Evaluating the Potential for Species Reintroductions in Canada

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Abstract: Species reintroductions and translocations are increasingly useful conservation tools for restoring endangered populations around the world. We examine ecological and socio-political variables to assess Canada's potential for future reintroductions. Biologically ideal species would be prolific, terrestrial, herbivorous, behaviorally simple, charismatic, easily tractable, or large enough to carry transmitters for post-release evaluations, and would have small home range requirements. Sociologically, Canada's large geographic area, low human density, high urban population, widespread protectionist views towards wildlife, and sound economic status should favor reintroduction success. Canada has implemented legislation to safeguard species at risk and, compared to developing countries, possesses substantial funds to support reintroduction efforts. We support the reintroduction guidelines put forth by the World Conservation Union (IUCN) but realize that several challenges regarding these parameters will unfold in Canada's future. Pressures from the rates of species loss and climate change may precipitate situations where species would need to be reintroduced into areas outside their historic range, subspecific substitutions would be necessary if taxonomically similar individuals are unavailable, and in crisis situations, reintroductions may need to be attempted before historic population decline factors are fully understood. Given a sound understanding of population threats, sufficient habitat, and adequate resources, some Canadian species that show promise for successful reintroduction are the Queen Charlotte Island ermine (*Mustela erminea haidarum*), American badger (*Taxidea taxus jeffersonii*), pallid bat (*Antrozous pallidus*), barn owl (*Tyto alba*), white-headed woodpecker (*Picoides albolarvatus*), and stinkpot (*Sternotherus odoratus*).

Key Words: reintroduction, translocation, endangered species, conservation, Canada

Introduction

Faced with a global extinction crisis, reintroductions and translocations are becoming increasingly important conservation tools for restoring endangered species populations. In 2002, the IUCN/SSC Reintroduction Specialist Group held a strategic planning workshop where reintroductions were shown to be growing in global conservation significance because they (1) are increasing in number, (2) are attracting public attention, (3) are regionally important, and (4) can use flagship species to facilitate habitat conservation (IUCN/SSC Reintroduction Specialist Group 2002). Biological, logistical, organizational, and even legal challenges have limited the success of many reintroductions, and thus far, very few have led to the reestablishment of viable populations in the wild (Griffith et al. 1989; Beck et al. 1994; Fischer and Lindenmayer 2000). The ultimate success of a reintroduction is profoundly influenced by various methodological, environmental, species-specific, and population-level factors (Wolf et al. 1998), yet reintroductions have routinely been carried out with minimal prior assessment of the likelihood of success.

Specific causal factors and their relative importance vary widely among species and programs making it difficult to identify general trends associated with success (Wolf et al. 1998). Griffith et al. (1989) used a comparative approach to test for general patterns underlying the success or failure of bird and mammal translocations based on the following predictors of success: (1) species' taxonomic class, (2) species' status, (3) habitat quality of the release area, (4) location of release relative to the species' historical range, (5) number of animals released, (6) program length, and (7) potential productivity of the species. Wolf et al. (1996) elaborated on this by considering the first five of these variables and incorporating an additional predictor: adult diet in the wild.

With available habitat and favorable conditions, reintroductions can be dramatically successful using relatively few individuals. For example, the burrowing bettong (*Bettongia lesueur*) was reintroduced to Western Australia in 1992 using only 42 individuals; by 1999, the population had increased to 260 (Short and Turner 2000). Similarly, after a 60-year absence, the western barred bandicoot (*Perameles bougainville*) was reestablished on the Australian mainland using only 14 founding individuals. These founders, released in 1995, were absent within two years, but successful juvenile recruitment by 1996 lead to a population of 116 in only three years. This increase in abundance was mirrored in a distribution expansion of nearly 50% from July 1998 to October 1999 (Richards and Short 2003). Even more strikingly, the reintroduction of 18 wood bison (*Bison bison athabascae*) into Canada's Mackenzie Bison Sanctuary in 1963 resulted in population growth that peaked at 2400 individuals by 1989 (Larter et al. 2000).

Successful reintroductions transcend species-specific effects of the released species and may reflect profoundly on an ecosystem scale. For example, the number of coyotes (*Canis latrans*) in Yellowstone Park's Northern Range dropped from 80 individuals in 12 packs to 36 in 9 packs after the grey wolf (*Canis lupus*) was reintroduced to the area. Within three years of the wolf release, 25–33% of annual coyote mortality was due to wolves, mean coyote pack size dropped from 6 to 3.6 adults, and the coyote population decreased by 55% (Crabtree 1998). Functionally, surviving coyotes increased their vigilance behaviors and altered their foraging patterns after the wolf reintroductions began (Switalski 2003). While the behavior of male bison (*Bison bison bison*) and elk (*Cervus elaphus*) was not affected, the vigilance of females increased significantly. Among elk, this was true for both females with calves (vigilance increased from 20% to 43%) and those without calves (vigilance increased from 12% to 31%) (Laundre et al. 2001). Changes in elk foraging patterns can even be detected at the plant community level. Elk pellet counts were significantly lower in habitats that wolves used frequently than in rarely used areas. Consequently, aspen sucker height was also significantly higher in areas of high wolf use.

Responses such as these suggest that the reintroduction of a single species can affect ecosystem responses on numerous trophic levels. Consequently, reintroductions, classically species-specific, potentially serve as a wider-ranging tool for ecosystem restoration.

In a Canadian context, the recent passing of the *Species at Risk Act* (SARA) necessitates recovery strategy planning for imperiled species (Government of Canada 2002). At the least, reintroductions will serve as a potential conservation tool for species restoration; at the most, reintroduction attempts could become a necessary component of species recovery strategies. Canada's assemblage of threatened species and socio-economic, political, and demographic characteristics suggest that the country may possess the requisite resources to qualify as a superior region for species reintroductions. In fact, most of the more than 700 reintroduction or translocation programs carried out every year around the world occur in North America (Griffith et al. 1989). A successful reintroduction program needs a holistic and truly interdisciplinary approach that integrates both biological and social sciences toward the goal of conservation (Miller et al. 1999); therefore, in this paper, we examine both ecological and social (nonbiological) aspects in the assessment of Canada's potential for species reintroduction programs. We define reintroduction as an attempt to establish a species in an area of its historical range where it has reached critically low numbers or has become extirpated. We include captive or a combination of wild-caught and captive animals in our definition. A translocation is a deliberate and mediated movement of wild-caught individuals or populations from one part of their range to another where conspecifics exist. Despite the growing number of fish, invertebrate, and plant reintroductions, this paper addresses only mammals (excluding otariids, phocids, and cetaceans), birds, reptiles, and amphibians, which represent 86% of global reintroductions to date (Stanley Price and Soorae 2003).

In this paper, we (1) review ecological parameters that maximize reintroduction success, (2) assess Canadian socio-political parameters that could affect reintroduction feasibility, (3) identify considerations for future reintroductions in Canada, and (4) identify Canadian species that would be biologically and sociologically favorable for reintroductions once threats are removed, habitat availability is secured, and funding is sufficient.

Ecological Considerations

Although valuational and organizational factors strongly influence species reintroduction success (Reading 1993; Yalden 1993), decisions about whether to use reintroduction as a conservation tool must be made on a case-specific basis after biological constraints imposed by life history have been considered (Seigel and Dodd 2002). Several reintroduction projects failed, at least in part, due to lack of attention to the biological requirements of the species (Beebee 1983; Berry 1986). Understanding the causes for the reintroduced species' decline and having suitable knowledge of the species' behavioral ecology and natural history will help ensure that we can realize the potential consequences of management decisions. Dodd and Seigel (1991) recommended that thorough knowledge of a species' life history requirements is a prerequisite to the adoption of any reintroduction strategy. Herein lies the irony: the rarest and most endangered species are often those that we know the least about. In this section, we examine life history characteristics that we consider are relevant for predicting superior candidates for reintroduction.

Species Status and Causes of Decline

Although one would expect reintroduced species to generally have global threatened or endangered status, this is often not the case. Beck et al. (1994) found that of the bird, mammal, reptile, and amphibian species that have been reintroduced worldwide, only 48% were globally threatened. This is surprising considering that one of the primary criteria for resorting to reintroduction as a conservation tool is an explicit conservation need; however, sometimes such a need is at a smaller scale, such as within a particular country, province, state, or region. Therefore, Canadian species we might consider as good candidates for reintroduction would likely be listed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) as Threatened or Endangered. This formal recognition of imperiled populations or species suggests that the need for conservation does exist, and it also affords released individuals heightened protection under SARA (Government of Canada 2002).

In contemplating a reintroduction program, the reasons for the species' decline should be one of the initial considerations. That is, what caused the species to become extirpated or reach such critically low levels that reintroduction is even being considered as a conservation tool? Understanding the major threats faced by a species is vital, as is the ability to remove, or to some extent control, those threats. Fischer and Lindenmayer's (2000) review of reintroduction programs revealed that none of the programs that acknowledged the cause of a species' decline but failed to remove it, succeeded. It is difficult and expensive to restore critical habitat, while more direct, human-induced impacts may be more easily ameliorated. For example, overharvest of a species through hunting or collecting can be controlled to some degree through policy, laws, enforcement, and public education. Likewise, introduced nonnative predators have led to the decline of many native species worldwide. Using an effective control program for introduced predators prior to releasing native individuals would ensure that the native species has the best chance of surviving to reproduce and establish a sustainable wild population.

Diet and Feeding Behavior

It is essential to understand the nutritional requirements of the reintroduction candidate, primarily to enable evaluation of food availability in the release area. This task is much easier for herbivores than for carnivores or insectivores, as surveying vegetation requires far less effort and yields more reliable results than estimating biomass of animal or insect prey. Bird and mammal reintroductions in North America, Australia, and New Zealand revealed that herbivores were more likely be successful than carnivores and omnivores (Griffith et al. 1989). Furthermore, grazing herbivores may be even more suitable for reintroduction relative to browsers, and especially carnivores and insectivores, as they generally have less complex food searching and handling behaviors. This minimizes or negates the need for pre-release training for prey identification, hunting skills, food-processing ability, and prey caching—an undertaking that can be expensive and has demonstrated limited effectiveness (Kleiman et al. 1986; Snyder et al. 1987). Additionally, a generalist feeder with its broad diet would be more likely to utilize a single, year-round habitat obviating the need for specialized seasonal habitats or food. Moreover, if the species is part of a captive-breeding program, a diet of plant material in captivity is normally simple and economical.

Habitat and Spatial Characteristics

One of the strongest determinants of reintroduction success is suitable habitat availability (Griffith et al. 1989; Wilson and Stanley Price 1994). When the decline of a species is primarily due to habitat loss or fragmentation, chances of successful reintroduction diminish. Unless this can be properly addressed (i.e., an adequate area of critical habitat remains or has been restored), the release of individuals into these areas is a fruitless exercise; therefore, we favor selecting a species that inhabits a relatively common, nonthreatened, or well-protected habitat.

For reintroduction, terrestrial species might have advantages over animals that spend all or part of their life cycle in an aquatic environment. Wholly or partly aquatic species often require a variety of seasonal habitats with unique features for breeding, birthing, hatching, or overwintering. This complicates the task of finding suitable release areas. Furthermore, terrestrial animals would presumably be less susceptible to the impacts of environmental contaminants and stochasticity. Semlitsch (2002) suggests that chemical contamination is a potential factor in amphibian declines and that local habitat degradation or alteration (e.g., filling or draining wetlands, channelization of streams, creation of impoundments) is probably the major cause of these declines.

For many reasons, the ranging behavior of a reintroduction candidate should be minimal. Migratory species should generally be avoided as the migration route must be learned. Although reintroduced whooping cranes (*Grus americana*) have been successfully taught a migratory route using ultralight aircraft, the endeavor is based on a successful large-scale, long-term international recovery effort (CWS and USFWS 2003). Such an approach should not be undertaken lightly. As well, a reintroduction program involving migrating animals would face the added exigencies of multi-jurisdictional management (e.g., during post-release monitoring) and protection (e.g., species protection can vary significantly among political subregions). Species that exhibit minimal dispersal and have small home range areas require less critical habitat and are easier to monitor after release. Moreover, for reintroductions with a captive-breeding component, naturally wider-ranging carnivore species experience higher infant mortality in captivity than those which have smaller home ranges in the wild (Clubb and Mason 2003).

Morphology and Behavior

Species morphology warrants serious consideration when identifying suitable candidates for reintroduction. Certain physical attributes would benefit the program's chances of success, while others may be disadvantageous. For example, a morphological design that accommodates easy fitting of a radio-telemetry transmitter would help ensure reliable post-release monitoring, which is crucial to the ultimate success of a reintroduction. If morphology precludes practical use of radiotelemetry, then the species' behavior should allow for easy observability. Alternatively, species possessing physical characteristics that are considered products of human demand (e.g., horns, antlers, attractive fur, bear gall bladders) should be avoided unless sufficient protection is ensured.

A disproportionate number of reintroductions involve charismatic megafauna. For example, globally, nearly half of all bird reintroductions have involved Strigiformes and Falconiformes, and 62% of mammal reintroductions have involved Carnivora and Artiodactyla (Wilson and Stanley Price 1994). This trend seems to be continuing. Despite their popularity for reintroduction, these groups may not be the most practical choice. Smaller body size might be a beneficial attribute for a reintroduction species. Smaller animals tend to breed prolifically and require less space, which means more individuals can be accommodated during captive breeding. Balmford et al. (1996) concluded that the intrinsic rate of population growth of captive species in zoos is inversely proportional to body mass. Similarly, for reintroductions, smaller animals can be transported more easily and in larger numbers to the release site.

Species behavior is another important consideration when contemplating a reintroduction. Timing and level of daily and annual activity likely influences a species' chance of survival in a novel environment. A nocturnal animal may have a reduced risk of experiencing negative human encounters. Nonhibernators are not faced with the added challenge of finding a suitable overwintering site; however, a disadvantage of winter activity in many regions of Canada is the difficulty of surviving extreme weather and food scarcity. For example, ungulate overwinter survival depends on many factors such as winter severity and whether or not the animals have the necessary fat reserves to meet increased thermoregulatory demands and to offset nutritional stress caused by low quality forage (Sime et al. 1998).

Gregarious species with a low incidence of conspecific aggression and relatively simple social behavior would also be highly desirable for reintroduction. Individuals can benefit from group living by learning important survival strategies from conspecifics. When reintroducing African wild dogs (*Lycaon pictus*), Mills (1999) found that mixing wild and captive-bred individuals in a group led to increased success. Another advantage of gregariousness is group vigilance as an anti-predator strategy, which allows individuals to focus on feeding, drinking, reproducing, and ultimately, surviving.

It would also be prudent to avoid species with complex social behaviors for captive-breeding reintroductions. Unique behaviors must be learned before release, requiring intensive, often expensive, pre-release training regimes. Foraging, locomotory, anti-predator, and other behaviors that are essential for survival are more heavily dependent on learning and specific environmental experience among mammals and birds than among reptiles and amphibians (Beck et al. 1994). This is particularly important when captive breeding carnivores, as imprinting and close contact with humans should be avoided; it may also be necessary to use aversive therapy and devise ways of improving hunting skills (Yalden 1993). This, however, should not be an issue in translocation programs, as presumably, wild individuals are already experienced in their species-specific skills and social behaviors. Worldwide, translocations of birds and mammals were 75% successful, while captive-breeding reintroduction programs had only a 38% success rate. Of the captive-breeding reintroduction programs that incorporated pre-release training, 50% succeeded and only 32% had unsuccessful or indeterminate outcomes (Beck et al. 1994). This emphasizes the importance of equipping released animals with the necessary skills to survive and reproduce in the wild.

Reproduction and Population Dynamics

Theories predict that population persistence is more likely when there is a high rate of population increase (MacArthur and Wilson 1967). A higher reproductive rate is also an important factor that favorably influences the likelihood of establishing a new population (Crawley 1986; Ehrlich 1986); therefore, high reproductive potential of a reintroduced species, both in the wild and in captivity, can increase the chances of establishing a sustainable population if it leads to high recruitment. Hence, reproductive characteristics typical of r-selected species would, for the most part, be desirable.

There are many characteristics that maximize the reproductive success of a species. These include early sexual maturity and age at first breeding, short gestation or incubation period, large brood size, early weaning or fledging, short inter-brood interval, long reproductive life, and longevity. Some of these traits are more representative of r-selected species, while others are more typical of K-selected species. Griffith et al. (1989) found that reintroduced bird and mammal species that were early breeders with large broods, traits typical of r-selection, had a significantly greater chance of success than late breeders with small broods. Alternatively, Craig and Veitch (1990) found that a relatively long lifespan, typical of K-selection, increased the chances of a species' population becoming established.

Knowledge of the population dynamics of the species being considered for reintroduction is important to understand how the wild population might respond under a variety of influences. High recruitment rates and fecundity are typical of good productivity, and possessing these characteristics could lead to a higher intrinsic rate of population growth. This would facilitate more rapid achievement of the target population size in terms of sustainability. Additional factors that can profoundly influence the dynamics of a reintroduced population, such as natural mortality, must also be considered. For example, if factors such as predation and disease seriously impact the population, then predator control or vaccination may be required prior to release.

For birds, there exist many manipulative techniques for enhancing reproductive success. These include egg transfer from successful to unsuccessful populations, artificial incubation, stimulation of replacement egg clutches, cross-fostering between related species and individuals, and artificial insemination and imprinting (Saint Jalme 2002).

Socio-political Considerations

Historically, reintroduction programs have focused on biological parameters, yet people’s values and perceptions of wildlife are very important to the success of a species’ reintroduction. Reintroduction programs involving endangered species rarely succeed if they do not actively consider and incorporate the values, attitudes, behaviors, and desires of the local people (Reading and Kellert 1993). The differential influences of education, experiences, culture, demographics, and social institutions produce differential values (Johnson et al. 1996). With its stable political system, thriving economy, and generally positive attitudes of its people towards wildlife, Canada could serve as a suitable location for a species reintroduction program.

Demography and Socio-economics

Canada is a very large country with an extremely low human population density (3 individuals/km^2) compared to other countries (e.g., Bangladesh: 1034 individuals/km^2; Rwanda: 325 individuals/km^2) (United Nations 2001). Like many countries, Canada’s population is concentrated in specific regions, resulting in large areas with few human inhabitants. With an annual growth rate of only 4% (including immigration), Canada’s population density is projected to remain low on a global scale (Statistics Canada 2002).

Traditionally, it has been reasoned that people living in urban areas are more likely to hold greater protectionist attitudes and values toward wildlife than those living in rural areas. With 80% of its inhabitants in urban areas (Statistics Canada 2002), Canada is likely to hold more protectionist wildlife values than countries with greater rural populations. Moreover, greater than 40% of rural Canadians enjoy observing or caring for wildlife around their homes (Environment Canada 1999). This suggests that rural Canadians may express more protectionist opinions than traditionally expected. The number of both urban and rural Canadians participating in nature-related and wildlife viewing activities is increasing (Environment Canada 1999), which may lead to increased support for reintroduction programs over time.

Socio-economic status also shapes people's values of wildlife. Those with more formal education and higher incomes tend to display greater naturalistic and ecological values and attitudes toward wildlife (Reading and Clark 1996). Globally, Canada ranks high in economic status and education with 53% of its adult population having post-secondary education (Statistics Canada 2001). The high proportion (85%) of Canadians participating in nature-related activities (Environment Canada 1999) reflects this elevated national socio-economic status. Furthermore, leaving a healthy environment for future generations is the primary issue by which Canadians define their identity (Ekos Research Associates 2002).

Canada's demographics and socio-economic situation suggest it is a country with protectionist views of wildlife indicating that there is likely widespread support for species reintroductions. Evidence for such support can been seen in many reintroductions such as that of the swift fox (*Vulpes velox*) and Vancouver Island marmot (*Marmota vancouverensis*). Successful volunteer and charitable organizations were formed for both species to aid reintroduction efforts through fundraising and public awareness initiatives; however, not all species reintroductions receive such widespread support from citizens. For example, support for the proposed reintroduction of wolves in New Brunswick was lowest among hunters, those who fear wolves, and those with low levels of formal education (Lohr et al. 1996). The primary reason for the opposition was the belief that the availability of deer and moose for hunting would decline (Lohr et al. 1996). Although many social factors suggest that Canada is a good location for species reintroductions, addressing social perceptions and economic impacts of reintroductions remains important.

Political Stability and Long-term Support

Reintroductions tend to be lengthy, costly, and complex programs, which makes long-term government commitment and stability important. The reintroduction of the swift fox in southern Canada serves as an excellent example of this. The program required twenty years of commitment and financial support from individual organizations, universities, and governments, and is still ongoing (Moehrenschlager and Somers 2004). Like many programs, this reintroduction shifted over time from a private initiative to a university project and finally to the responsibility of government agencies (Breitenmoser et al. 2001). The commitment of government agencies is usually required at some point in a reintroduction program, despite the fact that nongovernment organizations (NGOs) often provide the initial impetus. In less politically and economically stable countries, NGOs may initiate reintroduction programs, but frequent shifts in political power and ideology may hinder requisite long-term government support. Canada's stable democracy allows for long-term cooperative approaches between all levels of government and stakeholders. Tensions occasionally exist between Canada's provincial and federal agencies regarding the management of species at risk particularly if the distinction between lead and supporting agencies is unclear. Such conflicts can result in greater funds for species at risk as the different levels of government attempt to display their conservation leadership.

Geopolitical Simplicity

When more than one country becomes involved in coordinating a reintroduction program, an already complex process becomes even more complicated. Breitenmoser et al. (2001) observed that geopolitical and cultural differences are especially challenging in Europe and Africa. For example, the Alps are a well-suited range for the restoration of large carnivore populations (e.g., see Breitenmoser et al. [2000]—Eurasian lynx [*Lynx lynx*], and Boitani [2000]—grey wolf). However, the Alps fall within seven countries that speak many languages, and each country has unique legislative and wildlife management systems. The distribution of large carnivores is relatively limited in each country making cross-border cooperation essential, but government and cultural differences result in a very complicated and slow process (Breitenmoser et al. 2001). Canada is one of the world's largest countries encompassing vast areas under the management of relatively few political entities. This geopolitical simplicity allows for more geographically extensive initiatives, often a necessity for reintroduction programs.

Legislation

Successful reintroductions have occurred in developing countries (e.g., in Mauritius, the pink pigeon [*Columba mayeri*] [Swinnerton et al. 2000] and Mauritius kestrel [*Falco punctatus*] [Nicoll et al. 2004]). However, less developed countries frequently lack legislation and enforcement personnel to protect their endangered species; consequently, reintroduction may be risky. For example, the African wild dog reintroduction to Matetsi Safari Area in Zimbabwe failed, with five of the nine released animals being found in a local farmer's butchery (Woodroffe and Ginsberg 1997). Throughout southwestern Zimbabwe, free-ranging African wild dogs face high levels of human-caused mortality (adults: 88%; pups: 63%) with much of the mortality resulting from snaring and shooting (Woodroffe et al. 2004). Canada has implemented laws that aim to safeguard threatened wildlife and ensure their survival. Comparable legislation in the United States, the *Endangered Species Act* (ESA), elicits fear and hostility among certain sectors (Reading and Clark 1996). These public concerns are often based on real or perceived fears of the restrictive components of the ESA, negative attitudes towards wildlife in general, and real or perceived effects of past recovery programs (Reading and Clark 1996). To minimize such negative reactions, Canada's SARA is based on a paradigm of consultation and cooperation with stakeholders (Government of Canada 2002). While critics may argue that this could result in weaker species protection, this approach will hopefully alleviate problems that often arise from an all-encompassing legislation by addressing the protection of threatened wildlife on a species-specific basis.

Economics

Reintroduction programs generally require extensive funding over long periods. For example, the annual cost of reintroducing California condors (*Gymnogyps californianus*) was estimated at U.S.$1 million (Cohn 1993). At program inception, the cost of the Mexican wolf (*Canis lupus baileyi*) reintroduction was estimated at U.S.$7 million over nine years (USFWS 1996). Economic factors are undoubtedly the reason that the great majority of reintroductions have occurred in North America, Europe, Australia, and New Zealand (Wilson and Stanley Price 1994). This is not surprising as less developed countries have many demands on their limited financial resources, which results in few available resources for reintroduction programs, especially over the long term. While it can be argued that program costs are lower in developing countries, Canada, with its healthy economy, has many sources from which significant funding may be obtained, such as government agencies, NGOs, industry, and individuals. Financial support for endangered species in Canada rose from CDN$26.2 million in 2001–2002 to CDN$46.7 million in 2002–2003, an increase of 77%, with the number of contributing organizations increasing from 196 to 282 (Environment Canada 2003).

Considerations for Future Canadian Reintroductions

Critical Habitat Determination

Across taxa, the primary determinant of reintroduction success is the availability of suitable habitat. The determination of critical habitat is paramount for endangered species, in general, and is particularly crucial for legislative purposes under SARA (Government of Canada 2002). Aspects of habitat scale and quality on a species-specific basis are essential for such determinations and subsequent strategies invoking stewardship or punitive measures; however, delineating habitat needs is inherently difficult within and among species because animal presence does not necessarily reflect habitat preference. Here we draw on our experiences with reintroduction programs we are involved in to illustrate some potential challenges for Canadian reintroductions.

The Vancouver Island marmot is critically endangered under IUCN and is the most endangered mammal in North America, but the high alpine meadows that constitute its primary habitat do not appear to be limited on Vancouver Island. However, habitat quantity on a coarse scale is not necessarily related to habitat quality. The interplay of forage, predator dynamics, and potentially remote effects, such as climate change or acid rain, also complicate critical habitat determination.

Approximately 70% of mixed-grass native prairie has been lost in North America, limiting the total available recovery habitat for a prairie specialist such as the swift fox (Moehrenschlager and Sovada 2004); however, the combination of prey availability and intra-guild competition determines the suitability of seemingly pristine areas. Swift fox presence may be as dependent on the relative density of coyotes and red foxes (*Vulpes vulpes*) as on habitat-linked prey dynamics (Moehrenschlager et al. 2004).

The loss or degradation of wetland or riparian areas has been one of the factors leading to precipitous declines of the whooping crane (White 2001) and the northern leopard frog (*Rana pipiens*) (Seburn and Seburn 1998), yet for both species, coarse scale habitat availability far outweighs respective areas of occupancy. Indeed, lack of occupancy can be due to the need for habitat that is sufficiently suitable for all seasons and developmental stages. For example, the presence of a wetland does not mean that sufficient crustaceans will be available for whooping crane adults, or that dragonflies, which are crucial for whooping crane young, will be sufficiently abundant. Riparian areas that are conducive to the dispersal of reintroduced leopard frog metamorphs are sometimes insufficient for population sustainability, because inadequate pH, temperature, or dissolved oxygen levels in overwintering ponds can lead to dramatic mortality before the breeding season (Kendall 2000).

Critical habitat determination can be challenging for endangered species in general, but may be disproportionately more difficult for reintroduced species which may have individuals or subpopulations in marginal habitat. When a species is in decline, such as the Vancouver Island marmot, past extinction events may confound habitat availability/use assessments on a colony level. On an individual level, it is difficult to ascertain whether habitats selected for on a home range scale should be protected if individual survival rates are low; after all, lack of high quality habitats could be a primary determinant of such survival rates. Released animals can be poor indicators of habitat quality as they learn to adapt to a novel environment, attempt to find adequate resources, or find breeding partners. For example, translocated swift foxes had significantly greater daily distance travel rates than concurrently radio-tracked conspecifics for up to 50 days after release (Moehrenschlager and Macdonald 2003). In the case of swift foxes or leopard frogs, data from extant populations in other regions could be beneficial in determining critical habitat needs, but if the specific needs of species such as the Vancouver Island marmot or whooping crane are unknown, experimental reintroduction release probes may be necessary to determine their habitat needs.

Challenges of IUCN Reintroduction Guidelines from the Canadian Perspective

Clearly, reintroductions have the highest likelihood of success if the IUCN guidelines for reintroductions are closely adhered to and we are strong proponents of those guidelines. The three primary tenets of the guidelines are that (1) the factors of decline need to be understood before reintroductions are attempted, (2) release candidates should be taxonomically similar to those that existed at the release site, and (3) reintroductions should generally be avoided outside the historic range of a species (IUCN/SSC Reintroduction Specialist Group 1998).

As species extinction rates accelerate, conservationists will increasingly encounter situations where species slip to near extinction without the necessary data being acquired to explain the original factors of decline. This may already be true for the Vancouver Island marmot and regional populations of the northern leopard frog in Canada. If the factors of decline are not understood and, therefore, cannot be removed before reintroductions are implemented, difficult management decisions may need to be made. Following IUCN recommendations would mean that reintroductions for such species should not be attempted, which might result in regional extirpation or global extinction. Such an approach may be politically, socially, and ethically difficult to defend in a wealthy country such as Canada. The question remains whether reintroductions can succeed without an understanding of historic population decline factors.

The reintroduction of the swift fox proceeded without a complete understanding of the factors responsible for its historic decline. Although native prairie habitat loss, poisoning, trapping, and loss of bison biomass for scavenging have been implicated as potential factors in the species' decline (Herrero et al. 1991), data justifying these assumptions were absent. While these individual parameters or their cumulative effects explain disappearances of the species in some areas of its range, they do not adequately predict its presence and absence in other regions (Moehrenschlager et al. 2004). Nevertheless, reintroductions founded through adaptive management and later improved through sound science resulted in success. As part of a national reintroduction program, 942 swift foxes were released in Canada from 1983 through 1997. Translocated foxes that were monitored from 1994 to 1998 had higher survival rates than previously monitored captive-bred foxes and similar survival rates to resident, wild-born foxes (Moehrenschlager and Macdonald 2003). In 1997, the Canadian population was estimated at approximately 192 and 89 in respective subpopulations. Of foxes live-captured between 1994 and 1998, 88% were born in the wild within the reintroduced population (Moehrenschlager 2000; Moehrenschlager et al. 2003). By 2001, the number of individuals trapped on replicated townships increased significantly, and the known distribution of swift foxes increased three-fold since the previous census. While the population was previously fragmented in Canada and sparse in Montana, it is now connected because gaps within the known distribution are smaller than maximum dispersal distances of this species (Moehrenschlager and Moehrenschlager 2001). Swift fox reintroduction to Canada is arguably the most successful reestablishment of a nationally extirpated carnivore in the world, but the fact remains that the original factors of decline were poorly understood.

At times, the protection of taxonomic similarity between those species historically present and the release candidates may need to be sacrificed. In the case of the Vancouver Island marmot, individuals from the Mount Washington colony show great genetic distinctiveness from the rest of the population, which arguably should be preserved over time; however, the wild population fell to about 40 individuals before captive-breeding showed success, which necessitated the interbreeding of these geographic variants to meet demographic goals.

The IUCN recently projected that climate change will become a greater contributor to species extinctions than exploitation-related habitat use. As such, suitable, and perhaps even critical, habitat for some species could conceivably move beyond the boundaries of their historic ranges. Competitors of species at risk may be able to move into previously inaccessible areas. For example, red foxes are moving northwards into circumpolar regions, which is driving Scandinavian Arctic foxes (*Alopex lagopus*) to extinction (Tannerfeldt et al. 2003). Population viability analyses and recovery strategies frequently plan for population persistence over at least 100 years, but reintroduction-related programs in Canada may face the difficult dilemma of dealing with releases or metapopulation management outside the historic range of endangered species.

Defining Reintroduction Success

Upon reviewing 180 case studies on animal relocations which spanned 20 years, Fischer and Lindenmayer (2000) determined that the following were necessary components of relocation programs: (1) more rigorous testing for the appropriateness of the relocation approach, (2) establishment of widely-used and generally accepted criteria for judging success or failure, (3) better monitoring after relocation, (4) better financial accountability, and (5) greater effort to publish the results of relocations, even if unsuccessful.

In the planning phase, or as reintroductions show signs of success, the question continually arises as to how many individuals need to be restored for the program to be deemed successful (Pyare and Berger 2003). Proposed measures of success for reintroductions vary widely, and include (1) breeding by the first wild-born generation, (2) establishing a three-year breeding program in which recruitment exceeds adult death rate, (3) establishing an unsupported wild population of at least 500 individuals, and (4) establishing a self-sustaining population (Seddon 1999).

Debates ensue about the minimum effective population size required for population sustainability. Estimates of minimum viable population size depend largely on genetic parameters. Some argue that 50 individuals are sufficient to avoid short-term deleterious effects of inbreeding depression (Franklin 1980; Soulé 1980). Others believe that 500 is sufficient to maintain genetic variability in quantitative characters (Reed and Bryant 2000), while some believe that 1000–5000 individuals may be a safer number to strive for (Lynch and Lande 1998). If supportive breeding is used to supplement wild populations, the effective wild population size needed to prevent inbreeding depression decreases, but the variance effective size, which represents a minimal loss of heterozygosity, can potentially increase (Ryman et al. 1995).

Population viability is not solely dependent on population numbers but also on the cumulative genetic diversity of released individuals. Reintroductions of the Guam rail (*Rallus owstoni*) showed that the captive management and choice of release candidates can profoundly affect genetic diversity in the reestablished population (Haig et al. 1990). Pairs chosen to maximize allelic diversity, founder genome equivalents, or founder contribution to the population resulted in a more genetically diverse release population than pairs that were chosen randomly or based on maximum fecundity (Haig et al. 1990). Just as the effects of inbreeding depression differ drastically between populations according to their phylogeny, mating systems, and connectivity, genetic effects on the population viability of reintroduced populations remain unclear. The fact that the Mauritius kestrel recovered from 4 individuals to a population exceeding 700 birds reveals that some populations can exhibit viability despite having a small number of founders (Groombridge et al. 2001). Over time, minimum viable population estimates that are family-, or at least order-specific, should be determined to adequately assess reintroduction success.

Finally, reintroduction goals should not be restricted to population size assessments. Instead, we suggest incorporating all components of IUCN Red List assessments. Like current recovery objectives for the Mexican wolf (Paquet et al. 2001), goals should be outlined in terms of desired population trends in specific time frames, extent of occurrence, and areas of occupancy. Over time, such objectives could be further refined to address additional aspects such as genetic diversity, connectivity, and disease prevalence.

Potential Canadian Reintroduction Candidates

Reintroduction is an intensive and costly conservation tool that should be utilized when all other conservation options have been exhausted. In this paper, we cannot evaluate on a species-specific basis whether the factors of decline are well known, if sufficient habitat is available, or whether sufficient funds would be allocated, and we suggest that answers to these questions must precede any reintroduction. Should these aspects be resolved, issues will arise regarding the biology of the species and societal attitudes toward proposed reintroductions. We conclude by identifying Canadian animals that would be potential reintroduction candidates when the other reintroduction assessment parameters have been identified.

Since recommendations are made with conservation in mind, potential species have been selected from those designated as Threatened or Endangered by COSEWIC (COSEWIC 2003). In Table 1, we list examples of Canadian species that have been involved in conservation reintroductions and some of their relevant life history characteristics. Our purpose was to examine variables reviewed in this paper to determine how well characteristics of these species adhere to our recommended criteria. The citation for all national status designations is COSEWIC (2003) and all global status designations is IUCN (2004). All valuational and natural history information was taken from Nowak (1999) for mammals, del Hoyo et al. (1992, 1994, 1996, 1997, 1999, 2001, 2002, 2003, 2004) for birds, Hutchins et al. (2003a) for amphibians, and Hutchins et al. (2003b) for reptiles.

Table 1. Examples of Canadian reintroductions and translocations. Single values represent means and ranges refer to average minimum and maximum values. Status: EN=endangered, TH=threatened, LR=lower risk, LC=least concern, SC=special concern, NAR=not at risk, NE=not evaluated, VU=vulnerable.

Species	*Status*	*Mass*	*Feeding*		*Sexual maturity*	*Gestation/ incubation*	*Brood size*	*Lifespan*	*Migrate*	*Hibernate*	*Group-living*
	COSEWIC/ IUCN	*(kg)*	*type*	*behavior*	*(months)*	*(days)*		*(years)*			
MAMMALS											
Carnivora: Canidae											
Swift fox *(Vulpes velox)*	EN/LR	2–3	carnivore	hunter	10	52	3	6–8	no	no	no
Carnivora: Mustelidae											
Newfoundland pine marten *(Martes americana atrata)*	EN/LR	1	carnivore	hunter	15–24	28	3	6–8	no	no	no
Fisher *(Martes pennanti)*	NE/LR	1–5	carnivore	hunter	12	30	3	10	no	no	no
American badger *(Taxidea taxus jeffersonii)*	EN/LR	4–12	carnivore	hunter	4–12	210	3	14	no	no	no
Sea otter *(Enhydra lutris)*	TH/EN	15–45	carnivore	hunter	48–72	120–180	1	15–20	no	no	no
Artiodactyla: Bovidae											
Plains bison *(Bison bison bison)*	NE/LR	500–800	herbivore	grazer	36	285	1	20	yes	no	yes
Wood bison *(Bison bison athabascae)*	TH/LR	600–900	herbivore	grazer	36	285	1	20	yes	no	yes
Artiodactyla: Cervidae											
Woodland caribou *(Rangifer tarandus caribou)*	TH/LR	130–180	herbivore	grazer	28	228	1	< 5	yes	no	yes
Elk *(Cervus elaphus)*	NE/LR	230–320	herbivore	grazer-browser	24–48	250	1	15	no	no	yes
Rodentia: Sciuridae											
Vancouver Island marmot *(Marmota vancouverensis)*	EN/EN	3–7	herbivore	browser-grazer	24	30–32	4	< 10	no	yes	yes

Table 1. Examples of Canadian reintroductions and translocations. Single values represent means and ranges refer to average minimum and maximum values. Status: EN=endangered, TH=threatened, LR=lower risk, LC=least concern, SC=special concern, NAR=not at risk, NE=not evaluated, VU=vulnerable (cont'd).

Species	*Status*	*Mass*	*Feeding*		*Sexual maturity*	*Gestation/ incubation*	*Brood size*	*Lifespan*	*Migrate*	*Hibernate*	*Group-living*
	COSEWIC/ IUCN	*(kg)*	*type*	*behavior*	*(months)*	*(days)*		*(years)*			
BIRDS											
Passeriformes: Laniidae											
Eastern loggerhead shrike *(Lanius ludovicianus migrans)*	EN/NE	< 1	insectivore	hunter	12	13–16	4–7	< 11	yes	no	no
Strigiformes: Strigidae											
Burrowing owl *(Athene cunicularia)*	EN/LC	< 1	carnivore	hunter	12	28	5	< 9	yes	no	no
Falconiformes: Falconidae											
Peregrine falcon *(Falco peregrinus anatum)*	TH/LC	1	carnivore	hunter	36	28	2–6	< 13	yes	no	no
Anseriformes: Anatidae											
Trumpeter swan *(Cygnus buccinator)*	NAR/LC	9–13	herbivore	grazer	36	33–37	5	< 24	yes	no	no
REPTILES											
Testudines: Trionychidae											
Spiny softshell turtle *(Apalone spinifera)*	TH/NE	7–11	carnivore	hunter	96–120	82–84	4–32	< 50	no	yes	no
AMPHIBIANS											
Anura: Ranidae											
Oregon spotted frog *(Rana pretiosa)*	EN/VU	< 1	insectivore	hunter	36	14–21	650	< 4	no	yes	no
Northern leopard frog *(Rana pipiens)*	SC/NE	< 1	insectivore	hunter	36	7–21	3500	4	no	yes	no

Mammals

Artiodactyls and carnivores are charismatic species with notable public appeal; hence, it is far easier to garner support for them than for smaller, lesser-known species (Westman 1990). Artiodactyls listed by COSEWIC include the woodland caribou, (*Rangifer tarandus caribou*), Peary caribou (*R. t. pearyi*), and wood bison. Of these, only the wood bison is globally listed. Overhunting was the original cause for most artiodactyl declines, and it is a threat that may be relatively easily ameliorated. The highly successful wood bison reintroduction into the Northwest Territories can attest to this (Larter et al. 2000).

Many carnivores hold a similar public popularity as artiodactyls. Numerous species, such as the giant panda (*Ailuropoda melanoleuca*) and grey wolf, also project powerful cultural or symbolic values (Johnson et al. 1996). The swift fox and grey fox (*Urocyon cinereoargenteus*) are the only mid-sized carnivores listed by COSEWIC as Endangered and Threatened, respectively. Despite the failure of most carnivore reintroductions, the swift fox is considered to be one of the most successful reintroductions of a nationally extirpated carnivore in the world (Moehrenschlager and Somers 2004).

Perhaps the reintroduction of small carnivores, like mustelids, would have less public opposition than that of larger species, although this could be argued, in that public opinion towards the black-footed ferret (*Mustela nigripes*) reintroduction into Montana was less than favorable. However, in this case, it was not so much the reestablishment of the ferrets themselves that was opposed as the associated need to protect their primary prey, the black-tailed prairie dog (*Cynomys ludovicianus*), a species that elicits strong negative attitudes from local farmers (Reading and Kellert 1993). In a survey conducted in Britain, 81% of the respondents were prepared to pay at least £5 (> CDN$10) to support conservation of the Eurasian otter (*Lutra lutra*) (White et al. 1997). Also, in Britain, 65% of gamekeepers, 64% of farmers, and 89% of the general public supported the reintroduction of the pine marten (*Martes americana*) into England (Bright and Halliwell 1999). Bright (2000) reported that a higher proportion of mustelid species and subspecies are threatened compared to other mammal species (mustelids: 25/65 [38%]; all mammals: 647/4327 [15%]). In Canada, several mustelids are listed by COSEWIC, with the American badger (*Taxidea taxus* ssp.), Newfoundland marten (*M. a. atrata*), and wolverine (*Gulo gulo*) designated as Endangered, and the ermine subspecies on the Queen Charlotte Islands (*Mustela erminea haidarum*) and the sea otter (*Enhydra lutris*) listed as Threatened. In addition, the wolverine is listed globally as Vulnerable, and the sea otter is listed as Endangered. In Canada, there have already been several successful reintroductions of Newfoundland martens (Slough 1994) and sea otters (Love 1992).

The ermine shows outstanding potential for reintroduction. It is a nocturnal species with a very small home range, it inhabits a wide variety of habitats, and it has exceptional reproductive potential (e.g., females mature at 2–3 months and can produce up to 18 young annually). Moreover, the ermine’s efficiency in eradicating rodents makes them potentially valuable to humans. Badgers also show promise for reintroduction, being nocturnal, strongly territorial, long-lived, and producing young that wean at 6 weeks and quickly disperse thereafter. Like the ermine, they are also valuable to humans due to their rodent control capabilities. Badgers are also remarkable burrowers, and despite being considered a nuisance by ranchers, they provide shelter for other wildlife (especially the endangered burrowing owl [*Athene cunicularia*]).

Balmford et al. (1996) suggested that bats, being fast-breeding and social, may be good candidates for a cost-effective captive-breeding program. The pallid bat (*Antrozous pallidus*) is the only chiropteran that is listed nationally as Threatened and which may prove suitable for reintroduction. Pallid bats are nocturnal hibernators, highly social, long-lived, and prolific breeders.

Birds

The IUCN/SSC Reintroduction Specialist Group (1993) reported in their database, which documents *ex-situ* reintroductions worldwide, that birds were involved in 45% of the 150 reintroduction projects. Birds are popular with the public, and the preponderance of bird over mammal reintroductions is most likely due to the ease of manipulating or fostering eggs (Ounsted 1991).

There are many nationally listed passerines, however, their highly migratory nature generally renders them inappropriate for reintroduction. Ounsted (1991) suggested that bird reintroductions proven most likely to succeed are those which involve large and readily observed species, often for which humans hold a special affinity. Therefore, species belonging to the orders Strigiformes, Falconiformes, Gruiformes, or Galliformes may be most suitable.

Birds of prey account for almost half of all bird reintroductions; thus, raptor release techniques are probably better developed than for any other group (Wilson and Stanley Price 1994). In Canada, reintroductions of peregrine falcons (*Falco peregrinus anatum*) and burrowing owls have yielded encouraging results (Holroyd and Banasch 1990; Leupin and Low 2001). Endangered spotted owls (*Strix occidentalis caurina*) and threatened northern goshawks (*Accipiter gentilis laingi*) both inhabit old-growth forests, a quickly diminishing habitat that cannot be readily restored. Before considering reintroduction of these species, we must ensure a suitable area of critical habitat is available. Reestablishment of old-growth forest, however, would require a significant amount of time.

Alternatively, the endangered barn owl (*Tyto alba*) shows excellent potential for reintroduction. It is a sedentary, territorial species that can be found in a wide variety of habitats, and its diet is better studied than any other raptor. The barn owl, under the right conditions, is a very prolific species that breeds in its first year, produces up to 5 broods annually when prey is abundant, and lays as many as 16 eggs per clutch, of which most hatched chicks survive. Furthermore, the return of the barn owl could be a welcome sight to humans as the birds are highly effective at rodent control. Elsewhere in the world, such as in Sumatra, reestablishment of barn owl populations in oil palm plantations actually resulted in the replacement of second generation rodenticides (Heru et al. 2000).

The white-headed woodpecker (*Picoides albolarvatus*), which is federally listed as Endangered, also has potential for reintroduction. Both parents take part in brooding their 4–5 eggs for the very short 14-day incubation period, and they both acquire food for the chicks. White-headed woodpeckers also have high site fidelity and are primary cavity nesters, thereby providing homes for a variety of other forest species. Furthermore, white-headed woodpeckers are very tolerant of human disturbances, such as logging, provided that snags and stumps are left. The encouraging results from translocations of red-cockaded woodpeckers (*Picoides borealis*) in the United States (Carrie et al. 1999) highlight the reintroduction potential of Piciformes.

Reptiles and Amphibians

Balmford et al. (1996) suggested that, for captive-breeding programs, reptiles and amphibians are a currently neglected but potentially rewarding group that breeds quickly and at relatively low cost. Reptiles and amphibians are well represented by squamates and anurans, respectively, on the federal list of endangered and threatened species, although none are globally listed. Local support is crucial to any reintroduction program (Reading and Kellert 1993; Yalden 1993), and typically, squamates and anurans are perceived at best, in a neutral light, but more commonly in a negative light. This alone, may preclude initiation of a reintroduction program for many species within these orders.

Among reptiles, concern is generally shown for the larger and more charismatic or benign species (particularly tortoises) (Dodd and Seigel 1991). Because most amphibians lack parental care, they are also prime candidates for egg or larval translocation (Marsh and Trenham 2001). Moreover, anurans are largely prolific breeders and classically easy to propagate in captivity. Although anurans strongly adhere to the captive propagation criteria, causes for amphibian declines on a global scale remain poorly understood making *ex-situ* reintroduction programs challenging. Reintroduction success is probably unlikely for endangered species that are in decline for unknown reasons (Trenham and Marsh 2002), which is frequently the case for anurans. A striking example can be found in the northern leopard frog reintroduction into Alberta (Kendall 2002). Five years of captive-rearing led to the release of nearly 13,000 frogs; however, the behavior and survival of these reintroduced individuals remains unknown.

Canada represents the northern periphery of the range of most federally listed reptiles and amphibians; thus, the potential impacts of climate change may preclude them from reintroduction consideration. However, the threatened stinkpot (*Sternotherus odoratus*) stands out as a species with exceptional potential for reintroduction. Stinkpots can be found in a wide variety of habitats and have the most generalized diet of all kinosternids. They lay 1–9 eggs per clutch up to 2 times per year in the north (up to 4 clutches per year in the south). The pet trade and draining of swamps and ponds are the major causes of their decline, however, stinkpots often live in such high densities that they may be relatively unaffected by these threats.

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