

In the following three simulations (4-6), the percentage of female breeders and mortality was the same as set in the previous simulations, however the

baseline carrying capacity was increased by 18 (3 packs), N = 48 (Table 4.7 & Figure 4.15). These simulations may be slightly optimistic, because additional space is required for packs to form outside the Park. If an additional three packs form, the probability of extinction is further decreased (Figure 4.15).

Assessing the impact on Moose

The number of Moose killed per year by the Wolf population is shown for simulations 1 & 3 - (Figure 4.14) and 4 & 6 - (Figure 4.15; also see Table 4.8). The current Moose population in CBHNP is approximately 1,930 animals (2.3 Moose/km²; Table 4.5; Parks Canada unpub. data). Data on fecundity and mortality by age cohort do not exist and it is not possible to model population projections other than by using a very coarse kill rate (Smith *et al.* 2015; R. Smith, pers comm. 2015). To reduce the density to 0.5 Moose/km^2, approximately 1,510 Moose would have to be removed from the current population of 1,930 Moose in the Park. This reduction would require 148 Wolves at a kill rate of 2.8 Moose killed per Wolf per 100 days, or 10.2 Moose/Wolf/year (Messier 1994). A rate of 2.8 Moose killed per Wolf per 100 days is the expected kill rate when the Moose density is 2.3 Moose per km² (Messier 1994). It is unfeasible for the Park to support 148 Wolves because space for approximately 25 Wolf packs would be required and it is estimated that the Park can only sustain 3 to 5 packs. Scenario 6 implies the Moose population would reduce to a density of 1.9 Moose/km^2 (Table 4.8), but this density is still much too high than is desired by Parks Canada.

Based on the deterministic model calculations, a population of 30 Wolves could reduce the Moose population to 0.5 Moose/km² in 10 years (Table 4.9); however, a population of 16 Wolves inside the National Park would not reduce the density. At a growth rate of 0.1 the Moose population would increase to 2,158 by year 10 when subject to predation by the smaller Wolf population, as well the density would increase to 2.3 Moose/km² (Table 4.10).

Discussion

It is probable that more Wolf packs could become established in the Cape Breton Highlands given the high density of ungulate prey within the National Park and adjacent areas. Pack size is purported to be smaller in areas where ungulate prey is plentiful (Packard and Mech 1980; Messier 1985). In addition, the recorded variability of pack size as environmental conditions change can affect the number of packs than can potentially form in an area (Harrington *et al.* 1983; Fuller *et al*. 2003). Assuming an additional pack will form within the National Park (Simulations 1-3; Figure 4.14) is not unreasonable. Wolf pack territory sizes as small as 109 km² have been reported in areas where Moose constitute as the main ungulate prey (Peterson and Page 1988). Furthermore, if human attitudes are positive with respect to Wolf reintroduction on CBI, human-caused mortality will presumably be low in areas adjacent to the National Park, thereby allowing Wolves to establish pack territories. A larger number of packs in a small Wolf population can positively influence the reproductive rate (Mech 1995), therefore, population turnover rates are increased. If eight packs establish on Cape Breton Highlands, (modelled in Simulations 4 – 6), the population can remain viable over the long term (100 years) with as low as 50% of the adult females breeding each year.

The percentage of adult females breeding is arguably the most important limiting ecological factor affecting the probability of extinction in the modelled population. The population has a high probability of extinction when relatively few females are in the breeding pool in any given year. When the percentage of female breeders in the population is low, recruitment of available females that are not currently breeders to replace a recently deceased female breeder is delayed. This situation causes a decrease in the potential number of packs that can form, even if available space for Wolves to occupy is available. Mech *et al*. (2016) reported 35-36% of adult females breeding in the Superior National Forest (SNP), Minnesota, but that up to 58% of adult females would breed in any given year throughout Minnesota Wolf-range. Carroll *et al.* (2014) used an estimate of 50% adult females in the breeding pool for the baseline scenario in a PVA that

analyzed the extinction probability of the endangered Mexican Gray Wolf population. Therefore, it is likely that the percentage of adult females in the breeding pool could range from 35 to 55% or higher on CBI. Under some environmental conditions, the percentage of adult females in the breeding pool of a Wolf population have been reported to be as high as 80% (Mech *et al.* 2016). However, assuming that the percentage of females in the breeding pool of a Wolf population is > 60% may be optimistic (Carroll *et al.* 2014).

If the percentage of females in the breeding pool declines below 50 with a K of 36, a larger population size is required (modelled in Simulations 4-6) to offset the probability of extinction caused by fewer adult females breeding. However, to reach a higher population size, more space outside the Park is required and humans must be willing to share the landscape with Wolves. When the percentage of females in the breeding pool is 55 or higher, the population remains relatively stable when a carrying capacity is set at 36 animals. To ensure the percentage of female breeders is 55 or higher in a small population, Wolves cannot be exploited near park borders. Human exploitation adjacent to parks can influence the natural social structure and population regulation mechanisms within protected areas (Rutledge *et al.* 2010; Mech *et al.* 2016).

Although subject to much criticism (see Patterson and Murray 2008), the predictions from PVA models are thought to be sufficiently accurate to predict extinction probabilities when parameter data are extensive and reliable (Brook *et al*. 2000) and the purpose to which the PVA is being applied for is well reasoned (Coulson *et al.* 2001). Uncertainty in the accuracy and precision of results generated by PVA can be attributed to four dominant factors; (i) poor data, (ii) difficulties in parameter estimation, (iii) weak ability to validate models, and (iv) effects of alternate model structures (Beissinger and Westphal 1998; Brook *et al.* 2000). It may take several years to adequately gather population biology data, and the costs of long-term studies is often high (Beissinger 2002). Demographic parameters such as the percentage of females that will breed in a Wolf population each year, or litter size can be difficult to estimate, and requires extensive long-term research (Mech *et al.* 2016). Short-term study periods are

subject to sampling error; therefore, long-term studies are required to accurately estimate parameters. PVA-models are often difficult to validate because stochastic process cannot be precisely estimated, and there is often high uncertainty in the accuracy of life-history parameters used to populate models (Beissinger and Westphal 1998). The outcome from PVA analysis should not be taken as definitive. PVA modelling provides an estimation of the probability of extinction over time based on the respective model settings.

There has been considerable discussion among researchers as to whether Wolves control the density of their prey or if they substitute for another source of mortality. Under certain conditions Wolves can dramatically reduce ungulate populations (Mech and Karns 1977) and sometimes they merely compensate for the mortality that would have otherwise occurred if Wolves were not present (Ballard *et al.* 1987). Under most circumstances, Wolves co-exist with their prey by removing less fit individuals, thus allowing the secure segment of prime-age animals to survive and breed (Wilmers *et al.* 2003). The ability of Wolves to effectively control prey numbers varies per multiple factors (Eberhardt *et al.* 2003), however Wolves have been shown to exhibit top-down controlling forces in some cases (McLaren and Peterson 1994; Estes *et al.* 2011). Top down control causes changes in ungulate density at one trophic level caused by opposite changes at a higher tropic level. This mediates a cascading effect down a food chain, allowing vegetative growth to rebound (McLaren and Peterson 1994). Although this regulation is often only effective in the short term, the indirect effects of carnivores on community structure and diversity can be significant (Terborgh *et al.* 1999).

I suggest that more data related to the fecundity of the local Moose population on CBI is required to build a dynamic predator-prey model with *VORTEX* that accounts for long-term interaction between both the Moose and a hypothetical Gray Wolf population. These data are necessary to predict the interaction between both species over time. Without such data, only a static and basic deterministic prediction of the reduction in Moose population density can be completed. The functional response curve does not account for the dynamics of

Wolves and Moose interacting over time. The results of the static analysis indicated that more Wolves are required within the National Park to reduce the Moose population to desired densities, and therefore the primary goal of the reintroduction is not met. Based on the analysis the number of Wolves needed to reduce the Moose population are not likely to be able to persist on CBI.

Dynamic modeling in *VORTEX* using local biological data may produce a much different result. I suggest that this is a critical research need to effectively model the Moose population over time subject to various levels of Gray Wolf predation. A preliminary exploration of the impact Wolves would have on the Moose population using a deterministic approach revealed that 30 Wolves could reduce the Moose density to desired levels in 10 years, however a population this size would require area outside of the National Park. VORTEX modelling results indicated that is unlikely that Wolves would survive outside the National Park due to higher expected mortality rates in suboptimal areas. Therefore, only a small population of approximately 16 Wolves would be supported within the National Park, and this population size would not be able to reduce the density to desired levels over a 10-year period; the Moose population would still increase when subject to predation.

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Table 4.1. Life history parameters, catastrophe occurrence, initial population size, and estimation of carrying capacity for *Canis lupus*.

Parameter	Value	Reference
Reproductive System	Long-Term Monogamy	Mech 1970
Age of First Reproduction (♀/♂)	2 years	Peterson et al. 1984; Fuller 1989
Maximum Age of Reproduction	11 years	Mech 1988
Maximum Life Span	16 years	Mech 1970; Mech and Buitani 2003
Maximum Number of Broods per Year	1	Mech 1970
Maximum Number of Progeny per Brood	11	Mech 1970
Sex Ratio at Birth	default 50%	Mech 1975
Density Dependent Reproduction	90% low den, 60% at K	L. D Mech Pers. Comm 2017
% Adult Females Breeding	35, 36 SDEV10	Vonholdt *et al.* 2008; Mech *et al.* 2016
% Adult Males in Breeding Pool	100	Carroll *et al.* 2014
Mortality % - Baseline (age class)	0-1: 30%SDEV10, other: 10%SDEV3	R. Lacy Pers Comm 2017
Catastrophes (CPV Outbreak)	2 % ann. prob; reduce pup surv.75%	Mech and Goyal 1995
Initial Population Size	2~10	Vary - examine output in VORTEX
[1]Carrying Capacity	26 -35	-
[1]Based on estimate (Table 5) *IN* Chapter 3		

Table 4.2. Life history parameters, catastrophe occurrence, initial population size and estimation of carrying capacity for C*anis lycaon*.

Parameter	Value	Reference
Reproductive System	Long-term Monogamy	Pimlott et al. 1969
Age of First Reproduction (♀/♂)	2 years	Theberge and Theberge 2004
Maximum Age of Reproduction	10 years	Theberge et al. 2006
Maximum Life Span	15 years	Mech and Hertel 1983; Thiel and Wydeven 2011
Maximum Number of Broods per Year	1	Pimlott et al. 1969
Maximum Number of Progeny per Brood	8	Graham Forbes pers comm. 2017
Sex Ratio at Birth	default 50%	-
Density Dependent Reproduction	80% low den, 33% at K	Theberge and Theberge 2004
% Adult Females Breeding	57 SDEV10	Theberge and Theberge 2004
% Adult Males in Breeding Pool	80	Theberge and Theberge 2004
Mortality % - Baseline (age class)	0-1: 30%SDEV10, other: 10%SDEV3	R. Lacy Pers Comm 2017
Catastrophes	2 % ann. prob; surv. & repro decline 50%	Theberge and Theberge 2004
Initial Population Size	2~10	Vary - examine output in VORTEX
[1]Carrying Capacity	25 - 47	-
[1]Based on estimate (Table 6) *IN* Chapter 3		

Table 4.3. Baseline *VORTEX* scenario settings for *Canis lupus*.

	Scenario Name:
Model Intricacies~	**"Baseline - Packs-2Pops-Lupus"**
Demographic Inputs:	See Table 1
Mortality% (In Park)	0-1: **30**%SDEV10, other: **10**%SDEV3
Mortality% (Out Park)	0-1: **40**%SDEV10, other: **20**%SDEV3
Number iterations:	100
Number of years (time step):	100
Inbreeding Depression:	YES (6.29 Lethal Equivalents)
State Variables:	YES (3) - See methods for description
Dispersal Modifier:	YES (Based on K inside Park)
Catastrophes:	YES (See Table 1)
Initial Pop Size:	(5) - In Park & Out Park
Carrying Capacity:	16 In Park(3 packs), 14 Out Park(2 packs)
Harvest:	NO
Supplementation:	NO

Table 4.4. Baseline *VORTEX* scenario settings for *Canis lycaon*.

	Scenario Name:
Model Intricacies~	**"Baseline - Packs-2Pops-Lycaon"**
Demographic Inputs:	See Table 2
Mortality% (In Park)	0-1: **30**%SDEV10, other: **10**%SDEV3
Mortality% (Out Park)	0-1: **40**%SDEV10, other: **20**%SDEV3
Number iterations:	100
Number of years (time step):	100
Inbreeding Depression:	YES (6.29 Lethal Equivalents)
State Variables:	YES (3) - See methods for description
Dispersal Modifier:	YES (Based on K inside Park)
Catastrophes:	YES (See Table 2)
Initial Pop Size:	(5) - In Park & Out Park
Carrying Capacity:	17 In Park(3 packs), 16 Out Park(3 packs)
Harvest:	NO
Supplementation:	NO

Table 4.5. Available data on population size and density of Moose in Cape Breton Highlands National Park, 2002-2015, from Parks Canada aerial surveys (Parks Canada 2015).

Year	Population Estimate	Mean Density (moose/km²)
2002	2805.09	2.47
2004	3987.6	4.22
2006	1711.93	1.42
2008	1168.97	1.11
2011	2101.99	2.7
2015	1769.53	1.84
Average (*2011-2015*):	**1936**	**2.3**

Table 4.6. Baseline and simulations 1-3. The % females breeding is linearly increased through simulations 1-3. Mortality rates (no human-induced mortality) is set equal for simulations 1-3. The K is also set equal to 36 (1-3). The % females breeding is a function of biological control. Mortality is representative of wolves receiving full legal protection from humans.

Parameter Input	baseline	simulation 1	simulation 2	simulation 3	[3]Direct Control (*fn*)
% Females breeding	35	45	50	55	Biological
[1]Mortality	30/10-40/20	30 / 10	30 / 10	30 / 10	human
[2]Packs (In, Out Park)	3,2	4,2	4,2	4,2	human
Carrying Capacity	30	36	36	36	human
[1]Wolves (Sim. 1-3) (In, Out Park) are not subject to human induced mortality risk (30% pups, 10% adults, R. Lacy pers. comm. 2017)					
[2](Sim 1-3)(4) packs inside park, (2) packs outside park					
[3]Variability of parameter value is directly biologically related, or human-influenced					

Table 4.7. Baseline and simulations 4-6. The % females breeding is linearly increased through simulations 4-6. Mortality rates (no human-induced mortality) is set equal for simulations 4-6. The K is also set equal to 48 (1-3). The % females breeding is a function of biological control. Mortality is representative of wolves receiving full legal protection from humans.

	Value				
Parameter Input	baseline	simulation 4	simulation 5	simulation 6	[3]Direct Control (*fn*)
% Females breeding	35	45	50	55	Biological
[1]Mortality	30/10-40/20	30 / 10	30 / 10	30 / 10	human
[2]Packs (In, Out Park)	3,2	5,3	5,3	5,3	human / biological
Carrying Capacity	30	48	48	48	human / biological
[1]Wolves (Sim. 1-3)(In, Out Park) are not subject to human induced mortality risk (30% pups, 10% adults, R. Lacy pers. comm. 2017)					
[2](Sim 1-3)(5) packs inside park, (3) packs outside park					
[3]Variability of parameter value is directly biologically related, or human-influenced					

Table 4.8. Number of Moose killed per year for an average Wolf population size (from simulations 1,3,4, and 6 in Tables 4.6 - 4.7) and subsequent Moose reduction densities exerted from kill rate of the population of 10.2 Moose killed per Wolf on an annual basis (Figure 4.1).

Simulation #	Population Size ($\bar{x}$ 50 years)	Pr. Extinction	Moose Killed	Subsequent Moose Density
1	23	0.52	234.6	2
3	25	0.35	255	1.99
4	27	0.29	275.4	1.97
6	32	0.03	326.4	1.91

Table 4.9. Deterministic model predictions of the current Cape Breton Highlands National Park Moose population subject to predation by 30 wolves over a 10-year period. The Moose population has an annual growth of 0.1 before Wolf predation.

Year	1	2	3	4	5	6	7	8	9	10
Moose Population	1936	1,799	1,654	1,499	1,336	1,164	986	804	623	447
Moose Area	950	950	950	950	950	950	950	950	950	950
Moose Density	2.0	1.9	1.7	1.6	1.4	1.2	1.0	0.8	0.7	0.5
[1]Moose Killed / 100 Wolf Days	2.7	2.7	2.7	2.6	2.5	2.4	2.3	2.2	2.0	1.7
Wolves	30	30.0	30.0	30.0	30.0	30.0	30.0	30.0	30.0	30.0
Wolf Days/Year	10,950	10,950	10,950	10,950	10,950	10,950	10,950	10,950	10,950	10,950
Moose Killed/Year	300	296	291	285	277	268	255	238	216	186

[1]The kill rate and Wolf/days/year are defined according to values in Messier (1994).

Table 4.10. Deterministic model predictions of the current Cape Breton Highlands National Park Moose population subject to predation by 16 wolves over a 10-year period. The Moose population has an annual growth of 0.1 before Wolf predation.

Year	1	2	3	4	5	6	7	8	9	10
Moose Population	1936	1,954	1,972	1,993	2,015	2,039	2,066	2,094	2,125	2,158
Moose Area	950	950	950	950	950	950	950	950	950	950
Moose Density	2.0	2.1	2.1	2.1	2.1	2.1	2.2	2.2	2.2	2.3
[1]Moose Killed / 100 Wolf Days	2.7	2.7	2.8	2.8	2.8	2.8	2.8	2.8	2.8	2.8
Wolves	16	16.0	16.0	16.0	16.0	16.0	16.0	16.0	16.0	16.0
Wolf Days/Year	5,840	5,840	5,840	5,840	5,840	5,840	5,840	5,840	5,840	5,840
Moose Killed/Year	160	160	161	161	161	162	162	162	163	163

[1]The kill rate and Wolf/days/year are defined according to values in Messier (1994).

Figure 4.1. The functional response of wolves to changing Moose density (from Messier 1994). The kill rate is shown per 100 Wolf days.

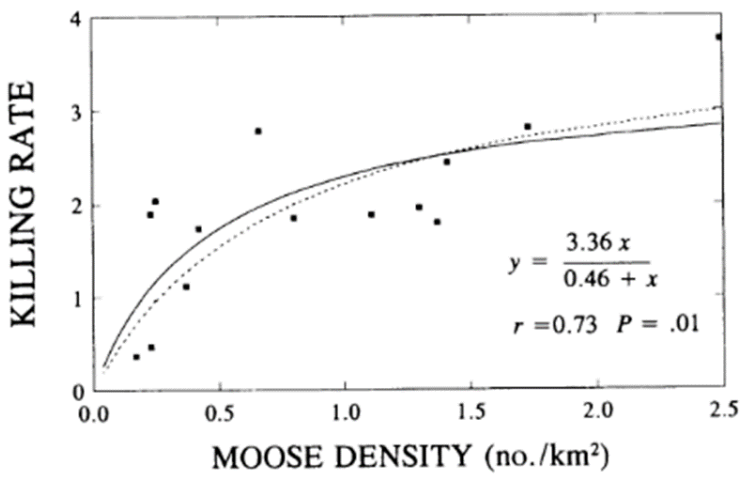


Figure 4.2. Baseline scenario of probability of extinction for *Canis lupus*.

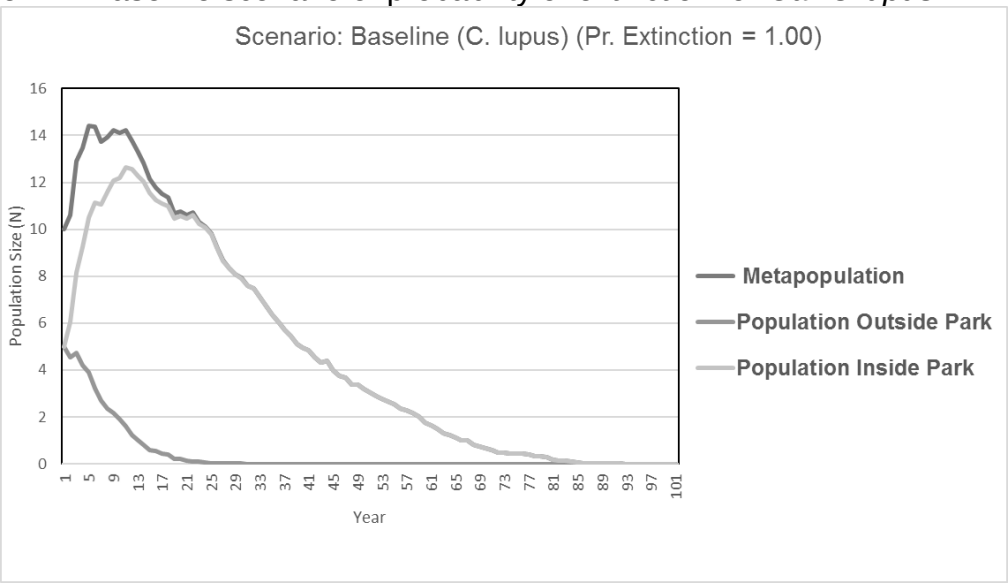


Figure 4.3. Probabilities of extinction for *Canis lupus* when the initial population size is varied.

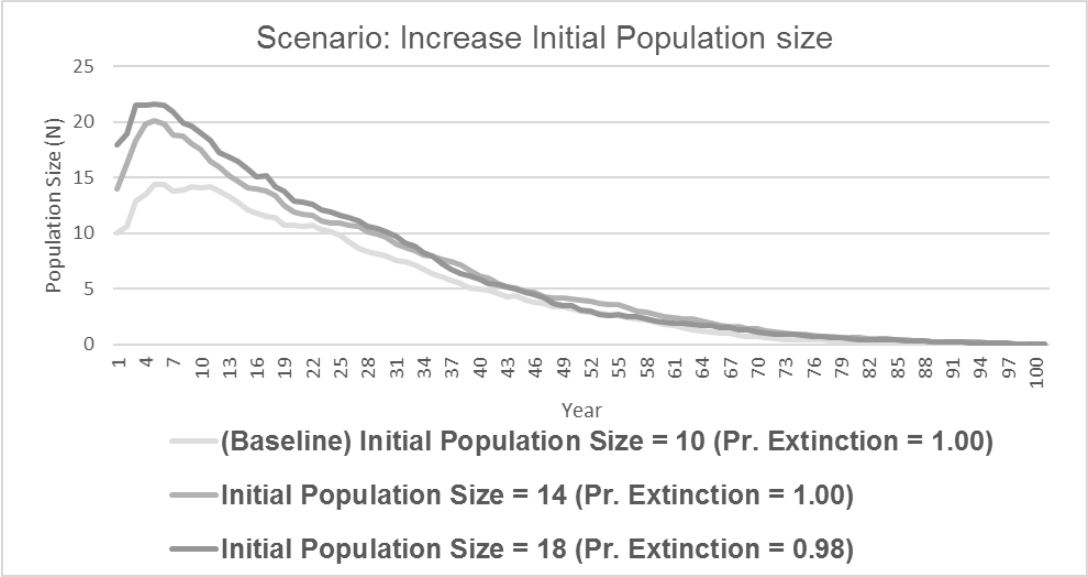


Figure 4.4. Probabilities of extinction for *Canis lupus* when the % of adult females breeding increases.

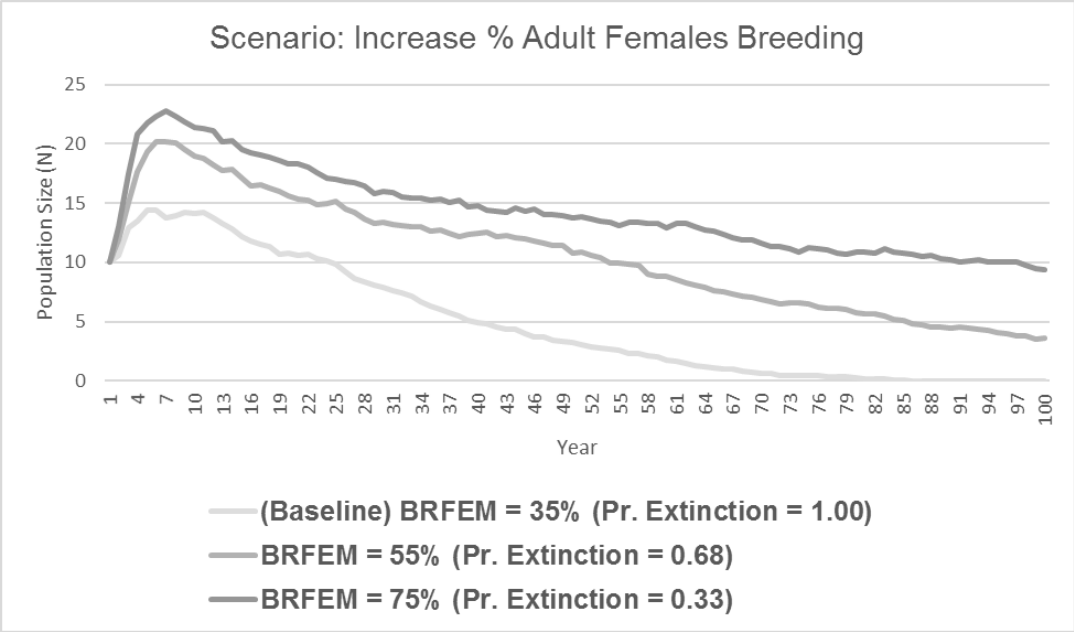
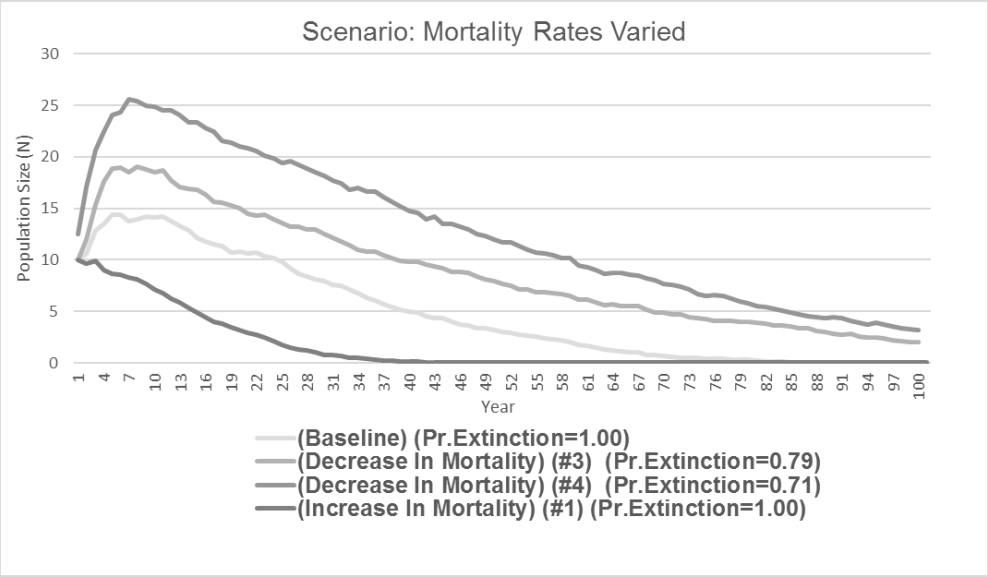


Figure 4.5. [1]Probabilities of extinction for *Canis lupus* when the mortality rates are decreased (Simulations 3-4) and increased (Simulation 1).



[1](*Baseline*) Mortality INPARK = (pups30%SDEV10), (adults10%SDEV3), OUTPARK = (pups40%SDEV10), (adults20%SDEV3). (*Increase in Mortality Sim # 1*) INPARK = (pups30%SDEV10), (adults10%SDEV3), OUTPARK = (pups50%SDEV10), (adults30%SDEV3). *(Decrease in Mortality Sim #3)* INPARK = (pups15%SDEV10), (adults5%SDEV3), OUTPARK(pups25%SDEV10), (adults15**%**SDEV3). *(Decrease in Mortality Sim #4)* INPARK = (pups15%SDEV10), (adults5%SDEV3), OUTPARK = (pups15%SDEV10), (adults5%SDEV3).

Figure 4.6. Probabilities of extinction for *Canis lupus* when the carrying capacity is increased.

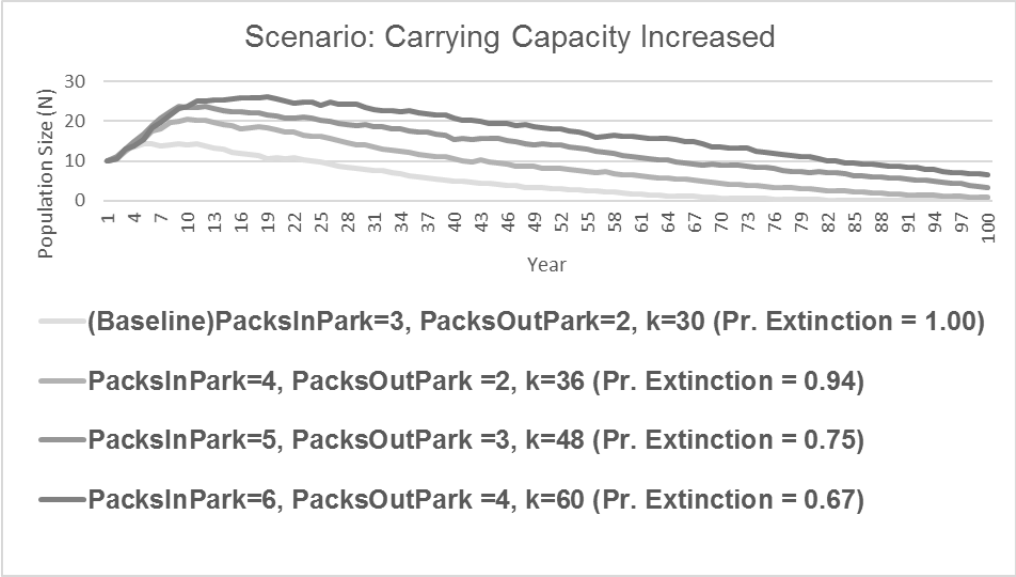


Figure 4.7. Probabilities of extinction for *Canis lupus* when the population is supplemented.

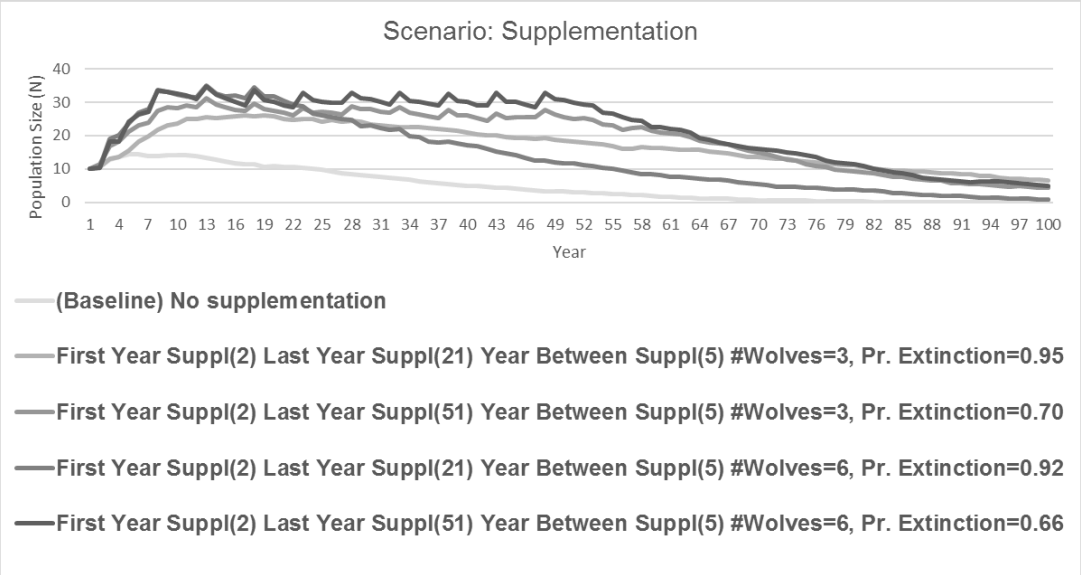


Figure 4.8. Baseline scenario of probability of extinction for *Canis lycaon.*

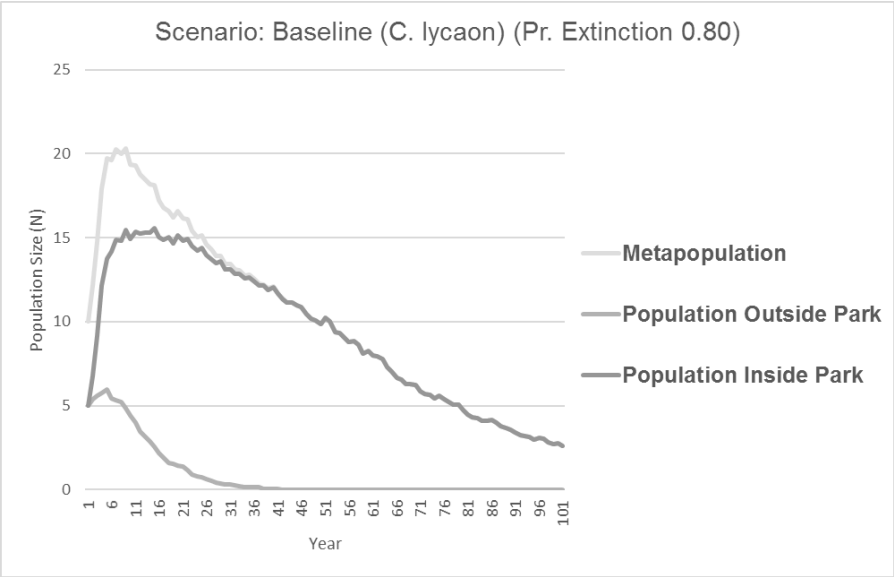


Figure 4.9. Probabilities of extinction for *Canis lycaon* when the initial population size is varied.

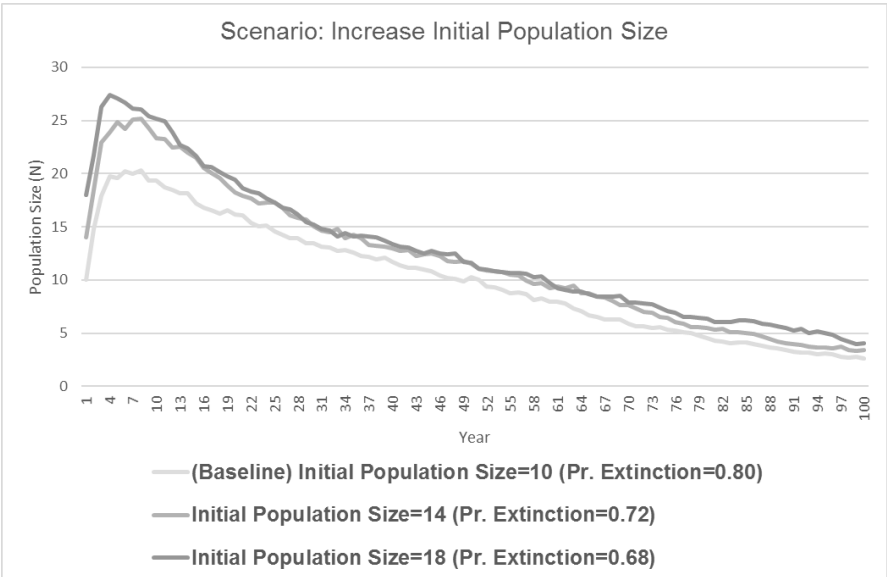


Figure 4.10. Probabilities of extinction for *Canis lycaon* when the percent of adult females breeders are decreased and then increased.

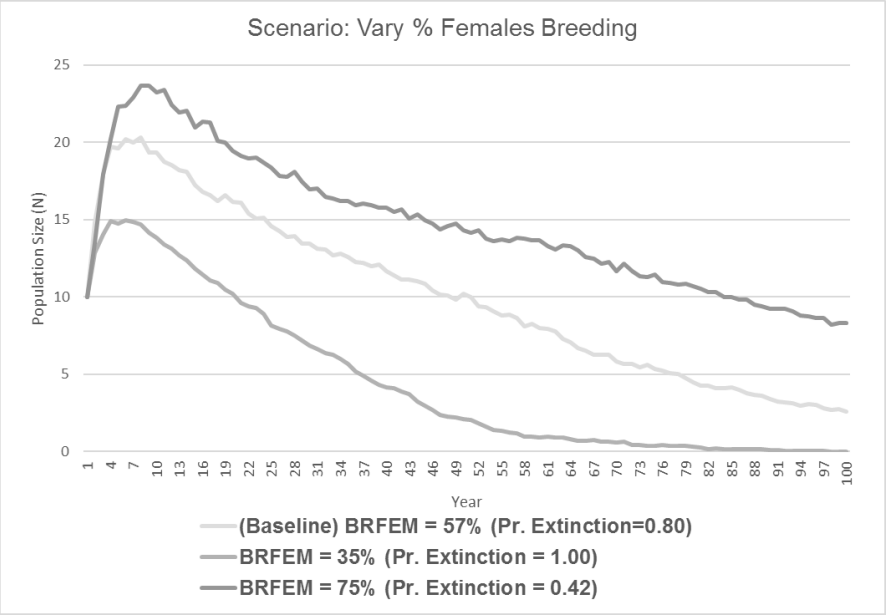
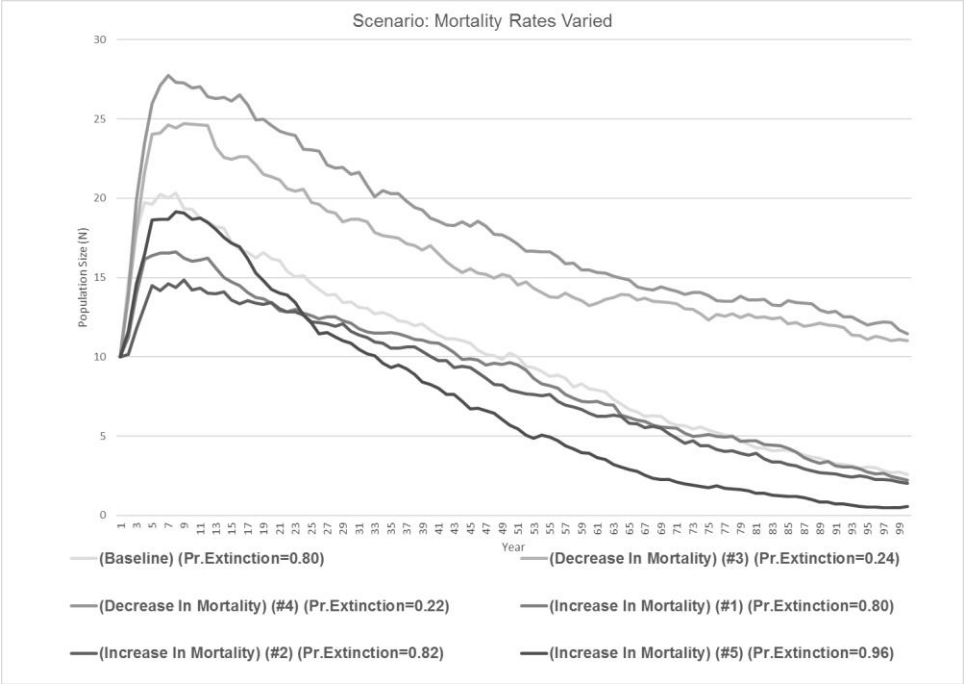


Figure 4.11. [1]Probabilities of extinction for *Canis lycaon* when the mortality rates are decreased (simulations 3 & 4) and increased (simulations 1, 2 and 5).



[1](*Baseline*) Mortality INPARK = (pups30%SDEV10), (adults10%SDEV3), OUTPARK = (pups40%SDEV10), (adults20%SDEV3). (*Decrease in Mortality Sim #3*) INPARK = (pups15%SDEV10), (adults5%SDEV3), OUTPARK = (pups25%SDEV10), (adults15%SDEV3). *(Decrease in Mortality Sim #4)* INPARK = (pups15%SDEV10), (adults5%SDEV3), OUTPARK = (pups15%SDEV10), (adults5%SDEV3). (*Increase in Mortality Sim # 1*) INPARK = (pups30%SDEV10), (adults10%SDEV3), OUTPARK = (pups50%SDEV10), (adults30%SDEV3). (*Increase in Mortality Sim # 2*) INPARK = (pups30%SDEV10), (adults10%SDEV3), OUTPARK = (pups60%SDEV10), (adults40%SDEV3). (*Increase in Mortality Sim # 5*) INPARK = (pups35%SDEV10), (adults15%SDEV3), OUTPARK = (pups40%SDEV10), (adults20%SDEV3).

Figure 4.12. Probabilities of extinction for *Canis lycaon* when the carrying capacity is increased.

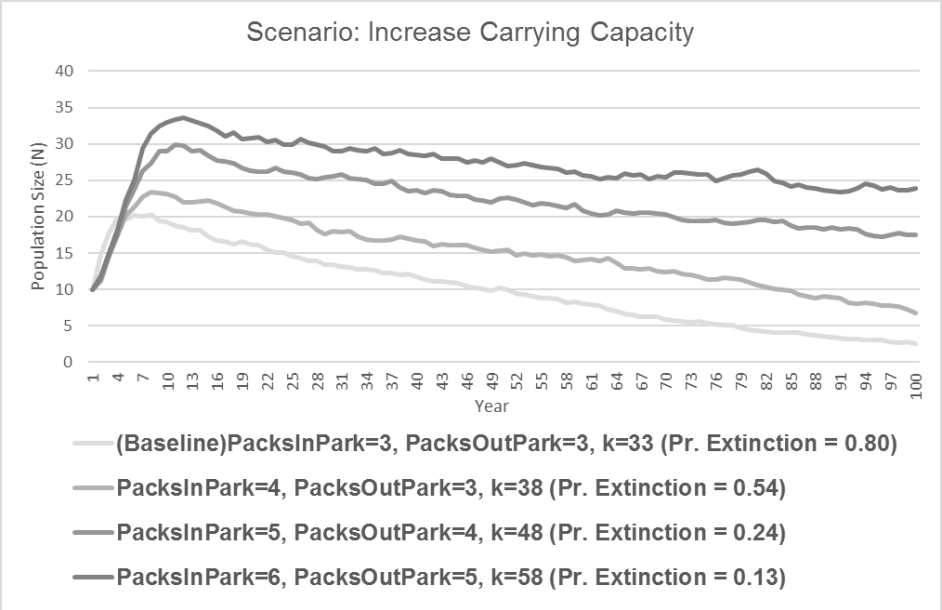


Figure 4.13. Probabilities of extinction for *Canis lycaon* when the population is supplemented.

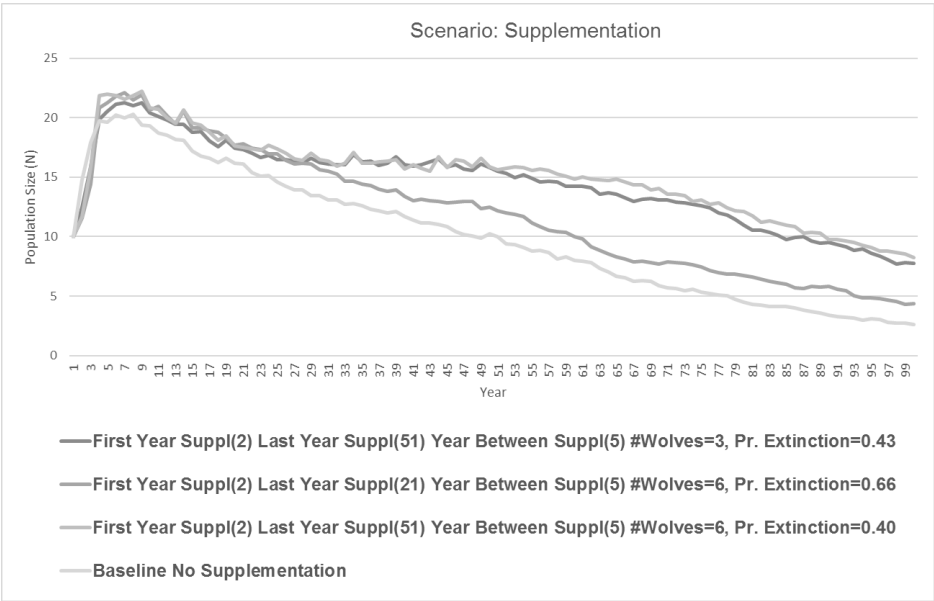


Figure 4.14. Probabilities of extinction for *Canis lupus* under baseline simulation, and simulations 1-3.

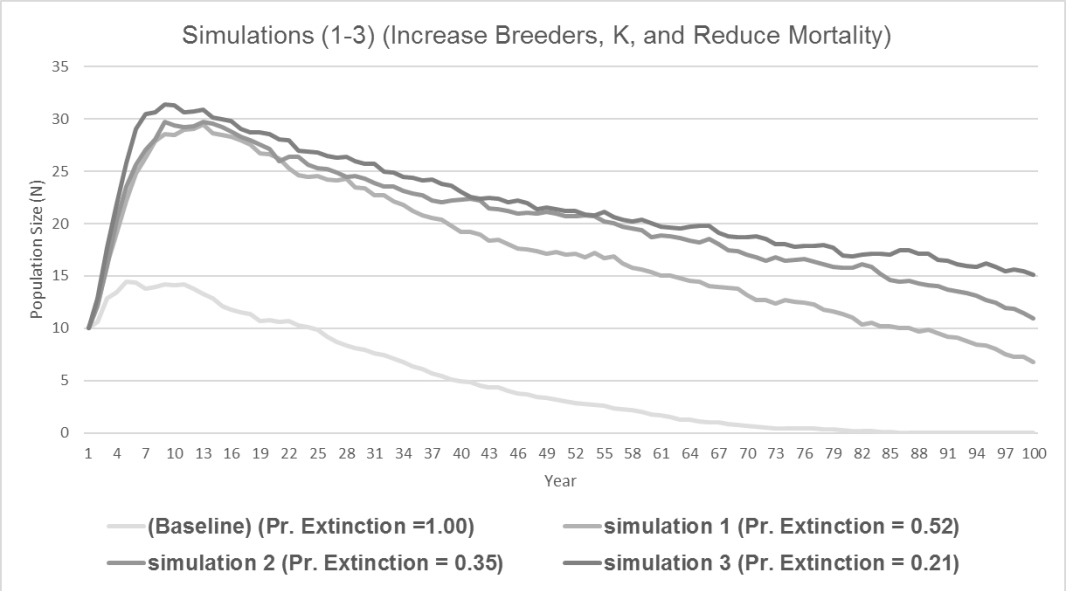
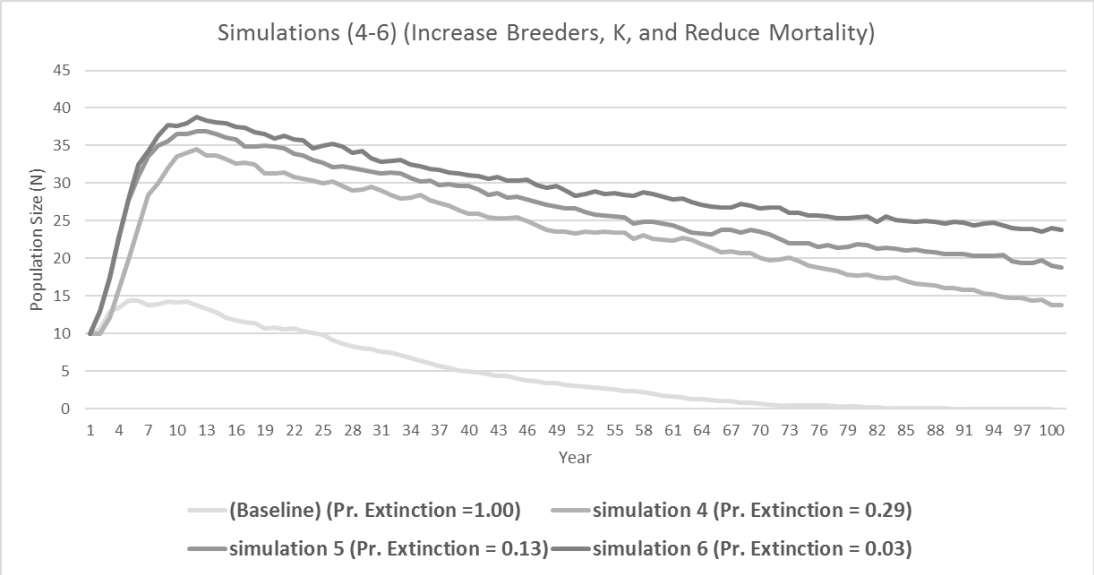


Figure 4.15. Probabilities of extinction for *Canis lupus* under baseline simulation, and simulations 4-6.



Chapter 5: General Discussion

There is an extensive body of literature that suggests Wolves and Moose can co-exist, and only under rare circumstances do Wolves deplete their prey population (Noss *et al.* 1996, Mech and Peterson 2003). Wolves rely heavily on large ungulate populations such as Moose for survival and reproduction (Mech 1974). Wolves will test and exhibit predation pressure on all segments of an ungulate population, but are often only successful in killing weak or distressed individuals, thus the secure prime-aged segment of the population survives (Allen 1979, Packard and Mech 1980). Wolves are known to be effective in controlling hyperabundant populations of Moose as well as other ungulates, and exhibiting regulatory effects that prevent Moose from exploiting local plant communities (Messier and Joly 2000, Soulé *et al.* 2003). However, there is still debate on whether Wolf predation directly controls prey density, or just simply substitutes for other forms of mortality (Eberhardt *et al.* 2003, Mech and Peterson 2003). Nonetheless, large carnivores such as the Wolf have an important role in the maintenance of biological diversity in forest segments where large ungulates such as Moose are present (McLaren and Peterson 1994).

As an apex predator, Wolves would become part of the Cape Breton Highlands National Park ecosystem. If a population of Wolves could become established, they may reduce the Moose population to a certain extent and prevent the population from reaching hyperabundant levels. Hyperabundance causes substantial and landscape level change in vegetation structure and composition (Soulé *et al.* 2005). Wolves exert “top-down” effects which indirectly increase vegetation biomass by reducing ungulate densities and influencing their behavior and movement patterns (Hebblewhite *et al.* 2005). This ecological role cannot be fully replaced by human action (Ripple *et al.* 2014).

In regards to national park policy, it would be ideal to restore a Wolf population that could exert indirect influences on vegetation through predation. Reintroduction of Wolves in Cape Breton Island (CBI) would provide a unique opportunity for scientists to test this hypothesis, considering Wolves have never been reintroduced on a landscape where Moose constitute the primary large prey

base and White-tailed Deer are secondary. The ability of Wolves to facilitate a top-down trophic cascade has been of interest to park managers since it mitigates the negative impacts and ecological consequences of hyperabundant ungulates (Parks Canada 2000; Carroll *et al.* 2001).

Wolves are habitat generalists and can survive in many environments. Ideal locations for Wolf restoration requires low persecution from humans, and abundant large ungulate prey populations. This thesis reports that there is available space for Wolves to form several territorial packs (< 50 animals) on CBI; however, mortality rates would need to be low and humans must limit their impact to ensure the long-term survival of the population. The area of CBI outside of the National Park would require full legal protection. Wolf mortality outside of protected areas can greatly affect the viability and ecological role of Wolves within the protected area (Forbes and Theberge 1996, Rutledge *et al.* 2010). Furthermore, since Wolves have long dispersal ranges, and sometimes occupy home ranges than are greater than the extent of the National Park, protected areas outside of the Park are mandatory for Wolf survival.

Moose constitute the primary prey source for Gray Wolves on Isle Royale, and both species have co-existed for several decades (Peterson 1999). Wolves have never depleted the Moose population on Isle Royale to levels where they are at risk of local extinction. Therefore, it is possible that Wolves would not cause Moose to become extinct on CBI. However, the population dynamics between Moose and Wolf on Isle Royale occur where human harvest of Moose is absent; the potential additive effects of human harvest with Wolf predation on Moose in CBI is unknown and would require data not presently available.

The ability of Wolves, especially the Eastern Wolf, to recolonize the area may also be affected by their propensity to hybridize with Eastern Coyote, which has been present on the Island for several decades (Patterson and Messier 2003). This potential for severe introgression (‘genetic swamping’) would be a concern for Park Managers because the more abundant Coyote would be available for breeding with the fewer, introduced Eastern Wolf females. Also, increased mortality rates that would be likely to occur outside the national park,

could alter pack structure of Eastern Wolves, resulting in a deterioration of the alpha-only breeding system. In Algonquin Provincial Park, Ontario, there is evidence that Eastern Wolf have less introgression from Eastern Coyote in protected areas where trapping is infrequent. The theory is that higher mortality rates removes dominant (alpha) breeders and the survivors are breeding with a wider range of mates, including Eastern Coyote (Rutledge *et al.* 2010). The larger Gray Wolf may be less likely to hybridize with Eastern Coyote (Pilgrim et al. 1998; Benson *et al.* 2012), and it perhaps could cause a reduction in Coyote numbers, and as a result be the more favorable Wolf species for reintroducing to the area.

Theoretical population modeling gave insight to the hypothetical viability of the Wolf population. Variables that were most influential to the viability of the population were identified as mortality rates, carrying capacity and the percentage of breeding females. They appear to limit the long-term viability of a hypothetically reintroduced Wolf population on CBI. To ensure long-term viability (Pr. Extinction <0.25) over a 100-year time horizon mortality rates outside of the National Park should match those inside the Park (modelled as 30% pups and 10% adults). Carrying capacity should be at a minimum of 36 Wolves, and the percentage of adult females breeding at any given time in the 50-55% range. Human attitudes and actions directly affect each of these factors. Humans can allow Wolves to exist on the landscape by providing enough protected area that does not allow hunting or trapping activities. Either of these activities greatly reduces the ability of Wolves to maintain a self-sustaining population, and therefore should be initiated in areas where Wolves are present (Haber 1996).

The number of Wolves required to limit Moose to desired densities was estimated using the best available local data with the functional response of Wolves to changing Moose density on a yearly basis (Messier 1994). However, based on this analysis, there is not enough space (approximately 5,625 km² is required) on CBI to support the number of Wolves required to reduce Moose densities in Cape Breton Highlands National Park to desired levels. There is only approximately 750km² of optimal and 720km² of suboptimal habitat on CBI. Using

VORTEX to model dynamic predictions of the Moose population reduction over a set time horizon may produce a very different result due to incorporation of dynamic species interactions. To complete this would require specific Moose population data (R. Lacy, pers. comm. 2017), and in the period of this MScF thesis such data were not available. However, a preliminary deterministic modelling approach showed that 30 Wolves could reduce the Moose population density if there was no human-induced mortality outside of the National Park and Wolves received full legal protection from harvesting and trapping. It is very unlikely that 30 Wolves would persist as mortality outside of the Park is expected to be high, and therefore only 16 Wolves would be able to occupy the area within the National Park. This small population size inside the National Park would not be viable and based on the deterministic modeling results, this population would not be large enough to reduce approximately 2,000 Moose within the park to a density of 0.5 Moose/km^2.

Notwithstanding the requirements of a long-term viable population, to accurately predict whether Wolves could limit the Moose population over time, I recommend that age-specific fecundity and mortality data, as well as survival rate data of the local Moose population be attained. Gathering more data on the local Moose population in CBI may result in more accurate predictions of how the two populations might interact over time. I also suggest that there be a cost-benefit analysis that measures conservation dollars spent on further Moose research vs. conservation dollars spend on achieving the initial management objective of Moose reduction. For example, it may cost less to utilize another means of Moose population reduction, verses spending several years to investigate the affect Wolves may exhibit on the population.

Furthermore, if Parks Canada is interested in building on this project, I recommend a local study of human attitudes towards Wolves in CBI to be completed. A Wolf reintroduction is much more likely to succeed if all interest groups are willing to co-exist with them on the landscape (Mech and Peterson 2003). When people fear an animal, they are less willing to want to co-exist with it (Bergstrom 2017). Sponarski *et al.* (2015) stated that residents in northern CBI

were more fearful of and had less tolerance for Coyotes, a much smaller canid than the Eastern or Gray Wolf, and therefore are less likely to participate in activities within the Park. This informs managers that when people are afraid of Coyotes they have a higher perceived risk of encountering one, and are much less tolerant of the species.

Sponarski *et al.* (2015) concluded that outreach and educational seminars are needed to inform residents that human-Coyote interaction are extremely rare, and that people should not have a high perceived risk of encountering one within the Park. If people are willing to understand the ecological role of Wolves and Coyotes, they tend to be much more willing to accept their presence. Human attitudes can directly reflect mortality risk for Wolves; therefore, positive attitudes are an integral component to ensuring Wolves are not exposed to excessive anthropogenic mortality.

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APPENDIX 1 - Part 1. [1]Quantifiable parameters (*biophysical, infrastructure*) used in [2]referenced reintroduction assessments, recovery plans habitat suitability models, and related literature. Includes Pre-GIS studies in North American, Post-GIS studies in Europe, and Post-GIS thesis in North America.

		Biophysical								Infrastructure				
	Reference	Land Cover	Prey Base	Water Availability	Climate	Topography	Hydrology	Winter Severity	Land Area	Human Density	Road Density	Land Ownership	Land Use	Livestock Abundance
Pre GIS (North America)	1	x	x	x	x	x				x	x		x	x
	2	x	x	x				x	x	x	x	x	x	x
	3	x	x	x	x			x	x	x			x	x
	4	x	x	x		x			x	x			x	x
	5	x	x	x	x	x		x	x	x	x		x	x
	6	x	x						x	x	x	x	x	x
	7	x	x	x	x	x			x	x		x	x	x
	8	x	x	x		x		x	x	x		x	x	x
	9	x	x		x	x		x	x	x	x	x	x	x
	10	x	x						x	x	x	x	x	x
Post GIS (Europe)	11	x			x				x	x	x	x	x	x
	12	x	x		x	x				x	x		x	x
	13	x	x		x					x	x			x
	14	x	x			x				x	x		x	
	15	x				x				x	x	x		x
	16	x	x		x	x				x	x		x	x
	17	x	x			x				x	x		x	x
Post GIS (Thesis)	18	x			x	x				x	x	x	x	x
	19	x	x		x	x				x	x	x	x	x
	20	x	x		x					x	x	x	x	x
	21	x	x			x			x	x	x			
	22	x	x			x	x			x	x	x	x	x
	23	x	x	x	x	x				x	x	x	x	

[1]Light black x's indicate parameter was only qualitatively considered in the referenced study.
[2]1(Bednarz 1989), 2(Bennett 1994), 3(Carley and Mechler 1983), 4(Parker 1987), 5(Parker 1990), 6(USFWS 1978), 7(USFWS 1982), 8(USFWS 1987), 9(USFWS 1990), 10(USFWS 1992), 11(Cayuela 2004), 12(Glenz et al. 2001), 13(Jedrzejewski et al. 2008), 14(Jedrzejewski et al. 2005), 15(Jedrzejewski et al. 2004), 16(Massolo and Meriggi 1998), 17(Zlantanova and Popova 2013), 18(Claeys 2010), 19(Houts 2001), 20(Larsen 2004), 21(Swan 2005), 22(Jacobs 2009), 23(Whitaker 2006).

APPENDIX 1 - Part 2. [1]Quantifiable parameters (*biological, population data and sociological*) used in [2]referenced reintroduction assessments, recovery plans habitat suitability models, and related literature. Includes Pre-GIS studies in North American, Post-GIS studies in Europe, and Post-GIS thesis in North America.

		Biological						Population Data								Sociological	
	Reference	Pack Home Range	Historic Occurrence	Hybridi-zation?	Effects on Scaven-gers	Disease	Intra-specific Comp.	Survival Rates	Pack Size	Density	Sex/Age Ratio	Natality	Fecundity	Mortality	Dispersal	Public Attitude & Opinion	Public Behavior & Impact
Pre GIS (North America)	1			x		x									x		x
	2	x	x	x	x	x	x	x	x	x		x		x	x		
	3	x	x	x												x	x
	4		x	x								x	x			x	x
	5	x	x	x								x				x	x
	6	x	x		x											x	x
	7		x	x				x	x			x				x	x
	8	x	x						x	x	x	x	x	x	x	x	x
	9	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	10	x	x	x	x	x		x	x					x		x	x
Post GIS (Europe)	11	x	x					x						x	x	x	x
	12																
	13	x	x						x	x					x	x	x
	14	x	x							x							
	15		x						x								
	16	x	x														
	17	x	x							x							
Post GIS (Thesis)	18	x	x						x	x					x	x	
	19	x								x					x	x	x
	20	x	x						x	x					x		
	21		x			x			x				x		x		x
	22	x	x	x		x		x	x	x	x		x		x	x	x
	23	x	x	x	x								x			x	x

[1]Light black x's indicate parameter was only qualitatively considered in the referenced study.
[2]1(Bednarz 1989), 2(Bennett 1994), 3(Carley and Mechler 1983), 4(Parker 1987), 5(Parker 1990), 6(USFWS 1978), 7(USFWS 1982), 8(USFWS 1987), 9(USFWS 1990), 10(USFWS 1992), 11(Cayuela 2004), 12(Glenz et al. 2001), 13(Jedrzejewski et al. 2008), 14(Jedrzejewski et al. 2005), 15(Jedrzejewski et al. 2004), 16(Massolo and Meriggi 1998), 17(Zlantanova and Popova 2013), 18(Claeys 2010), 19(Houts 2001), 20(Larsen 2004), 21(Swan 2005), 22(Jacobs 2009), 23(Whitaker 2006).

APPENDIX 1 - Part 3. [1]Quantifiable parameters (*biophysical, infrastructure*) used in [2]referenced reintroduction assessments, recovery plans habitat suitability models, and related literature. Includes Post-GIS studies in North America.

		Biophysical								Infrastructure				
	Reference	Land Cover	Prey Base	Water Availability	Climate	Topography	Hydrology	Winter Severity	Land Area	Human Density	Road Density	Land Ownership	Land Use	Livestock Abundance
Post GIS (North America)	24	x	x		x	x		x	x	x	x	x	x	x
	25	x	x	x	x	x		x	x	x	x	x	x	
	26	x	x		x	x			x	x	x	x	x	x
	27	x	x	x						x	x	x	x	
	28	x	x						x	x	x		x	
	29	x	x	x					x	x	x		x	x
	30	x	x		x					x	x	x	x	
	31		x			x				x	x	x	x	x
	32	x	x			x			x	x	x	x	x	
	33	x	x						x	x	x	x	x	x
	34		x		x				x	x	x		x	x
	35								x		x	x	x	
	36	x							x	x	x		x	
	37	x							x	x	x		x	
	38	x				x					x	x	x	
	39	x	x		x	x			x	x	x	x		
	40	x	x		x	x								
	41	x	x			x								
	42		x								x			
	43	x	x		x	x					x	x	x	x
	44	x	x							x	x	x	x	x
	45	x	x							x	x	x	x	
	46										x			
	47	x	x			x				x	x	x	x	x
	48		x						x		x			
	49	x	x							x	x	x	x	
	50	x	x	x						x	x	x		
	51	x	x			x			x	x	x			
	52	x	x	x	x	x			x	x	x			
	53								x	x	x			

[1]Light black x's indicate parameter was only qualitatively considered in the referenced study.
[2]24(Carroll et al. 2003), 25(Paquet et al. 1999), 26(Ratti et al. 1999), 27(Sneed 2001), 28(Belongie 2008), 29(Beerman 2009), 30(Carroll 2003), 31(Carroll et al. 2001a), 32(Carroll et al. 2001b), 33(Carroll et al. 2004), 34(Carroll et al. 2006), 35(Gehring and Potter 2005), 36(Harrison and Chapman 1997), 37(Harrison and Chapman 1998), 38(Houts 2003), 39(Larsen and Ripple 2006), 40(Mcloughlin et al. 2004), 41(Milakovic et al. 2011), 42(Mladenoff and Sickley 1998), 43(Mladenoff et al. 1999), 44(Mladenoff et al. 1997), 45(Mladenoff et al. 1995), 46(Merrill 2000), 47(Oakleaf et al. 2006), 48(Potvin et al. 2005), 49(Wydevan et al. 2001), 50(Wydevan et al. 1999), 51(Shaffer 2007), 52(Switalski et al. 2002), 53(Thiel 1985).

APPENDIX 1 - Part 4. [1]Quantifiable parameters (*biological, population data and sociological*) used in [2]referenced reintroduction assessments, recovery plans habitat suitability models, and related literature. Includes Post-GIS studies in North America.

		Biological						Population Data								Sociological	
	Reference	Pack Home Range	Historic Occurrence	Hybridi-zation?	Effects on Scaven-gers	Disease	Intra-specific Comp.	Survival Rates	Pack Size	Density	Sex/Age Ratio	Natality	Fecundity	Mortality	Dispersal	Public Attitude & Opinion	Public Behavior & Impact
Post GIS (North America)	24	x	x						x	x	x		x	x	x	x	
	25	x	x	x	x	x	x		x						x	x	x
	26		x	x		x				x					x	x	
	27		x							x							x
	28	x	x														x
	29	x	x						x	x					x	x	
	30	x	x					x	x				x	x	x		x
	31	x	x		x		x	x		x			x	x	x	x	x
	32									x							
	33	x	x					x	x				x	x	x		x
	34	x	x					x	x	x			x	x	x		x
	35	x								x					x		
	36	x		x						x						x	x
	37			x						x							x
	38	x	x												x		x
	39		x						x	x				x	x		x
	40		x			x	x										
	41	x	x				x		x						x		x
	42							x		x							
	43	x	x					x		x				x	x		
	44	x								x				x		x	
	45	x															
	46																
	47	x	x											x			x
	48	x	x							x							
	49	x	x			x	x							x		x	x
	50																
	51	x	x	x													
	52	x	x		x				x	x					x	x	x
	53		x			x	x										x

[1]Light black x's indicate parameter was only qualitatively considered in the referenced study.
[2]24(Carroll et al. 2003), 25(Paquet et al. 1999), 26(Ratti et al. 1999), 27(Sneed 2001), 28(Belongie 2008), 29(Beerman 2009), 30(Carroll 2003), 31(Carroll et al. 2001a), 32(Carroll et al. 2001b), 33(Carroll et al. 2004), 34(Carroll et al. 2006), 35(Gehring and Potter 2005), 36(Harrison and Chapman 1997), 37(Harrison and Chapman 1998), 38(Houts 2003), 39(Larsen and Ripple 2006), 40(Mcloughlin et al. 2004), 41(Milakovic et al. 2011), 42(Mladenoff and Sickley 1998), 43(Mladenoff et al. 1999), 44(Mladenoff et al. 1997), 45(Mladenoff et al. 1995), 46(Merrill 2000), 47(Oakleaf et al. 2006), 48(Potvin et al. 2005), 49(Wydevan et al. 2001), 50(Wydevan et al. 1999), 51(Shaffer 2007), 52(Switalski et al. 2002), 53(Thiel 1985).

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