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The *Endangered Species UPDATE* published this issue in partnership with the Defenders of Wildlife. Defenders of Wildlife is a leading nonprofit conservation organization recognized as one of the nation's most progressive advocates for wildlife and its habitat. Defenders uses education, litigation, research and promotion of sound conservation policies to protect wild animals and plants in their natural communities. Defenders has been a national leader in wolf restoration and protection for two decades. Founded in 1947, defenders is a 501(c)(3) membership organization with more than 430,000 members and supporters. It is headquartered in Washington, DC with field offices around the country.

Last November, Defenders of Wildlife hosted *Carnivores 2000*, a conference on carnivore conservation in the twenty-first century. The conference, held in Denver, Colorado, spanned the full taxonomic range of carnivores and addressed both biological and sociological issues impacting carnivore conservation in North America and abroad. Due to its overwhelming success, Defenders of Wildlife will hold a second conference, *Carnivores 2002*, in Monterey, California November 18 to 20, 2002. This conference will likewise encompass the full range of marine and terrestrial carnivores. For more information, visit www.defenders.org/carnivores2002. We hope to see you there.

The editors and staff would like to sincerely thank B. Moose Peterson/WRP for contributing photographs to this issue, and Rochelle Mason for contributing her series, *Focus on Nature*.

Endangered Species UPDATE

A forum for information exchange on endangered species issues
July/August 2001 Vol. 18 No. 4

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Cover: Black bear (*Ursus americanus*). Photograph by B. Moose Peterson/WRP.

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Carnivore Conservation in the Twenty-first Century

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The dawn of this new millennium is proving to be an interesting time for carnivores. In some areas predators seem to be rebounding after years of declining populations resulting from habitat loss and human persecution. For example, in this volume, Bangs et al. (p. 147) discuss the remarkable recovery gray wolves (*Canis lupus*) are making in the lower 48 states, and Sneed (p. 153) describes plans to restore wolves into the Grand Canyon ecoregion. Grigione et al. (p. 129) report signs of jaguars (*Panthera onca*), ocelots (*Leopardus pardalis*), and jaguarundis (*Herpailurus yaguarondi*) on both sides of the Mexico-U.S. border. Smeeton and Weagle (p. 167) describe successful reintroductions of the swift fox (*Vulpes velox*) into the great plains of North America.

In addition, new technologies have fostered improved research in both field studies and laboratory settings. Improvements in radio-telemetry, remote-sensing work, geographic information systems (see Gaillard p. 107 and Wydeven et al. p. 110), genetics (see Farrell p. 133 and Fascione et al. p. 159), and computer modeling (see Pitt et al. p. 103) have led to increased knowledge of the myriad factors affecting carnivore conservation. Finally, the rigorous application of ecological theory to conservation questions has lead to greater understanding of population dynamics and behavior within imperiled carnivore populations (see Powell p. 98; Zuercher p. 115; Fredrickson and Hedrick p. 164; and Cypher et al. p. 171). Understanding issues such as genetics, natural history, habitat needs, and predator-prey relationships will enable scientists to manage and conserve carnivore populations well into the future.

Unfortunately, not all carnivores are showing such positive signs of recovery (see Sorenson p. 120 and Hazell p. 142), and all predators still face innumerable threats. Habitat loss, competition with humans for resources and human persecution are some of the major issues with which wildlife managers must contend. Anti-predator sentiment remains strong in some arenas as well (see Bildstein p. 124; Jackson p. 138; Berg p. 186; Ford p. 190; Mason et al. 175; and Andelt p. 182).

Continuing to expand our knowledge of carnivores will be essential to combating these issues and finding new and innovative ways to enable humans to co-exist with healthy carnivore populations. Forums such as the Defenders of Willdife's conferences (*Carnivores 2000* Denver, CO November 2000; and *Carnivores 2002* Monterey, CA November 2002), and the *Endangered Species UPDATE* are vital to the continued success of carnivore conservation efforts. We thank the authors for their contributions to this edition and their work on carnivore conservation.

Theory and Methods in Carnivore Conservation

Who Limits Whom: Predators or Prey?

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Abstract
Animal populations can be limited by the availability of food (limited from the bottom of the food chain up), by predators (limited from the top of the food chain down), or by the interaction of these two processes. Whether carnivores, in particular, limit the populations of their prey, or are limited by those prey, has long been controversial and is critical to conservation of prey and predatory species. I return to the question because it is a good question that we wish to have answered and, in part, because it has no simple answer. Our knowledge of ecological communities has matured to the point that we can tease the question apart, look at its pieces, and find conditional answers. Predator-prey models suggest that predators may limit prey populations on one scale while food limits prey, and prey limit predators on another scale. Predator populations are always limited by the availability of their prey. Data from the literature supports action on two time scales.

Introduction
"Who limits whom: predators or prey?" is an old question that resurfaces regularly because, in part, it is a good question that we wish to have answered and, in part, because it has no simple answer. I address the question once again because our knowledge of ecological communities has matured to the point that we can tease the question apart, look at its pieces, and find conditional answers. These answers provide a basis for managing populations of predators and prey, for conserving endangered species, for understanding what kinds of population fluctuations we can affect and what kinds we can not, and for knowing where in an ecological system our management energies should be targeted.

The debate regarding who limits whom has been viewed from two predominant perspectives. The Bottom-Up perspective emphasizes that plant defenses against herbivores and herbivore defenses against predators are so strong that communities are limited at each level predominantly by food (Murdoch 1966; Polis 1999; Polis & Strong 1996; Strong 1992; White 1978). From this perspective, predators in species-rich, terrestrial communities rarely have large effects on the populations of their prey but, instead, are limited by them. Herbivore populations limit the populations of their predators because predator populations are small and herbivores are well adapted to avoid their predators.

In contrast, a Top-Down perspective was outlined by Elton (1927), then made more rigorous by Hairston et al. (1960) in their "The World is Green" paper, and has been explored extensively by Fretwell, Oksanen and coworkers (Fretwell 1977 ,1987; Oksanen et al. 1981; Oksanen and Oksanen 2000) as the Exploitation Ecosystem Hypothesis. From this perspective, plants are vulnerable to folivores but are seldom severely defoliated because of predation pressure on herbivores. The Exploitation Ecosystem Hypothesis is not limited to productive, species-rich systems and is most often tested using communities with vertebrate predators and prey.

Here, I ignore the abilities of plants to defend themselves and explore the effects on herbivores and predators of varying the productivity of plants that is available to herbivores. For convenience, I call this available productivity of plants "primary productivity."

Model predators and prey
Using a difference equation model similar to that used by Boyce and Anderson (1999), incorporating Type 3 functional responses (Holling 1959) and density-dependent population growth for both predators and prey, I developed a system with three trophic levels. The responses of this system to changes in primary productivity resemble the classic Top-Down trophic pattern (Figure 1a) except that the herbivore graph has a "hump" (Figure 1b). The model predicts that at high levels of primary productivity, predators limit their prey to lower population sizes than they do at intermediate levels of primary productivity. Similarly, when herbivores are not limited by predators, they limit the standing crop of vegetation. The model also predicts a most

basic characteristic of herbivore and predator populations: primary productivity establishes the conditions that allow herbivores, or herbivores and predators to exist. If primary productivity is not great enough, herbivore populations are not large enough to support predators and predator populations can not become established. Hence, predators are ultimately limited by their prey (Figure 2).

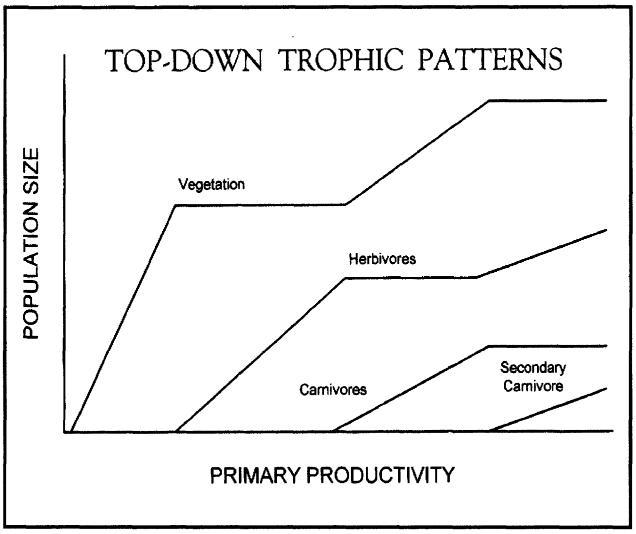

Figure 1(a). Classic Top-Down Trophic Patterns. At very low primary productivity, vegetation can not support herbivores. When primary productivity is capable of supporting herbivores, increased primary productivity leads to a population of herbivores that maintains a stable standing crop of vegetation. When primary productivity is high enough to support enough herbivores to support predators, the predators maintain a stable population of herbivores, and, therefore, the standing crop of vegetation increases. When primary productivity is capable of supporting a secondary carnivore, the population of primary carnivores remains constant, herbivore populations increase, and the standing crop of vegetation is constant once again. Population sizes are not drawn to scale so that carnivore populations can appear on the figure.

Admittedly, most predator-prey systems involve more than one prey and one predator species, yet some well-studied systems are quite simple. On Isle Royale in Lake Superior, for example, moose (*Alces alces*) have been basically the only prey for wolves (*Canis lupus*) for nearly five decades (McLaren and Peterson 1994). Similarly, muskoxen (*Ovibus moschatus*) generally are prey only for wolves, or wolves and Man (Gray 1987; Mech 1992). In the Upper Midwest states, wolves and white-tailed deer (*Odocoileus virginianus*) form the dominant predator-prey system (Mech 2000).

To introduce a bit more realism into my model, I varied the annual primary productivity and the annual predation rate on herbivores stochastically (Figure 3a, 3b). Such stochastic variation in primary productivity mimics annual variation due to weather. I modelled the stochastic variation in availability of prey as a variation in wolf carrying capacity, which could result from changes in snow conditions. For parameter values that lead to realistic relative densities of predator and prey, stochastic variation of primary productivity or predation rate, or both, affects all three trophic levels (Figure 3b). If stochasticity in primary productivity is superimposed on a long-term cycle, which might be caused by cyclical variation in climate, for example, the cycle may be seen in the predator and prey populations (Figure 3c). A researcher studying such a predator-prey system might conclude that interactions between predator and prey populations

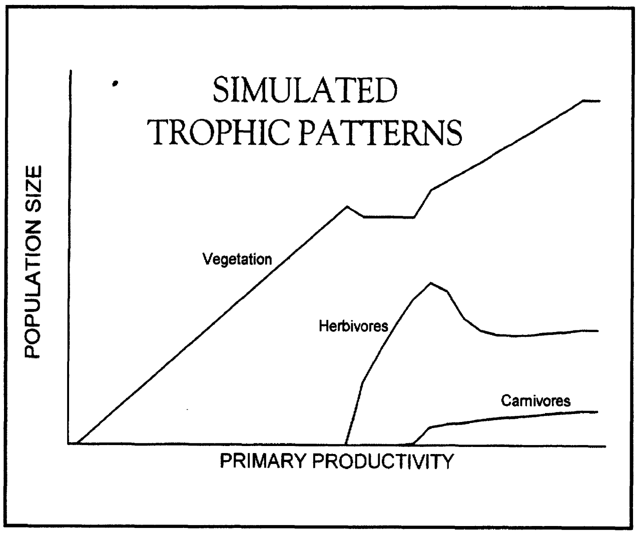

Figure 1(b). Simulated Top-Down Trophic Patterns mimic the classic pattern except that the graph of the herbivore population has a "hump" at levels of primary productivity that are capable only of supporting low carnivore population sizes.

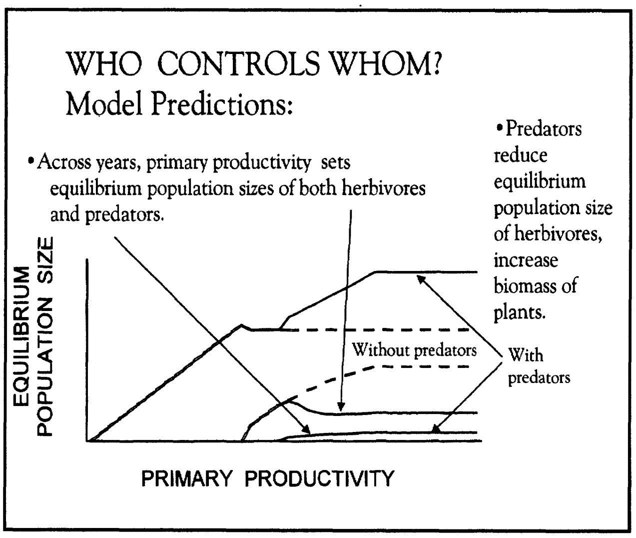

Figure 2. Who Limits Whom? Over long time spans, primary productivity dictates when herbivores and carnivores can maintain populations. Predators reduce long-term populations of herbivores below the level supportable by primary productivity and allow consequent increases in the standing crop of vegetation.

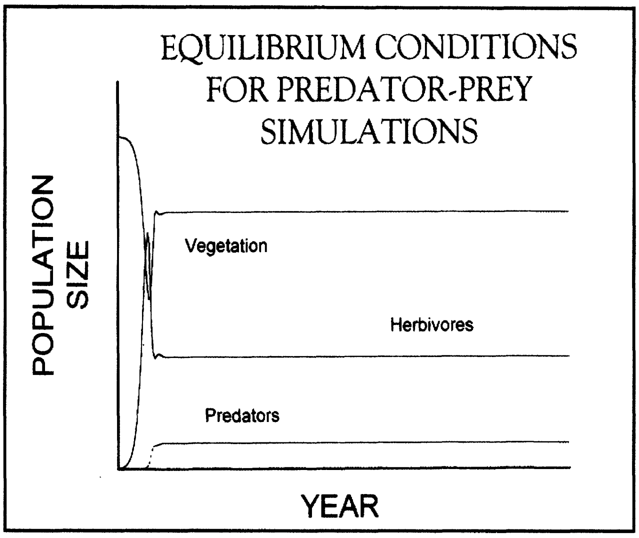

Figure 3(a). Sample population trajectories for the standing crop of vegetation and for population sizes of herbivores and predators.

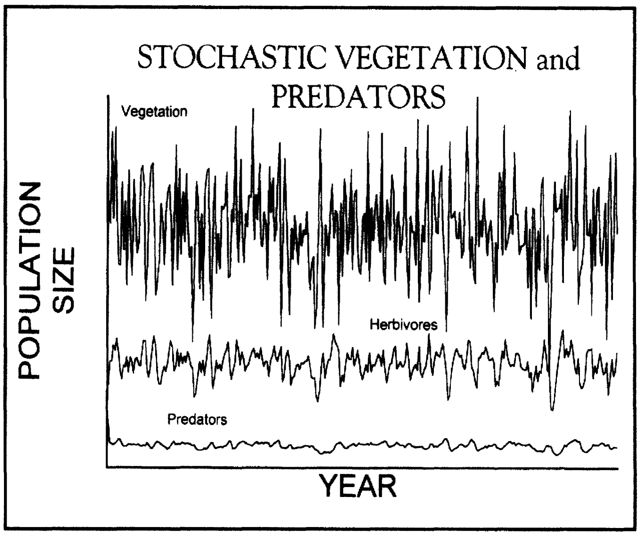

Figure 3(b). Trajectories for vegetation, herbivores and predators using the same model parameters used in Figure 3a except that primary productivity and predation vary stochastically. Year-to-year variation in herbivore population size depends more on variations in primary productivity during the preceding two years than on variations in predation pressure.

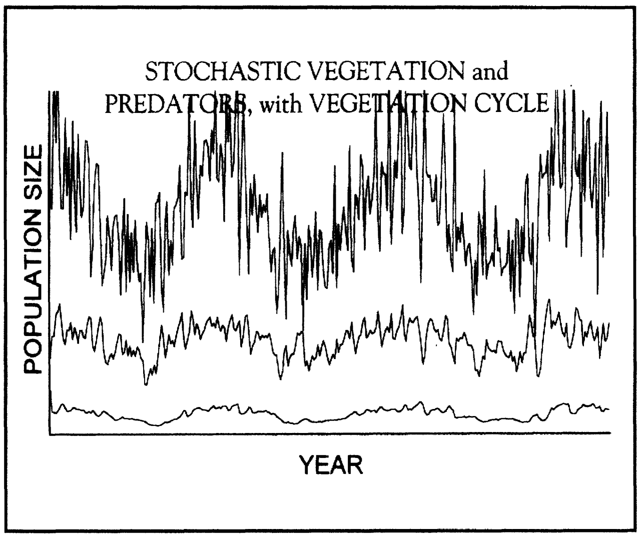

Figure 3(c). Same as Figure 3b except that a long-term cycle is imposed on primary productivity, such as might be caused by climate cycles. The cyclic variation in primary productivity caused cyclic variation in predator and prey populations might be mistaken as cycles caused by interactions between the population of predators and prey.

lead to population cycles when the cycles are actually driven from below by variation in primary productivity.

Consistent with the results of Boyce and Anderson (1999), I found that variation in primary productivity had a profoundly greater effect on year-to-year changes in herbivore populations than did variation in predation efficiency. The vast majority of the year-to-year variation in the model herbivore population could be explained by the productivity of vegetation during the preceding two years, while only about a quarter could be explained by the variation in predation rates. On a year-to-year basis, the model predicts that variation in food affects population dynamics of herbivores more than does variation in predation (Figure 4).

Stochastic variation of predation rate leads to an approximate 10% decrease in the long-term, average predator populations. This result is expected for predators with a Type 3 functional response. Because predation rate increases in a concave-downward fashion with high and increasing prey populations, variations in prey population size below the average population size lead to larger changes in predation than do variation in prey populations above the average. For the same reason, stochastic variation in primary productivity leads to a similar, though smaller, decrease in long-term, average prey populations. Stochastic variation in both primary productivity and predation, however, leads to a modest (ca. 2%) increase in prey populations, because of the decreased predator populations.

Who controls whom, predators or prey? The model predicts that each controls the other but the control acts at different levels, or on different scales. Ultimately, predator populations are controlled by the populations of their prey, which are, in turn, dependent on the primary productivity of their foods. When primary productivity can support an herbivore population large enough to support carnivores, however, predation then decreases the long-term, average size of the population of its prey (Figure 2). In contrast, on a year-to-year basis, variation in productivity of food causes more variation in herbivore population sizes than does variation in predation rates (Figure 4).

Real predators and prey

Do real populations of predators and prey act as the model predicts, with control acting on two scales?

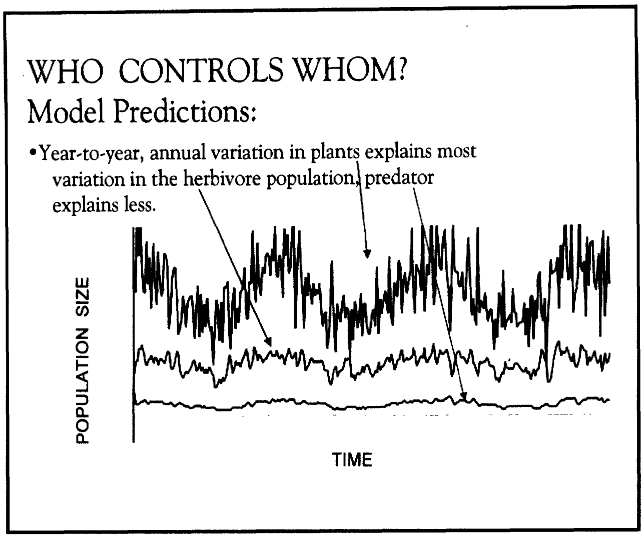

Figure 4. Who Limits Whom? Annual variation in plants explains most of the variation in the herbivore population, while variation in predation rate explains considerably less.

Results from field studies support this hypothesis.

The data for communities with vertebrates in arctic and north temperate ecosystems reviewed by Oksanen and Oksanen (2000) are consistent with control on two scales. Similarly, data for communities with herbivorous or predatory insects and small, vertebrate predators reviewed by Schmitz et al. (2000) are consistent with control on two scales.

McLaren and Peterson (1994) argued from data on wolves, moose and balsam fir (*Abies balsamia*) on Isle Royale, that predation by wolves controlled the moose population. In addition, moose browsed their major winter food, balsam fir, so heavily that production of fir was limited when the moose populations was high. Reanalysis of their data yields results consistent with model results: production of fir explains a significant amount of the year-to-year variation in the moose population, while year-to-year variation in the wolf population does not.

I studied fishers (*Martes pennanti*) and their prey, particularly porcupines (*Erethizon dorsatum*), in Michigan's Upper Peninsula in the 1970s (reviewed by Powell 1993). Fishers had been extirpated in the Upper Peninsula by the 1930s, and by the 1960s, porcupine populations had grown to previously unknown sizes. Fishers were reintroduced in the 1960s, and thus I studied a growing population. Porcupines had two limiting resources: winter food and safe winter dens. During winter, porcupines denned near their food trees. During the years of high population density, porcupines browsed hemlocks and white pines heavily during winters, often stunting tree growth. Where small stands of hemlocks dwindled and died, porcupines abandoned the associated dens. In my study area, most dens were hollow trees or hollow logs. Hollow logs with holes at both ends were acceptable to porcupines before fishers were reintroduced (Brander 1973; Brander and Books 1973; Powell and Brander 1977). As the fisher population grew, however, fishers killed porcupines in log dens with two entrance holes. A safe den near good winter food became in short supply. Fishers caused a significant decrease in the porcupine population (Figure 5).

Wehausen (1996) also documented a resurgent predator population causing the distinct decease of a formerly healthy prey population, in this case mountain lions (*Puma concolor*) causing the decrease of a mountain sheep (*Ovis canadensis*) population.

Thus, field data support the hypothesis that populations of real predators and prey exhibit controls on two scales. Most annual variation in populations of prey appears controlled by annual variation in food supplies. Yet, when primary productivity is high enough to support communities of prey and their predators, predators are capable of limiting prey populations to levels below the levels they would reach without predation. In addition, when prey populations do not support predators, they limit the abundance of their food.

Management implications

Reduction of a predator's population size to increase the size of a prey population is often considered by management agencies (Gasaway et al. 1983). The models presented here indicate that reducing the population of a predator will have little effect on the year-to-year fluctuations in the populations of its prey but may affect the long-term mean population size of prey. Figures 1, 2 and 3a illustrate how, when primary productivity is high enough, predators might limit prey from reaching the population size the food supply can support. Consequently, reducing the population of a predator can be one option to consider, with caution, when faced with a threatened or endangered population of a prey species.

Reduction of a predator population must be considered with caution for several reasons. First, Figure 3c illustrates that cyclic and stochastic fluctuations in primary productivity can lead to large fluctuations in an herbivore population that might be confused with a predator-prey cycle. The models presented here suggest that the year-to-year variation in abundance of herbivores are caused more by fluctuations in food supply than by changes in predation. The same may be true of long-term cycles of prey populations. Second, most predators prey on several prey species, which, in turn, have more than one predator, yet the models presented here deal with single predator and prey populations. The switching of prey by predators may prevent any one predator from limiting any one prey population (Murdoch and Oaten 1975), and the removal of one predator species may simply offer opportunity to

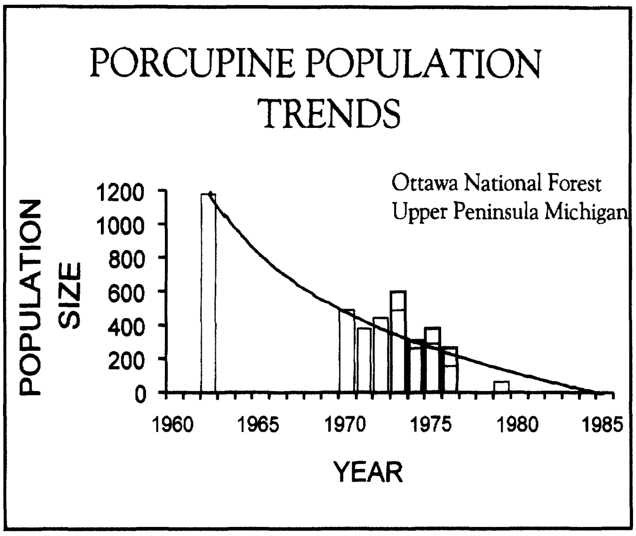

Figure 5. Changes in the porcupine population on the Ottawa National Forest. In 1962, porcupines lacked predators, and the population density was extremely high. Fishers were reintroduced in the mid-1960s, and by the late 1970s porcupines were rare. Double bars for porcupine populations in 1973-76 present data for both the study sites sampled in 1962 and 1970-73 and for an expanded number of study sites.

another. Finally, long-term control of a predator population when primary productivity varies stochastically may lead to extirpation of the predator, especially when the prey is also subject to control, as through harvest. If long-term harvest of a prey species exceeds 10% of its equilibrium population size, and primary productivity fluctuates as in my models, any control of the predator population could lead to its extirpation within decades.

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A New Approach to Understanding Canid Populations Using an Individual-based Computer Model: Preliminary Results

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Abstract
Ensuring the welfare of wild canid populations depends upon the ability to integrate species biology, the environmental aspects upon which those populations depend, and the factors controlling species abundance. Toward this end, we developed an individual-based computer model using Swarm to mimic natural coyote populations. Swarm is a software platform that allows the user to describe individual behaviors for all individuals, link those behaviors in each concurrent time step, and assemble behaviors and objects in a hierarchical framework. Our model stands apart from previous modeling efforts because it relies on field data and explicitly incorporates behavioral features, such as dominance and territoriality, as major determinates of species demography. Individual variation, such as status within territorial social groups and age-based reproduction are assumed, but assumptions typically associated with most demographic models are not needed. The eventual goal is to incorporate other environmental components such as prey abundance and/or competing carnivores. This type of model could also provide insights into potential management alternatives for when the gray wolf is removed from endangered status in Minnesota.

Introduction
Ensuring the welfare of wild canid populations depends upon the ability to integrate our best understandings of species biology, the environmental aspects upon which those populations depend, and the factors controlling species abundance (Gese et al 1989; Murray et al 1999). Previously, biologists and managers have relied upon insights provided by many analytical and computer models of animal populations. Canid populations, however, differ from other species because they are highly territorial and have a specific social structure, relatively low density (Knowlton 1972; Sillero-Zubiri and Gotelli 1995; Knowlton et. al 1999). Analytical models are not suited to include the individual characteristics that are critical to canid populations and past computer models have not incorporated territoriality and social structure (Connolly and Longhurst 1975). Toward this end, we developed an individual-based computer model using the Swarm modeling system to provide a better understanding of canid population dynamics. We use coyotes (*Canis latrans*) to parameterize the model for this exercise, but the model could easily be adapted to many other canid species with similar population structure. This paper is a preliminary summary of a model that will be presented in greater detail elsewhere (Pitt et al. in preparation)

The model
Swarm is a software platform that allows the user to describe individual behaviors for all individuals, link those behaviors in each concurrent time step, and assemble behaviors and objects in a hierarchical framework (Savage and Askenazi 1998; Railsback et al. 1999; SDG 2001). Our model stands apart from previous modeling efforts because it relies on field data with all population parameters derived from data sets and published papers, and explicitly incorporates behavioral features, such as dominance and territoriality, as major determinates of species demography (Connolly and Longhurst 1975; Knowlton et al. 1999; Pitt et al. 2001). Individual variation,

such as status within territorial social groups and age-based reproduction are specified and assumptions typically associated with most demographic models are not needed.

The coyote population model was divided into 100 packs and a collection of non-territorial animals. Each individual is characterized by sex, age, status, and pack membership. Pack size was not limited but the likelihood of subordinates increased with the number of animals in the pack. Individuals could change status or pack membership by dispersing from natal packs, replacing a dominant animal, or by moving to a pack from non-territorial status.

As with most animals, the probability of mortality increases with age. Mortality rates are higher for non-territorial animals than pack members (Gese et al. 1989). In addition, mortality rates increase with the density of non-territorial animals because they would potentially share a common area and the probability of encountering other animals would increase with density. Thus, density increases would either result in less food or an increase in the number of negative encounters.

Although subordinate coyotes occasionally produce offspring in natural populations, in our model only alpha females breed (Knowlton et al. 1999). The birth rate is based on a normal distribution with the mean based upon pack size. Few would disagree that the number of offspring produced is a function of the health of the animal. There has been, however, continued disagreement over what is a good indicator of health. Some evidence from captive coyote studies suggests that old (>8 years) animals will produce fewer offspring (Green et al. 2001). The most contentious argument is that litter size is a function of food supply or density (Crabtree and Sheldon 1999; Knowlton et al. 1999). Field evidence for and against this argument has been mixed (Gier 1968; Todd et al. 1981; Knowlton and Stoddart 1983; Windberg 1995; Gese et al. 1996). The most likely reason for these mixed results is that the number of offspring produced is a function of the food supply for that particular female. The food supply is a function of the food in the territory and the number of animals in the pack. Studies that have attempted to determine the relationship between offspring produced and density of food supply have looked at entire populations and large land areas (Gier 1968; Todd et al. 1981; Knowlton and Stoddart 1983; Windberg 1995; Gese et al. 1996). Thus, the relationship would only be observed if most packs were similar in size, food supply was homogenous across the landscape, and territories were identical in size. The likelihood of all of these factors being similar in one population would be low and extremely rare between populations, so this relationship would not be observed under most conditions on a population basis. For these reasons, we set the mean number of pups produced as a function of the number of pack members. In this model, territories are identical so we could ignore differences between territories.

The second part of this modeling exercise is the management model, which allows us to examine the effects of managing the population (Pitt et al. 2000). The management model combines the population model and a manipulation model so we can investigate the effect of removing individuals on population size and the resistance and resilience of the population. Herein, only random removal individuals will be considered.

Model output

We ran the population model under three management scenarios: no removal, pulse removal (a proportion of animals were removed in year five and the population then allowed to recover), or press removal (a constant proportion of animals were removed every year after year five). The populations were evaluated according to structure, the resistance to removal (proportion of animals removed required to have an effect for more than one year), and the resilience of the population (how quickly the population recovered under various removal levels).

With no removal, the population was stable and population size ranged from 350 to 700 adult animals with 15 to 35% of the population being non-territorial. The reason for this stability was animals were forced out of packs as they matured and non-territorial animals had a higher mortality rate then animals in packs. The population exhibited source-sink dynamics. Average pack size in this simulation was about four but varied from one to eight.

To determine the effect of pulse removal on the population, we let the population run for five years and then randomly removed 10 to 90% of the adult population in one year and then examined the response of the population. All populations recovered within one year when less than 60% of the population was removed (Figure 1). Basically, the population was reduced until new offspring were produced. The number of transients decreased as animals moved into packs and fewer animals moved out of packs. Populations subject to removal had younger age structures. When more than 60% of the population was removed, the population required more than one year to return to the population size prior to removal.

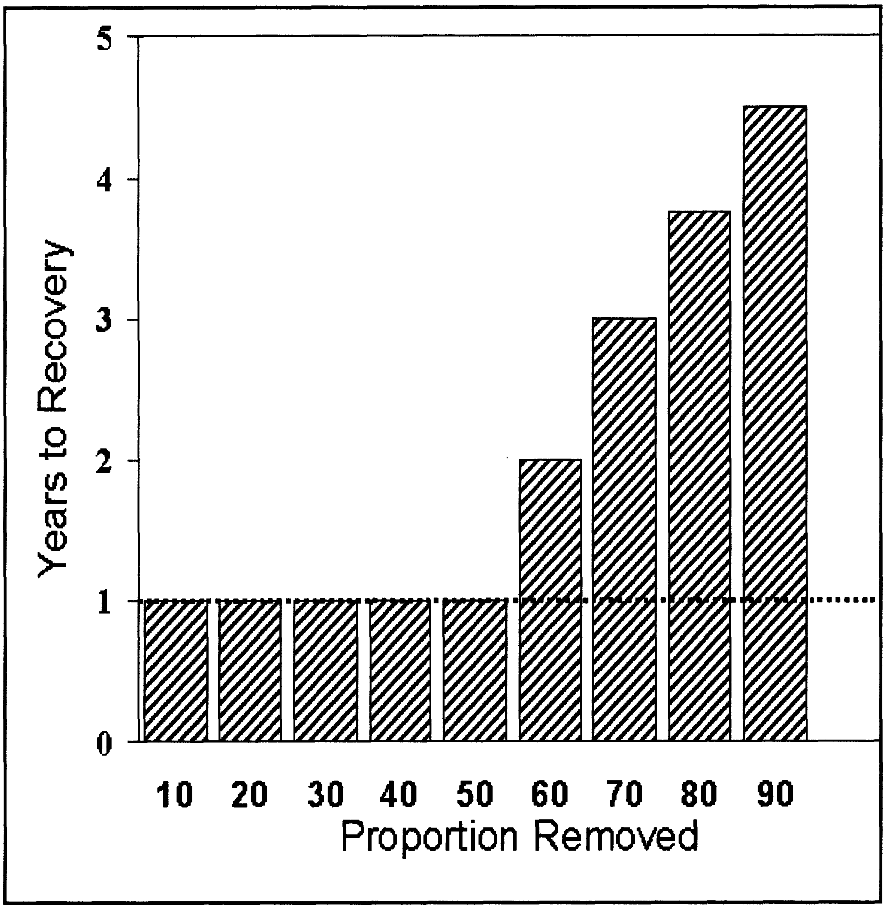

Figure 1. Number of years required for a population to return to pre-removal size after a certain proportion of animals are randomly removed from the population in one year. The horizontal line indicates the threshold where annual reproduction compensates for animals removed.

This removal proportion is lower than reported in previous computer models (Connolly and Longhurst 1975). The population recovered within five years, however, even with 90% removal in one year. When a large (>70%) proportion of the population was removed, the number of non-territorial animals decreased. In natural populations less time would be required to recover than what was depicted in the model because in the model, territories remained even at low densities, animals were not allowed to move out of their territories to mate, and animals were not allowed to move in from surrounding areas. Furthermore, we did not reduce natural mortality rates at low densities.

To determine the effect of sustained or press removal on the population, we let the population run for five years and then randomly removed 10 to 90% of the adult population each year and examined the response of the population. When removal was less than 60% of the population, population size was the same as an unexploited population, and it did not decline, even after 50 years of simulation. The population structure, however, differed from an unexploited population. The population with press removal at 50% had fewer transient animals (10 to 25%), a younger age structure, and higher reproduction than an unexploited population. High removal rates (>70% per year) resulted in an initial loss of non-territorial animals and after seven years the entire population was removed. In natural populations, a population decline could take several more years because territories remained in the model and animals did not move to mate, natural mortality rates were not reduced at low densities, and animals did not move in from surrounding areas. Removing more than 70% of the population annually would become logistically difficult at low densities. Territoriality would likely dissolve at low densities, animals would move to mate, natural mortality may be reduced, animals would immigrate into the population, and the high removal rates could not be achieved.

Implications for management

These simulations suggest that coyotes and other canid populations are very resistant and resilient to change. A population decrease was not observed until more than 60% of the population was removed annually. The populations are buffered against change by the high reproductive capacity and the non-breeding animals in the population, subordinates and non-territorial animals. Non-breeders would replace breeding individuals lost from the population so the reproductive capacity of the population is not reduced. Coyote and other canid populations are resilient because they have a high reproductive capacity. These conclusions may provide insight into the potential effects of disease on Ethiopian wolves (*Canis simensis*) or the potential management of timber wolves (*Canis lupus*). These species would also have similar population characteristics as was displayed in this model.

In the future, the model analysis will be expanded to investigate various types of removal, e.g. selective versus random removal. The management model allows the user to remove animals of a specific status, litters of offspring, during a particular month, and/or animals in a specific area. In addition, we can also investigate the effects of reproductive control or the effects of disease on populations. To determine the effects of unequal resource distribution, we can also vary resources among packs and over time as part of the foraging and predation models. Other components we contemplate adding are competing carnivores, predator-prey interaction, as well as, management cost-benefit programs to the model.

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Habitat issues

Mapping Occupied Habitat for Forest Carnivores in the American West and Estimating their Conservation Status

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Abstract
*Conservation concerns are escalating due to the small numbers, reduced range, and increasingly fragmented distribution of wide-ranging forest carnivores in the American West—specifically the lynx (*Lynx canadensis*), wolverine (*Gulo gulo*), and fisher (*Martes pennanti*). Observation data from the U.S. Forest Service and state natural heritage programs were compiled and mapped. Occupied habitats and population centers for the three species were identified and population sizes estimated based upon recorded densities and distribution. The results indicate small, isolated populations well below what may be necessary for long-term viability.*

Introduction
Some of the smaller forest carnivores of the American West—the lynx (*Lynx canadensis*), wolverine (*Gulo gulo*), and fisher (*Martes pennanti*)—have not yet grabbed headlines like bears (*Ursus* spp.) and wolves (*Canis lupus*). Yet mounting concerns about their viability indicate that large carnivores are not the only species in jeopardy because of past and ongoing habitat loss and fragmentation, and excessive human-caused mortality. While these lesser known species were never persecuted like the wolf, grizzly bear (*Ursus arctos*), and other species that pose a risk to livestock, their conservation status is now suffering due to neglect.

Critical information about the habitat requirements and population sizes of these animals is lacking, leaving conservationists ill-informed at where and how to initiate conservation strategies. Existing range maps are too coarse to prioritize specific areas for protection, such as individual mountain ranges or watersheds. There are few published population estimates, and they are limited to specific study areas and fail to address the conservation status of each species throughout its range.

We can no longer take for granted the survival of wide-ranging, low-density forest carnivores. As we have done for the grizzly bear and the wolf, we should make conscious decisions about where in the western U.S. we will maintain and restore forest carnivores, and protect them and their habitat in these areas accordingly. The capacity of these areas to support forest carnivores should be assessed to ensure that sufficient habitat is protected to provide for their long-term survival and recovery. Given the forest carnivores' large ranges and low densities and the fragmented nature of suitable habitat that remains, a conservation strategy will likely require restoration and maintenance of a connected network of population centers across the western U.S. and Canada (e.g., McKelvey et al. 2000a). The objective of this paper is to identify occupied habitats and population centers for the lynx, wolverine, and fisher based on available presence/absence data, and to estimate population sizes within these areas based upon recorded densities, as a first step toward devising a long-term conservation strategy for these species.

Mapping occupied habitat
To identify occupied habitats and population centers for the lynx, wolverine, and fisher, observation data from the U.S. Forest Service and the natural heritage programs of eight western states were gathered (Maj and Garton 1994; McKelvey et al. 2000b). These data were not collected through standardized survey techniques but include observations made by workers in the field, trapping records, and museum specimen records. State natural heritage programs maintain this data and attempt to filter out unreliable observations.

GIS software was used to highlight parcels of public lands where the observations were located. Private land was excluded from this analysis, because of the difficulty delineating borders around point locations on private lands. Administrative boundaries were convenient for this purpose on public lands, and though they have no biological basis they serve the purpose of delineating the large landscape features relevant to this scale of analysis. This methodology unfortunately excludes important areas of carnivore habitat on private lands, but the vast majority (>80%) of the carnivore observations occurred on public lands.

These areas were sorted by date to provide an approximation of current versus historic range. Observations made after 1990 were used to delineate current range while observations prior to 1990 denote historic range for the purpose of this analysis. Overlapping data for all three species revealed occupied habitat for all three species, and for various combinations of the three (Figure 1).

Estimating current population status

Population centers for each species were identified using observation data plus additional information found in scientific literature. Population numbers were estimated by multiplying the area occupied by a species (as determined by current observation data) by density estimates obtained from field studies in those or similar areas. For example, an analysis of wolverine observations in Idaho indicates three areas where sightings are concentrated: the Selkirk Mountains of northern Idaho, the Lochsa and Kelly Creek drainages in north-central Idaho, and the Sawtooth and Smoky Mountains in south-central Idaho (Groves 1988). Delineation of the specific clusters was subjective, but they account for 21%, 18%, and 22% of the 89 "probable" wolverine reports respectively. Each of these areas encompasses approximately 20,000 square kilometers of public lands, which comprises 90 to 100% of these areas. Copeland (1996) estimated one wolverine per 90 to 248 km^2 within his study area in the Sawtooth Mountains, based upon his analysis of 1,050 relocations of 19 wolverines by ground and aerial telemetry. A conservative estimate of one wolverine per 200 km^2 across each of these areas indicates three subpopulations of 100 wolverines each in Idaho (Figure 2). Delineation of potential subpopulations for each of the forest carnivores and estimating their numbers provides a starting point to assess the conservation status of these species relative to other imperiled species like grizzly bears and wolves.

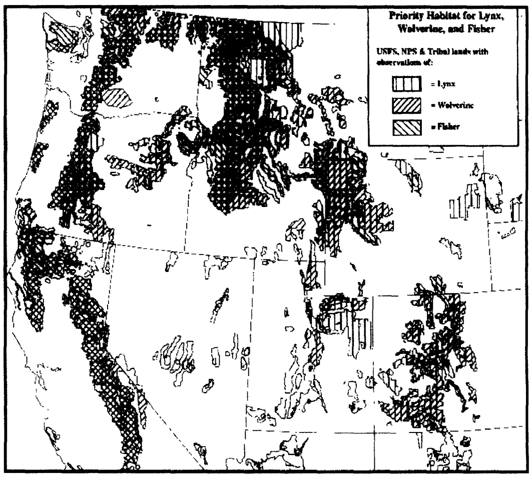

Figure 1. Occupied habitat for lynx, wolverine, and fisher

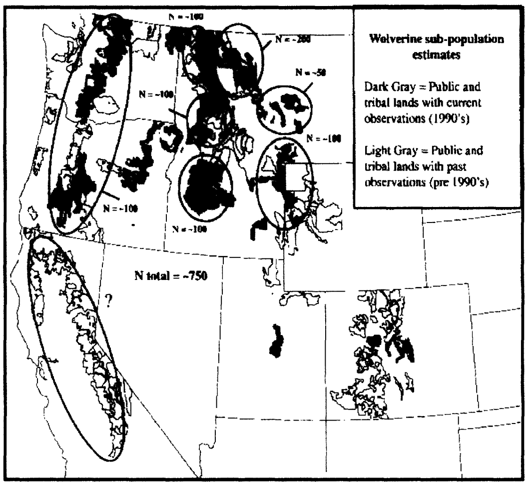

Figure 2. Wolverine sub-population estimates

Results

Occupied habitat for all three forest carnivores occurs throughout northern and central Idaho, northwestern Montana, portions of the Greater Yellowstone Ecosystem, Washington (northeastern corner, North Cascades), and Oregon (South Cascades). Lynx presence diminishes to the south, notwithstanding recent progress restoring lynx to the southern Rockies. Fishers are rarely observed south or east of the Bitterroot Range along the Idaho/Montana border, although populations persist in Cascades of southwestern Oregon, and in both the northern and southern Sierra Nevada Range. Wolverine range appears to have decreased over recent decades as follows. Known to historically occupy the Cascades of the Northwest and south into the southern Sierra Nevada, current sightings throughout this western extent of their range are rare. In Washington and Oregon, there have been only seven verifiable sightings since 1986, and despite an extensive helicopter survey in the Cascades, current wolverine presence is confirmed only in the North Cascades of Washington (K. Aubry personal communication). There has not been a verified wolverine observation in the Sierra Range since before 1990 (Zielinski, personal communication).

Population estimates indicate current numbers of all three species may be well below what is needed to ensure their long-term viability, if you consider that a minimum effective population size of 500 individuals may be necessary (Soulé

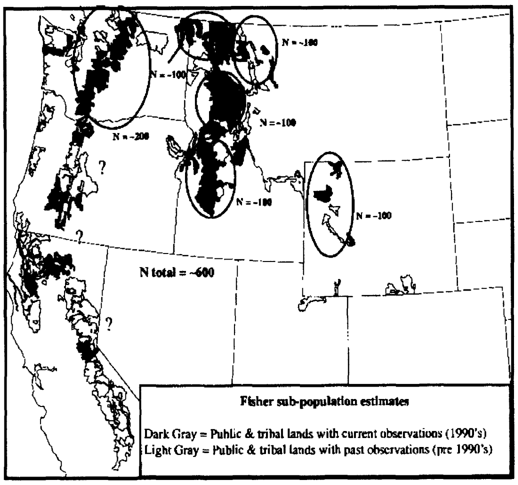

Figure 3. Fisher sub-population estimates

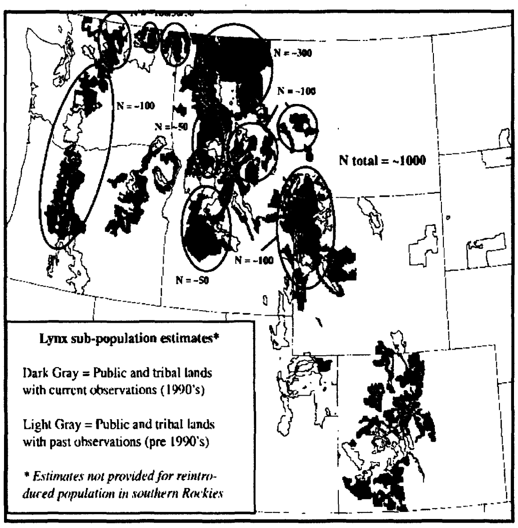

Figure 4. Lynx sub-population estimates

1987), and that just a fraction of the individuals within these carnivore populations are actively breeding and thus count toward the effective population size (Ruggiero et al. 1994). The fisher appears to be the most imperiled of the three in the West, with an estimated total population of perhaps 600 animals, including populations that may be functionally isolated in the southern Sierra, northern Sierra, southwestern Oregon, and portions of Greater Yellowstone (Figure 3). The wolverine is more widespread than the fisher, but its low densities coupled with recent declines in the Sierra and Northwest are cause for concern. Population estimates for both the fisher and the wolverine are less than estimated numbers of lynx (Figure 4) and grizzly bear across the western U.S., both of which are listed as threatened species under the federal Endangered Species Act.

Discussion

This analysis provides a coarse but useful first step toward identifying occupied habitat for the lynx, wolverine, and fisher, toward assessing their conservation status across the American West and prioritizing areas for their protection. Observation data are inherently biased by observer effort and reliability, and by imperfect reporting and organization of observations. These data should be refined with more careful screening and supplemented with new information regarding the conservation status and distribution of these species. Additional data from Canada are needed to develop a comprehensive, range-wide conservation assessment and strategy for the forest carnivores. Despite these limitations, this analysis provides an initial step toward the design and implementation of a forest carnivore conservation strategy that encompasses occupied habitat within the western U.S. for one or more of these species.

Acknowledgements

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Road Density as a Factor in Habitat Selection by Wolves and Other Carnivores in the Great Lakes Region

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Abstract

*Although wolves (*Canis lupus*) and many other carnivores are habitat generalists, certain landscape features can be used to predict suitable habitat. Thiel examined the concept of road density as an important factor in the persistence of wolf populations in Wisconsin prior to the 1960s and found a relationship with the disappearance of breeding wolf populations when average road density exceeded 0.58 km/km^2. Mladenoff and colleagues examined road density in the early 1990s as a factor in predicting favorable habitat of wolves colonizing Wisconsin between 1980 and 1992, and found that areas with road densities less than 0.45 km/km^2 had greater than a 50% probability of being colonized by wolf packs. Mladenoff and colleagues updated this work in the late 1990s by examining 23 packs colonizing Wisconsin between 1993 and 1997; 78% continued to occupy areas with road densities below 0.45 km/km^2. In a recent examination of radio-collared wolves in Wisconsin, a total of 60% of human-induced mortality occurred at road densities above 0.63 km/km^2. Although road density may become less of a factor as human tolerance changes, and wolf populations increase, it continues to be an important factor in habitat selection by wolves and probably other carnivores.*

Introduction

Gray wolves (*Canis lupus*) are generalists in their use of habitat, and historically have been found in most regions across temperate, boreal, and arctic regions of North America (Mech 1995). Despite this generalist nature of habitat use, landscape features, especially those relating to human impacts, can be used to predict suitable wolf habitat (Corsi et al. 1999; Massolo and Meriggi 1998; Mladenoff et al. 1995). Road density has frequently been used as a landscape feature to predict suitable wolf habitat (Corsi et al. 1999; Frair 1999; Fuller et al. 1992; Jensen et al. 1986; Mech et al. 1988; Mladenoff et al. 1995; Thiel 1985).

Early development of concept of road density

As a graduate student under Aldo Leopold, Thompson (1952) studied wolf food habits in northern Wisconsin in the late 1940s, about 10 years before wolves disappeared from the state. Thompson (1952) warned that development and opening of roads across the logged forests of northern Wisconsin could cause wolves to become extirpated from the state. As predicted, wolves were extirpated from Wisconsin by 1960 (Thiel 1985; Wydeven et al. 1995).

Thiel (1985) examined the disappearance of breeding populations of wolves in Wisconsin from 1926 to 1960. Using State Highway Commission Reports, he determined that breeding wolves disappeared from 10 Wisconsin counties after road densities in these counties exceeded 0.48 to 0.68 km/km^2 ($\bar{X}$=0.58 km/km^2). The value of 0.6 km/km^2 has since been used frequently as the threshold level at which wolf populations can be maintained. This level was found to correspond well with areas of occupied wolf range in Minnesota (Fuller et al. 1992; Mech et al. 1988), Michigan, and Ontario (Buss and Almeida 1998; Jensen et al. 1986).

GIS analysis of road density

Through elimination of bounties and protection by the 1973 Endangered Species Act, wolves were provided protection that allowed re-colonization of Wisconsin in the 1970s (Wydeven et al. 1995). Mladenoff et al. (1995) used a geographic information system (GIS) to assess landscape features that contributed to re-colonization of 14 Wisconsin wolf packs from 1980 to 1992. Known pack territories of radio-collared wolves were compared to 14 random non-pack areas scattered across northern Wisconsin. Areas occupied by wolf packs (80% isopleth of harmonic mean) had average road densities of 0.23 km/km^2 (Table 1). Road density was based on paved roads, and improved dirt and gravel roads that appeared as solid lines on U.S. Geological Survey (USGS) 1:100,000 quadrangle maps (Mladenoff et al. 1995). Other important features of wolf pack areas included lack of urban and agricultural areas, extensive forest (X=93%), high percentage wetland (X=29%), mostly public lands and industrial forest land (X=80%), and low human population density. Human density compared closely to road density.

Table 1. Average landscape variables for 14 wolf packs, random non-pack areas (n=14) and overall Wisconsin study area (modified from Mladenoff et al. 1995).

Variable	Pack Territories	Non-Pack Areas	Study Area
Land Cover			
Urban Areas	0%	0.2%	1.0%
Agriculture	2%	28%	21%
Total Forest	93%	63%	73%
Upland Forest	68%	51%	59%
Lowland Forest	25%	12%	14%
Marsh or Bog	4%	2%	2%
Water	1%	6%	4%
Land Ownership			
Public Lands	70%	24%	27%
Private Industrial	10%	1%	5%
Other Private Land	21%	75%	66%
Density			
Road Density	0.23 km/km^2	0.74 km/km^2	0.71 km/km^2
Human Density	1.52 km/km^2	5.16 km/km^2	7.43 km/km^2
Deer Density	8.58 km/km^2	8.38 km/km^2	8.22 km/km^2

Mladenoff et al. (1995) used road density as the main factor used in a logistical regression model that predicted areas of suitable wolf habitat (Table 2). Areas with < 0.45 km/km^2 were considered highly suitable wolf habitat and were estimated to have > 0.50 probability of being colonized by wolf packs. Minnesota had the most extensive area of highly suitable habitat, and packs expanded outside perceived suitable habitat in some areas (Berg and Benson 1998). Fuller et al. (1992) indicated that in 1989, 88% of wolf pack areas had road densities < 0.70 km/km^2. As predicted, most areas of highly suitable habitat were occupied by wolves in Michigan (James Hammill, personal communication).

Based on the logistical regression model, areas with > 0.60 km road/km^2 have less than a 10% chance of being occupied by wolf packs. Thus, the GIS analysis agrees with the threshold found by Thiel (1985).

Roads as a factor in wolf habitat in other studies

A habitat model developed in the Great Lakes region was used as a basis for assessing potential wolf habitat in the northeast U.S. (Harrison and Chapin 1998; Mladenoff and Sickley 1998). Mladenoff and Sickley (1998) estimated 53,500 km^2 of potential habitat in Maine and New Hampshire and 16,020 km^2 in New York. Harrison and Chapin (1998) estimated 48,787 km^2 of potential habitat in Maine and New Hampshire, and 14,618 km^2 in New York. Mladenoff and Sickley (1998) relied mainly on road density values, while Harrison and Chapin used a combination of road density and human population density.

Road density has also been found to be important in predicting

Table 2. Area of wolf habitat probability classes from a logistical regression model and corresponding road density for portions of three Great Lakes states (from Mladenoff et al. 1995).

Probability Class (P)	Road Density km/km2	Minnesota Area (km2)	Michigan Area (km2)	Wisconsin Area (km2)
>0.50	<0.45	50,200 (72%)	29,348 (70%)	14,864 (25%)
0.10 - 0.49	0.45 - 0.60	10,612 (15%)	7,160 (17%)	7,160 (21%)
<0.10	>0.60	9,328 (13%)	5,476 (13%)	32,100 (54%)

wolf habitat in Italy (Corsi et al. 1999; Massolo and Meriggi 1998). The estimated area of highly suitable habitat in Italy (14,200 km^2) (Corsi et al. 1999) was similar to the area estimated in Wisconsin (14,864 km^2) (Mladenoff et al. 1995).

Kohn et al. (2000) conducted a variety of studies examining the relationship of wolves to roads in northwest Wisconsin. Shelley and Anderson (1995) found road densities in northwest Wisconsin wolf territories to average 0.33 km/km^2 for eight pack areas. Although wolves selected areas of low road density, travel areas selected by packs were generally close to trails and forest roads (Gehring 1995). Unger (1999) found that wolves selected den sites in roadless or low road density areas; dens were generally located more than 1 km from improved roads. Frair (1999) found that wolves most frequently used areas away from roads, and average road density in wolf territories was 0.25 km/km^2; road density was found to be the best predictor of suitable wolf habitat.

Recent examinations of road densities in Wisconsin

Mladenoff et al. (1999) examined 23 additional wolf territories that colonized Wisconsin from 1993 to 1997. Five packs (22%) exceeded the 0.45 km/km^2 threshold of road density and one (4%) exceeded the 0.60 km/km^2 threshold. Thirteen packs were radio collared and provided more precise data on area of pack use; two (15%) exceeded the 0.45 km/km^2 threshold and one (7%) exceeded the 0.60 km/km^2 threshold. In general, the road density model continued to be a good predictor of suitable wolf habitat. The one territory that exceeded the 0.60 km/km^2 threshold was in a state wildlife area that had a higher road density (0.71 km/km^2), but greater access control may have nullified the effects of higher road density.

The wolf population in Wisconsin increased from 15 wolves in 1985 to 248 wolves in 2000, totaling 66 packs (Wydeven and Wiedenhoeft 2000). We examined whether packs occurred within areas of suitable habitat as illustrated by Mladenoff et al. (1995, 1999). Of 66 packs in Wisconsin, 53 (80%) were contained in areas mapped as highly suitable wolf habitat ($P > 0.50$), seven (11%) were contained within marginal wolf habitat ($0.50 > P > 0.10$), and six (9%) occurred in areas mapped as poor wolf habitat ($P < 0.10$). Thus, packs continued to occur mainly in areas of low road density. Even those packs occurring in areas that seemed unsuitable (road density > 0.60 km/km^2), were within 10 km of areas of highly suitable habitat.

We recently examined the relationship between wolf mortality and road density. Fifty radio-collared wolves died in Wisconsin between 1979 and 1999. The road densities for 47 wolf mortalities were collected (Figure 1). The average road density for natural mortalities was 0.65 km/km^2 and for human-induced mortalities was 0.78 km/km^2.

Highest natural mortality was at the road density range of 0.63 to 0.84 km/km^2, areas considered poor habitat. We initially had expected natural mortality to be highest in areas of most suitable habitat where wolves most frequently occurred, but higher rates at higher road density make sense. Wolves dying from natural mortality, died mainly from disease or intraspecific strife. Diseased animals lose their fear of humans, and often wander off by themselves into poor habitat. An adult female with severe mange crawled into a garage in 1993. Mortality from intraspecific strife usually occurs near the edge of a territory and often pack boundaries are near roadways. Thus even wolves dying from natural mortality are more likely to be killed closer to roads.

Human-induced mortality peaked at relatively high road densities (Figure 2). Most shootings and vehicle collisions occurred at road densities of 0.84 to 1.14 km/km^2. A total of 60% of human-induced mortality occurred at a road density greater than 0.63 km/km^2.

In general, wolves appear more likely to be killed at higher road densities. Although most wolves spend little time at these higher densities, they are at a much greater risk of being killed in these areas.

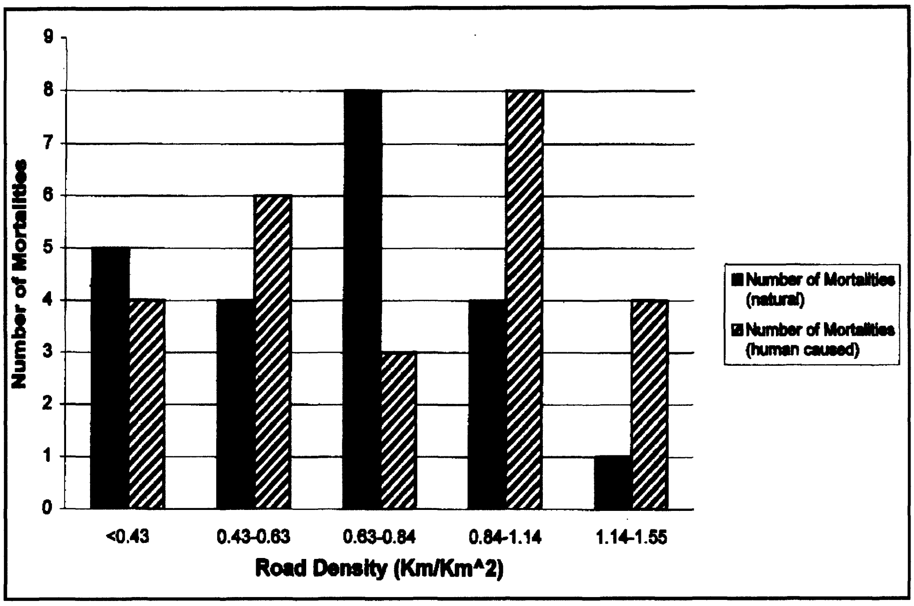

Figure 1. Cause of mortality of radio-collared wolves in Wisconsin 1979 to 1999 in relation to road density.

Discussion

As human attitudes toward wolves improve and wolf populations continue to increase, road densities may become less of a factor in wolf habitat selection (Mech 1995). In the Great Lakes region, road density provides a useful proxy for human disturbance and risk of mortality. In mountainous terrain where ungulate distribution is very patchy, road density may be a less useful index (Diane Boyd, personal communication), but in the generally homogeneous landscape of the northern Great Lakes region it continues to be useful. Whether road density serves as a good predictor of wolf habitat in northeastern U.S. remains to be seen.

Exceptions to the usefulness of road density will continue to occur. Mech (1989) cites an example of areas with road densities of 0.73 km/km^2 supporting wolf packs in Minnesota but having large wilderness reservoirs nearby. Merrill (2000) indicates that a military base in Minnesota with a road density of 1.42 km/km^2 has supported a wolf pack for over six years. Strict control on access, and human activity limited to day light hours, can nullify the effect of road density. In Wisconsin, packs occur in the Crex Meadow Wildlife Area and Necedah National Wildlife Refuge that are classified as low probability of pack occupation, but stricter access control by the management agency reduces the effects of high road densities. Wolves do not have an aversion to roads, and readily travel on roadways if traffic levels are low (Gehring 1995). Wolves learn to avoid roads with high traffic volumes, but readily use gated roads (Thurber et al. 1994). Although wolves seem to cross secondary roads, they vary in willingness to cross busy highways (Frair 1999). Some dispersers do extensive crossings of highways (Kohn et al. 2000; Mech et al 1995; Merrill and Mech 2000).

Road density as a habitat factor has applicability to other carnivores. Roads affect movements and harvest of black bear populations (*Ursus americanus*) (Brody and Pelton 1989). Bobcat (*Lynx rufus*) avoid certain types of roads and seem more attracted to areas with low traffic volume (Lovallo and Anderson 1996). American marten (*Martes americana*) may be impacted by trapping when access is high and avoid crossing large open areas, which could be impacted by road density (Chapin et al. 1998). Other carnivores that require large home ranges may also be affected by road density.

Management and research recommendation

Road density appears to be an important habitat factor for wolves and other carnivores; therefore public lands managed for these species should maintain suitable habitats with low densities of roads. Forested areas managed for wolves should maintain overall road densities of 0.6 km/km^2. Core wolf habitat should be managed at road densities of 0.45 km/km^2. If productive packs exist in areas at much higher road density, a reduction to lower densities would not be necessary, but attempts should be made to avoid increasing road density or changing traffic levels.

On public forest lands, new logging roads should be closed or obliterated after logging operations are completed. Where possible, temporary, winter-only roads should be used, because these roads cause least damage and revert back to vegetated areas. Areas within 100 meters of den sites should be kept undeveloped, and logging roads and trails should stay more than 100 meters from those sites.

The impact of road density is not well known for other carnivores; therefore research on habitat use should include assessments of road density as a habitat value. The impacts of various types of roads on carnivores should be studied and impacts from all-terrain vehicles (ATVs), snowmobiles, and other off-road vehicles should also be investigated. Roads are important habitat variables for carnivores that need to be more carefully researched.

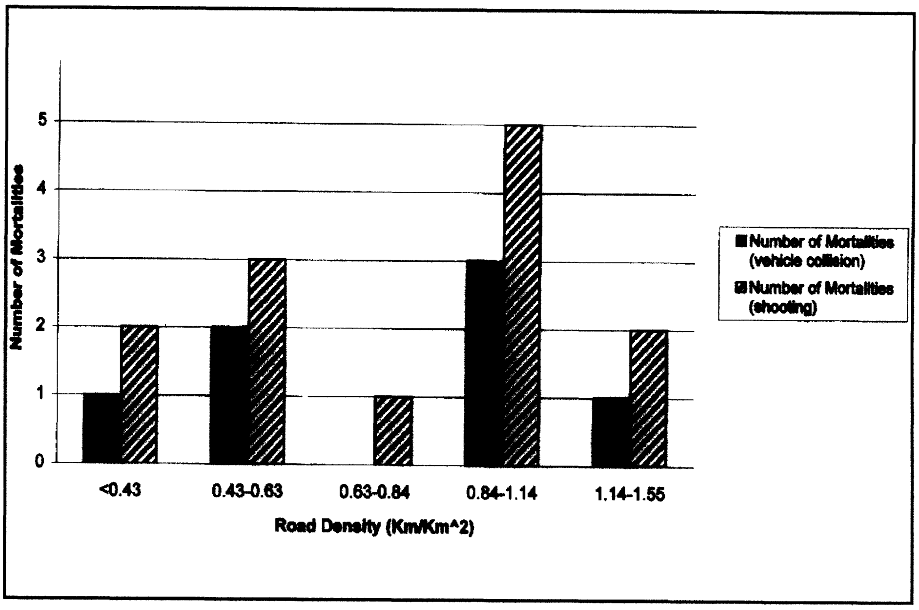

Figure 2. Human causes of mortality of radio-collared wolves in Wisconsin 1979 to 1999 in relation to road density.

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A Predator-Habitat Assessment for Felids in the Inland Atlantic Forest of Eastern Paraguay: A Preliminary Analysis

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Abstract

*Jaguar (*Panthera onca*), puma (*Puma concolor*), and possibly six species of small cats (ocelot,* Leopardus pardalis*; margay,* Leopardus wiedii*; oncilla,* Leopardus tigrinus*; Geoffroy's cat,* Oncifelis geoffroyi, *pampas cat,* Oncifelis colocolo, *jaguarundi,* Herpailurus yagouaroundi*) co-exist within Mbaracayú Forest Nature Reserve in eastern Paraguay. At the landscape level, this 64,000-hectare island of Inland Atlantic Forest, surrounded by agricultural land, is a mosaic of forest habitats and interspersed grasslands. Habitats on the reserve include low, medium, and high forest, as well as dry and wet grasslands. The adaptive nature of most predators led us to predict that felids would occur uniformly across habitats. Tests of independence between species and habitats, however, suggest distinct associations between felids and habitats. Explanations for habitat affinities include interference competition or simply following prey to their preferred habitats. Thus, while felids in the inland Atlantic forest may be habitat generalists across their entire range, they exhibit some habitat preferences within Mbaracayú reserve, possibly as an adaptation to interspecific competition and/or prey availability. Further research is needed to determine whether these patterns continue long-term or are an artifact of the timing of our current data collection efforts.*

Reserva Natural del Bosque Mbaracayú

The Mbaracayú reserve is the largest tract of undisturbed forest in eastern Paraguay covering approximately 64,000 hectares of the quarter million hectare upper Jejui watershed. Only the combined reserve areas in Brazil and Argentina around Yguazu Falls contains more of the Alto Parana formation of the Atlantic Rainforest. Located at approximately 55° west and 24° south, the reserve is mostly between 150 to 300m in elevation and is drained to the west by the Paraguay River. The area averages about 1800 mm of rainfall per year but is characterized by extreme unpredictability in monthly patterns from year to year. The typical dry season lasts from May to September (Sanchez 1973; FMB unpublished data). Seasons are associated with marked temperature fluctuations, with average daily high-low temperatures of 14 to 25 ° C in July and 22 to 34° C in January. The reserve contains about 90% of all species classified as rare or endangered within Paraguay (FMB 1992) and was chosen as the top priority conservation site in eastern Paraguay (Keel et al. 1993). The Mbaracayú reserve is located in the traditional homeland of Ache hunter-gatherers. The Ache have exceptional knowledge of the Paraguayan forest because they lived off wild resources until recently and most adults have spent most of their lives in the forest (Hill and Hurtado 1989). Further, the Ache are allowed exclusive rights to hunt and forage within the reserve.

Felid species within Mbaracayú Reserve

The diversity of mammalian carnivores (Mammalia, Carnivora) existing within the bounds of the reserve is high with published lists of species ranging between 13 to 17 (Hill et al. 1997; FMB 1997; MNHNP 1996). There is disagreement about the occurrence of some species, with the following listed: three or four canids [bush dog (*Speothos venaticus*), maned wolf (*Chrysocyon brachyurus*), crab-eating fox (*Cerdocyon thous*), pampas fox (*Pseudalopex gymnocercus*)], two procyonids [South American coati (*Nasua nasua*), crab-eating raccoon (*Procyon cancrivorus*)], four or five mustelids [tayra (*Eira barbara*), lesser grison (*Galictis cuja*) long-tailed otter (*Lontra longicaudus*), giant otter (*Pteronura brasiliensis*), hog-nosed skunk (*Conepatus chinga*)], and, five to eight felids [jaguar (*Panthera onca*), puma (*Puma concolor*), ocelot (*Leopardus pardalis*), margay (*L. wiedii*), oncilla (*L. tigrina*), jaguarundi (*Herpailurus yagouaroundi*), Geoffroy's cat (*Oncifelis geoffroyi*), pampas cat (*O. colocolo*)]. Many of these carnivore species, including all canids and felids, are listed by the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), either on Appendix I or Appendix II (Table 1). We chose to focus on felids as a first analysis of habitat associations by predators in Mbaracayú reserve.

The jaguar is found in many habitats throughout its range (southwestern United States through northern Argentina). This is the largest native cat in the western hemisphere with males weighing as much as 136 kg. Jaguars are a top predator, and, in Mbaracayú Reserve, evidence of predation by a jaguar on a puma has been documented (K. Hill, personal observation). Jaguars are thought to be dietary generalist and take advantage of any prey available. Pumas currently exist over a larger area (northern Canada through southern Chile and Argentina) than jaguars and function as top predators in much of their range. Pumas are also large cats (males weighing as much as 103 kg); they are considered deer specialists, although they feed on a diverse prey spectrum.

Several species of small cats potentially co-exist within Mbaracayú Reserve. The presence of the ocelot, oncilla, jaguarundi, and Geoffroy's cat at this site is confirmed by voucher specimens at the Natural History Museum of Paraguay (MNHNP 1996). Margays are considered residents (Hill et al. 1997) and pampas cat is rumored to exist in the cerrado portion on the western edge of the reserve (E. Esquivel, personal communication). This discrepancy of confirmed felid species, coupled with the limited number of local names for small cats in the area (Table 2), led us to consider all small cats as a group.

Major habitat types within Mbaracayú Reserve

Until recently, categorizing the forest mosaic of Mbaracayú was difficult. Sources for identifying individual plants and some plant communities have been available for reference for several years (Lopez et al. 1987) but no consistent description of key habitat types was available. The publication of "Plantas Comunes de Mbaracayú" provides a scientifically based habitat classification system for studies within Mbaracayú (Marín et al.

Table 1. CITES listings for mammalian carnivores in Paraguay (MAG-CITES 1999).

Carnivore Species	Scientific Name	CITES Listing
Canidae		
bush dog	*Speothos venaticus*	Appendix I
maned wolf	*Chrysocyon brachyurus*	Appendix II
pampas fox	*Pseudalopex gymnocercus*	Appendix II
Mustelidae		
long-tailed otter	*Lontra longicaudus*	Appendix I
giant otter	*Pteronura brasiliensis*	Appendix I
Felidae		
jaguar	*Panthera onca*	Appendix I
puma	*Puma concolor*	Appendix II
ocelot	*Leopardus pardalis*	Appendix I
margay	*Leopardus wiedii*	Appendix I
oncilla	*Leopardus tigrina*	Appendix I
jaguarundi	*Herpailurus vagouaroundi*	Appendix II
Geoffroy's cat	*Oncifelis geoffroyi*	Appendix I
pampas cat	*Oncifelis colocolo*	Appendix II

Table 2. Indigenous and local common names for possible small cat species in Reserva Natural del Bosque Mbaracayú. No Ache names exist to distinguish Margay, Geoffroy's cat, or Pampas cat. Jaguarete'i, and Tiríka are each used for multiple cat species (Villalba and Yanosky 2000, MAG-CITES 1999, FMB 1997).

English Common Name	Ache Name(s)	Paraguayan Common Name(s)
ocelot	Kajá	Jaguarete'i, Jaguatiríka
margay	--	Jaguarete'i, Tiríka
oncilla	Kajaminí	Tiríka
Geoffroy's cat	--	Tiríka, Tiríka'i
pampas cat	--	Osio
jaguarundi	Mberembo, Mbekrymbá	Jaguarundi, Eira

1998). Habitat descriptions provided by the Ache were synthesized into the categories outlined in Marín et al. (1998). In our treatment of habitats within Mbaracayú we recognize three (low, medium, and high) forest types, grasslands, and cerrado.

Low forest

Located on dry land above stream drainages, low forests are characterized by trees <15 m high with a typical diameter at breast height (dbh) <10 cm and none with a dbh >25 cm. The understory is dominated by bromeliads (Bromeliaceae) and ferns (Blechnaceae, Cyatheaceae). Major fruiting plants within low forest are *Faramea porophylla* and *Coussarea contracta* within the Rubiaceae. Low forest constitutes approximately 30% of the area of the Reserve.

Medium forest

Medium forest is characterized by variability. Some portions of medium forest are dominated by bamboo, either the large *Guadua angustifolia* bamboo or the small diameter *Merostachys clausseni* bamboo. There are few trees associated with the large bamboo stands and the tree height associated with small bamboo understory ranges from 15 to 25 m. *Copaifera langsdorfii* or *Hexachlamys edulis* can be the dominant woody tree species. Other portions of medium forest are dominated by lianas. These areas are typically far from a permanent water source and ground cover is sparse. Although riparian forests are distinguished by the Ache as well as in several treatments of the reserve, here it is included in medium forest for the purpose of analyses. These areas are in the downslope near streams and rivers and are generally richer in fruit species. As with low forest, medium forest covers approximately 30% of the Reserve area.

High forest

High forest, the most common plant community within Mbaracayú reserve, is described in detail in Hill et al. (1996). The canopy of this forest is relatively tall (typically 12 to 20 m with some trees exceeding 30 m) and trees commonly have epiphytic orchids (Orchidaceae) and philodendrons (Araceae). It is found at intermediate elevations above the water table on gently sloping ground. The basal area of tree species >10cm dbh is 39m^2 per ha in high forest (Keel 1987). Ground cover in high forest is generally sparse including ferns, heliconias (*Heliconia psiattacorum*; Musaceae), and bromeliads. Approximately 30% of the Reserve area is comprised of high forest.

Grasslands

Grasslands are areas of open graminoid (i.e., *Loudetia flammida*, *Axonopus suffultus*, *Andropogon bicornis*) meadow with patches of cerrado-like vegetation (i.e., palms, *Butia paraguayensis*; Palmae). They are usually wet during part of the year and are characterized by poor drainage and a layer of black organic matter overlaying white sandy soils. Several leguminous species (*Mimosa spp.*, *Desmodium spp.*, and *Chamaecrista spp.*) also characterize this habitat. Wet grasslands are dominated by several grasses (*Setaria paucifolia* and *Paspalum plicatum*; Gramineae and *Rhynchospora glabosa*; Cyperaceae), some of which can reach two meters in height. These areas are wet throughout the year with standing water up to half meter deep during some parts of the year.

Cerrado

A large area of true cerrado habitat (~7,000 hectares) now exists within the reserve. Cerrado is a dry grassland/forest interface dominated by mixed-height grasses (*Annona coriacea*, *Dugetia furfuracea*; Annonaceae, and *Eragrostis polytricha*; Gramineae) and palms (*Butia paraguayensis*, Palmae). Because this area was not a part of the reserve when data collection was initiated, however, it has been excluded from surveys since its purchase.

Research question

Given the information at hand, we sought to determine if there are patterns of association between specific felids and major habitat types. We hypothesize that: felids in Reserva Natural del Bosque Mbaracayú exhibit no detectable habitat associations. Our rationale for this are (1) predators are generally highly adaptive, and (2) the habitats in Mbaracayú exist as a mosaic with no distinct patches available for exclusive residence.

Data collection and analysis

A stratified random sample of walking transects through the Mbaracayú reserve was employed as the field method of game censusing. Data for this analysis come from four census periods: June 1994 to April 1995, July 1995 to June 1996, July 1996 to May 1997, and July 1997 to May 1998. Five Ache assistants, walking in parallel, and a data recorder, walking 5 m directly behind the middle researcher, censused the same 91 transects each year of the study. One individual walked directly on the transect line while two assistants were located on each side, separated by 25 and 50 m. Each transect began at a specified GPS location and proceeded along a specified compass bearing throughout the day for approximately 5 km. Transects passed through all habitat types and were only suspended when water exceeded half meter in depth. Transects were continued on the far side of water bodies (swamp, streams, river, etc.) that had to be crossed. A GPS location and habitat description were noted for all types of animal encounters (for all animals at least 0.5 kg in size). The type of sighting was noted: (1) animal was seen, heard, or found in a burrow, or (2) identified through fresh sign or feces. All locations during the study were determined using a Trimble Pathfinder Pro Global Positioning System (GPS) receiver. Data were then categorized according to species (jaguar, puma, small cat) and habitat type prior to statistical analysis. In order to detect deviations from random habitat associations, a Chi-square test for independence between species occurrence and habitat was performed using SAS (SAS Institute Inc., Cary, NC 27513). Data were pooled across survey years to ensure a robust analysis.

Results and discussion

The Chi-square test resulted in a significant deviation from random ($\chi^2 = 22.29$, $P = 0.0012$) indicating that some species were non-randomly associated with habitats. The relationship between species occurrence within each habitat is displayed in Figure 1. Jaguars occur within high forest habitat greater than expected and in low forest and grassland habitats less than expected. This apparent preference for high forest by jaguars may be related to greater availability of prey species; an issue that requires further investigation. Jaguars appear to be the exclusive top predator in Mbaracayú reserve; their diet overlaps and includes all other cat species.

Pumas occur within all habitat types as expected. The lack of a pattern by pumas may reflect an overall high abundance of brocket deer, *Mazama spp.*, within Mbaracayú reserve (Hill unpublished data).

Small spotted cats occur within low forest and grassland habitats greater than expected and in high forest habitat less than expected. The apparent preference for low forest and grasslands by small spotted cats may reflect an avoidance of jaguars, and other large predators, or a response to more abundant prey. Data for small mammal and small bird population densities are not available.

Thus, while felids in the inland Atlantic forest may be habitat generalists across their entire range, they exhibit some habitat preferences within Mbaracayú reserve,

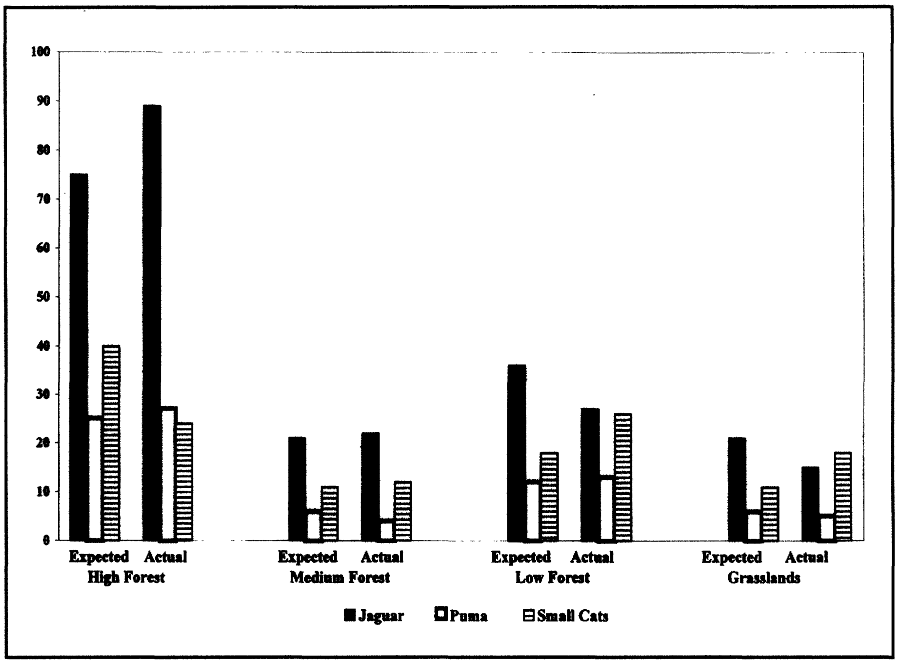

Figure 1. Expected and actual counts of jaguars, pumas, and small cats within four habitat types in Mbaracayú reserve.

possibly as an adaptation to prey availability and interspecific competition. Research is needed to test our hypothesis that small cats appear to be associated with distinct habitats, in response to potential predation by jaguars and pumas.

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Raptor Conservation

Status of the California Condor and Mortality Factors Affecting Recovery

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Abstract
*The California Condor Recovery Program (Program) is moving forward after the release of 116 California condors (*Gymnogyps californianus*) to southern and central California and northern Arizona. As of February 1, 2001 a total of 161 condors were alive of which 46 were in the wild: 24 in California and 22 in Arizona. Due to a high number of fatalities thus far (43), the Program remains challenging. Discovering the relative importance of present-day mortality factors and controlling especially problematic ones is critical to recovery efforts. The effects of lead toxicity as a result of contamination from spent rifle and shot ammunition, and possibly other sources, is the most significant threat presently. As a result, field release programs must focus a great deal of effort on ways to counter mortality until long-term solutions to this problem can be realized. This paper discusses from the perspective of Ventana Wilderness Society, the current status of the California condor, mortality factors that affect recovery, and management strategies to reduce fatalities.*

Current status of recovery program

In 1987, all remaining wild condors were brought into captivity to prevent their extinction and rebuild the population. By 1992, captive breeding at the Zoological Society of San Diego and Los Angeles Zoo increased the population to 52 birds and U.S. Fish and Wildlife Service (FWS) began releases of young condors to southern California. The Peregrine Fund began captive breeding in 1993 and releases to Arizona in 1996. Ventana Wilderness Society began releases in central California in 1997 to aid the recovery of the species. Captive breeding has brought the total population size to 161 birds as of February 1, 2001 (Jurek personal communication).

The California Condor Recovery Program Plan (1996) calls for two geographically separate wild, self-sustaining populations each having 150 individuals, 15 breeding pairs and a positive rate of growth as well as a third population with 150 individuals in captivity in order to qualify for reclassification from endangered to threatened. As of February 1, 2001, 116 different condors have been released at three different geographic locations and despite heavy losses the wild population maintains a fairly evenly distributed age structure (Table 1). A total of 27 condors have been removed to captivity for possible re-release and another 43 birds have died or were presumed dead. A total of 46 condors remain in the wild—24 in California and 22 in Arizona. The oldest members of the reintroduced population are now entering maturity and therefore successful breeding may occur in the near future. In order to attain a self-sustaining population, controllable mortality factors must be identified and reduced.

Mortality factors affecting recovery

California condors are a long-lived k-selected species with low reproductive rates (Temple and Wallace 1989). Wild condor populations cannot sustain themselves with annual mortality rates exceeding five to 7.5% for adults and 13 to 15% for juveniles (Verner 1978; Meretsky et al. 2000). Up to the early twentieth century, the condor population plummeted due to losses primarily caused by shooting, poisoning, and museum collecting (FWS 1996). Throughout the remaining 1900s the condor contin-

Table 1. California Condor releases, losses, and age structure in wild (as of February 1, 2001). (a) "Total" includes one bird (b) relocated to Arizona and therefore was released twice.

	Southern California	Central California	Arizona	Total
No. released	48	22	47	117(a)
No. removed to captivity or possible re-release	12	7	8	27
No. relocated	0	1(b)	0	1
No. died or presumed dead	26	0	17	43
No. 1-year-olds in wild	0	0	5	5
No. 2-year-olds in wild	1	5	1	7
No. 3-year-olds in wild	0	3	4	7
No. 4-year-olds in wild	2	6	2	10
No. 5-year-olds in wild	0	0	4	4
No. 6-year-olds in wild	7	0	6	13
Total in wild	**10**	**14**	**22**	**46**

ued to decline though the reasons were not entirely clear.

Known causes of death in the reintroduced population include: power line collisions/electrocutions, coyote (*Canis latrans*) and golden eagle (*Aquila chrysaetos*) predation, lead poisoning, accidental drowning, shooting, ethylene glycol poisoning, aspiration, cancer, and malnutrition (Jurek personal communication). Meretsky et al. 2000 determined that recent annual mortality rates of the reintroduced population are in excess of what is critical to maintain a self-sustaining population. In the following section, we will discuss current mortality factors to shed light on their impact on recovery efforts.

Electrocution/collision

Power line collisions and electrocutions represented a significant threat to the reintroduced population of condors during the first two years of release efforts. A total of 4 birds were lost to this hazard and the remaining individuals regularly perched on power poles, which led to their return to captivity (Snyder and Snyder 2000). Beginning with the 1995 release cohort, a negative conditioning program was initiated to train condors from perching on power poles (Wallace personal communication). The results of this training appear to be promising. Prior to aversion training, 31% (four of 13) of the condors released died to collision and or electrocution with power lines, while only 2% (two of 103) died of the same cause after the training was required for all releases. As a result, this threat was greatly reduced. The long-term effectiveness of this training, however, is unknown.

Predation

Coyotes and golden eagles combined killed eight young or inexperienced reintroduced condors. Although ravens and golden eagles are known threats to condor eggs and nestlings respectively (Snyder 1986), adult mortality from natural causes is virtually unknown (FWS 1996). The high rate of deaths to predation is likely a result of the reintroduced birds not having parental guidance or protection in the wild.

Shooting

Shooting greatly affected the original wild population since scores of condor deaths were attributed to this factor both done indiscriminately and for the purposes of museum collecting (Wilbur 1978). In the reintroduced population only two condors were shot and killed. While this threat may always loom over the recovery of this species, it does not, by itself, appear to be a significant problem at present.

Lead poisoning

Lead exposure and acute poisoning was first detected in the original wild population of California condors during the 1980s. Between 1984 and 1986 three condor deaths were attributed to lead poisoning and one bird showed a positive radiograph for a bullet fragment in its digestive tract (Janssen et al. 1986). Other evidence of the lead threat within the range of the species includes studies on condors (Wiemeyer et al. 1988), turkey vultures (*Cathartes aura*) and ravens (*Corvus corax*) (Wiemeyer et al. 1986b), and golden eagles (Pattee et al. 1990).

Condors encounter lead bullet/pellet fragments in their food supply, although the exact pathways are not well understood. Currently, lead rifle ammunition, in unrecovered carcasses and/or gut piles fed upon by the condor, may be the primary source of contamination. A single ingested lead fragment can be lethal (Risebrough personal communication). Four confirmed lead poisoning fatalities were documented among all reintroduced birds. To date, many of the reintroduced condors over the age of one year experienced elevated levels of lead in their blood at one time or another. Since 1997, 14 condors were successfully treated for acute lead toxicity using a technique called chelation therapy.

Without this intensive intervention effort, lead poisoning would easily account for the greatest number of deaths. So important is the lead threat resulting from spent ammunition in the condors' food supply that the California Condor Recovery Team is now recommending that FWS reduce lead contamination within condor range (Recovery Team Meeting February 14, 2001).

Other causes

The original wild population experienced some fatalities from drowning, leg-hold traps, and in one case a collision with a vertical pipe used as a surveyor marker (Koford 1953). No evidence exists to support any of these "other causes" playing a consistent role in the decline of the original wild population (Wilbur 1978). Accidental drowning (2), ethylene glycol poisoning (1), cancer (1), aspiration (1), and malnutrition (1) are each recorded causes of death in the reintroduced population. Although these causes combined may affect current recovery to some extent, individually they do not appear to be significant problems.

Unknown causes and disappearances

Out of 116 different condors released since the onset of the reintroduction program, a total of seven deaths of unknown causes and another 10 disappearances (presumed dead) occurred. These results undermine the ability of the Program to combat limiting factors.

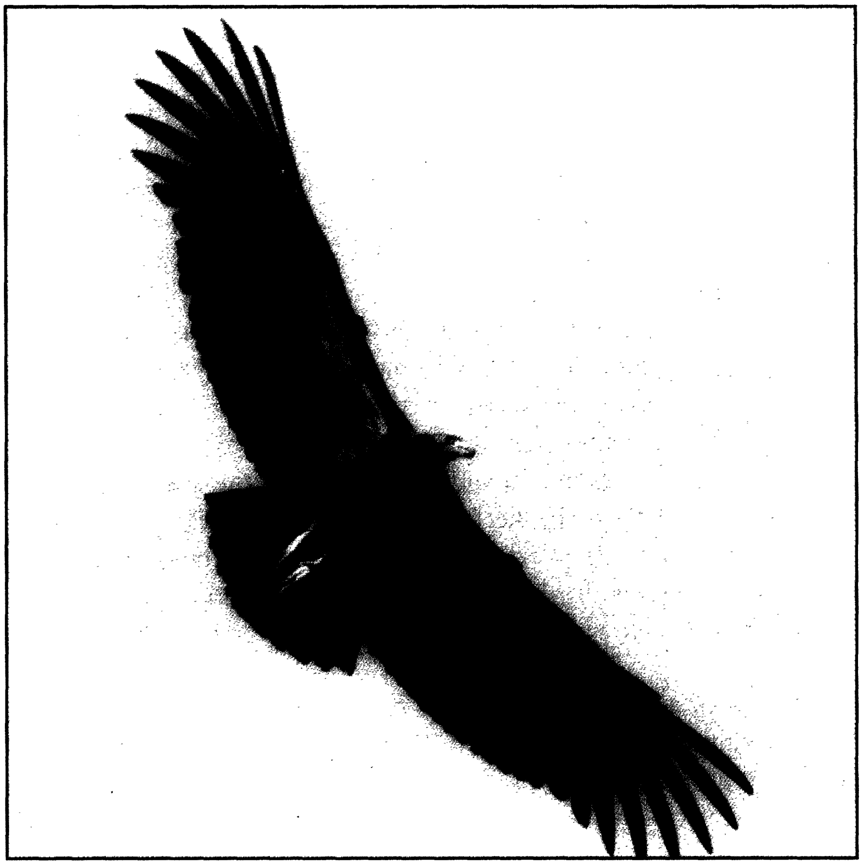
Management strategies

The authors support strategies to (1) reduce the threat of lead to condors through public education, promotion of viable non-lead ammunition alternatives and enhanced food subsidy efforts, (2) to improve monitoring efforts utilizing aerial and satellite tracking, and (3) to operate facilities in the field for post-release management. In the long run, measures taken to reduce the availability of lead in the condors' food supply represents the best chance for eventual full recovery.

Ventana Wilderness Society supports public awareness efforts to openly discuss the problems of lead poisoning in wildlife, especially within the range of the California condor. Overall, the general public knows little of this problem. The use of viable, non-toxic ammunition should also be encouraged when possible. Non-toxic shotgun ammunition is already commercially available and non-toxic rifle ammunition, called Ultimet, may soon be available (Oltrogge personal communication). Until then, education is an important step that should be expanded. Currently, clean food subsidy is provided at all release sites. In addition, we support placing clean food at specific locations near known nest sites as they are discovered.

In response to increasing losses to the reintroduced population and a growing number of unknown deaths and disappearances, Ventana Wilderness Society initiated an intensive (weekly) aerial tracking program for all condors in California beginning in fall of 2000 to augment the ongoing ground tracking effort. The authors also support the use of satellite telemetry on key individuals, especially those actively feeding on their own to discover sources of lead contamination and other threats.

Currently, condors are released both with and without post-release



California Condor, stud book #180, free-flying over Ventana Wilderness, California. Photograph by Kelly Sorenson

management facilities in the field depending on the release site. The authors support the use of post-release management facilities at condor release sites. Ventana Wilderness Society uses a "double-door," walk-in trap attached to post-release management facilities, that enables field managers to easily capture previously released condors. Routine evaluations of blood-lead levels can be achieved with minimal stress to individual condors and reduced staff effort. By increasing the level of monitoring both in terms of tracking and captures, field crews may be able to further reduce fatalities.

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Raptors as Vermin: A History of Human Attitudes towards Pennsylvania's Birds of Prey

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Abstract
Many species of raptors (hawks, eagles, and falcons) were considered vermin in Pennsylvania well into mid-twentieth century. Indeed, as recently as the 1930s and 1940s, even eminent conservationists were calling for the elimination of so-called harmful birds of prey. Raptors were unprotected in Pennsylvania throughout the eighteenth and nineteenth centuries, and in 1885 a 50-cent bounty was placed on all species of raptors. Although this particular bounty was repealed several years later, other bounties on diurnal raptors occurred sporadically until 1951. Bounties on several species of owls remained in force until 1969. Raptor protection, focusing on so-called beneficial species, first occurred in 1937. Bird-eating hawks, however, received only partial protection until 1969, and not all owls were protected until 1972, when the Migratory Bird Treaty Act of 1918 was amended to include birds of prey. At least part of the change in attitudes towards raptors can be attributed to activities at Hawk Mountain Sanctuary, the world's first refuge for birds of prey, which was founded in Kempton, Pennsylvania, in 1934. Over the past two decades, populations of Pennsylvania's raptors have rebounded from shooting and pesticide-era lows of the early and mid-twentieth century. Recently, many hunters and bird watchers in the state have suggested that populations of raptors may once again be too high. As a result, Pennsylvania's raptor conservationists again face some of the same human attitudes their predecessors faced more than a century ago.

Introduction
Raptors and other vermin (i.e. harmful or objectionable animals) were unprotected in Pennsylvania throughout the eighteenth and nineteenth centuries. Persecution of raptors in the commonwealth increased substantially in the latter half of the 1800s, when an overwhelming majority of rural residents considered raptors highly injurious. By 1885, animosity toward predatory birds had intensified to the point that the state legislature placed a 50-cent bounty on the heads of all diurnal birds of prey, as well as on all owls. During the next two years, 180,000 scalps were sent to the state capital in Harrisburg, by which time increased populations of destructive rodents and insects, together with fraudulent claims and a drain on the state treasury, induced the Pennsylvania legislature to repeal what by then many were calling the "fool hawk law" (Hornaday 1914). Raptors remained unprotected in the state until 1937, when all species of diurnal birds of prey, except for three accipiters (sharp-shinned hawk, *Accipiter striatus*; Cooper's hawk, *A. cooperii*; northern goshawk, *A. gentilis*) first received protection (Kosak 1995). Unfortunately, the new law was not particularly popular among Pennsylvania's hunters and farmers, and scant enforcement within the state continued to plague so-called "protected" species well into the 1960s. A bounty established on northern goshawks in 1929 was lifted in 1951, but it was not until 1969 that Pennsylvania granted this species, along with sharp-shinned and Cooper's hawks, full protection. Great horned owls (*Bubo virginianus*) remained unprotected statewide until March 1972, when the United States and several foreign signatories ratified an amended Migratory Bird Treaty Act of 1918. Since then, all species of diurnal birds of prey and owls have been protected in Pennsylvania. Here I detail how shifts in attitudes, both within and outside of the conservation community, contributed to this history.

Raptors as vermin
Raptors have had close associations with humans throughout history. Many longstanding human-raptor associations are positive, including the use of raptors in falconry, as national emblems and symbols of strength and courage, and as flagship species for broader conservation concerns. Unfortunately, raptors also have been scorned and feared, usually out of ignorance. Because of this they have been and continue to be heavily persecuted (e.g., Burnham 1990; Zalles

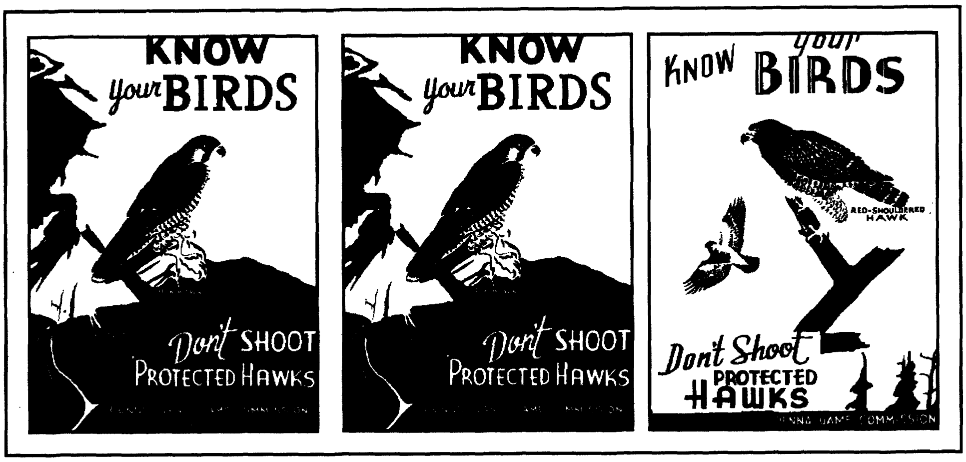

Figure 1. A series of three Works Progress Administration posters from the late 1930s commissioned by the Pennsylvania Game Commission in an attempt to help educate hunters regarding the identification of 'good', and therefore protected species of raptors (e.g., Red-shouldered Hawks and Peregrine Falcons), versus 'bad' hawks (e.g., Northern Goshawks). Note that the 'bad' goshawk is labeled "unprotected."

and Bildstein 2000). Indeed, to paraphrase Paul Errington (1946), "whatever else may be said of raptors and their predatory habits, they certainly do draw attention."

Systematic efforts to exterminate birds of prey can be traced to seventeenth century England (Gensbol 1984). Attempts at eradication, however, escalated substantially in the 1800s, when the advent of breech-loading guns increased the popularity of small-game hunting and placed hunters in direct competition with raptors (Newton 1990). Polls taken at the time of passage of the Pennsylvania Scalp Act of June 1885, a law that established a 50-cent bounty on birds of prey, suggested that the Act was supported by more than 90% of the public in most areas of the state. Not surprisingly, contemporary conservationists, including the president of the New York Zoological Society, William T. Hornaday, who referred to the Act as the "fool hawk law" (Hornaday 1914), and even Pennsylvania state veterinarian and author of *Diseases and enemies of poultry*, Leonard Pearson, considered the Scalp Act unjust, uneconomic, and simply wrong-headed (Pearson 1897). Although the law was rescinded in 1887, the commonwealth reinstated bounties on raptors in 1913, and maintained them on some species until well into the 1960s (Kosak 1995).

'Good' versus 'bad' hawks

Most conservationists who opposed the Scalp Act took exception to the Act's all-encompassing nature—American kestrels (*Falco sparverius*) were targeted along with northern goshawks—rather than to the notion that some hawks merited destruction (Hornaday 1913, 1914, 1931; Pearson 1897). The last half of the nineteenth and first half of the twentieth centuries, was an era of 'good' (i.e. rodent-eating) and 'bad' (i.e. bird-eating) hawks (Fisher 1893) (Figure 1). The idea that individual hawks and, in some instances entire species of hawks, were "chicken hawks," and as such merited destruction, extended well into the conservation community.

The writings of conservationists of the era reveal the animosity held for some raptors. Consider, for example, this passage from John Muir's *The story of my boyhood and youth* (1913): "When I went to the stable to feed the horses, I noticed a big white-breasted hawk [most likely a red-tailed hawk, *Buteo jamaicensis*], on a tall oak in front of our chicken house, evidently waiting for a chicken breakfast... I ran to the house for a gun, and when I fired, he fell... then managed to stand erect. I fired again to put him out of pain. He flew off... but then died suddenly in the air, and dropped like a stone."

Although the event that Muir related took place when he was a young boy in 1850s Wisconsin, the founder of the Sierra Club expresses no remorse when recalling it in his autobiography more than half a century later (Muir 1913).

Renowned conservationist William T. Hornaday's world of animal protection also included both "good" and "bad" hawks: "... 'chicken hawk or hen hawk' are usually applied to the red-shouldered [*Buteo lineatus*] or red-tailed species. Neither of these is really very destructive to poultry, but both are very destructive to mice, rats and other pestiferous creatures... Neither of them should be destroyed—not even though they do once in a great while, take a chicken or wild bird," however "[t]here are several species of birds that may at once be put under the sentence of death for their destructiveness of useful birds, without any extenuating circumstances worth mentioning. Four of these are Cooper's hawk, the sharp-shinned hawk, pigeon hawk [or merlin, *Falco columbarius*] and duck hawk [or peregrine falcon, *F. peregrinus*] " (Hornaday 1913).

Hornaday's distinction appears to have been both moralistic and utilitarian: "The ethics of men and animals are thoroughly comparative... Guilty animals, therefore, must be brought to justice" (Hornaday 1922). By 1931, Hornaday had dropped the Merlin from the list of "pest" birds, apparently because of its rarity, but retained the others, along with the great horned owl

(*Bubo virginianus*), barred owl (*Strix varia*), and eastern screech-owl (*Otus asio*) (Hornaday 1931).

The broader ornithological and birdwatching communities, too, took aim at certain raptors during the first three decades of the twentieth century. The prolific and highly regarded bird artist, Louis Agassiz Fuertes, writing in *National Geographic*, for example, commented that "The whole genus *Accipiter*, consisting of the [northern] goshawk, Cooper's hawk, and sharp-shinned hawk, are savage, bloodthirsty, and cold-hearted slaughterers, and are responsible in large measure for the anathema that is then portion of all hawks" (Fuertes 1920). And famed birdwatcher and president of the Connecticut Audubon Society, Mabel Osgood Wright, suggested helping songbirds by "shooting some of their enemies" including several species of hawks (Wright 1936). Similarly, Pennsylvania's official "state ornithologist," and later president of the Wilson Ornithological Society, George Miksch Sutton, wrote in the *Introduction to the birds of Pennsylvania* that: "[t]he sharp-shin is the enemy of all small birds...[and that it] and [the] Cooper's hawk, both bird killers, are fairly common and are rated as our most objectionable birds of prey. They are not protected in Pennsylvania." (Sutton 1928a).

Even Boy Scouts were instructed in the whys and wherefores of "good" and "bad" hawks. George E. Hix, a Brooklyn, New York, scoutmaster and Associate of the American Ornithologists' Union wrote in his *Birds of prey for Boy Scouts* "... [that] the beneficial hawks are the larger, slower species, [and that] the smaller swifter hawks are the ones which are destructive to wildlife... [and that these include] the goshawk, Cooper's, sharp-shinned, duck and pigeon hawks..." (Hix 1933).

Small wonder then that bird-eating birds were heavily persecuted in the first half of the twentieth century. Unfortunately, because many of the shooters either were unwilling or unable to separate the 'bad' hawks from "good," all species of raptors remained at risk (Broun 1949; Kosak 1995) (Figure 2). The impact of such shooting was little studied, probably because much of the conservation community condoned or even favored it. Even so, banding recoveries for Cooper's Hawks suggest that first year mortality due to shooting ranged from 28 to 47% in 1929 to 1940, and from 12 to 21% as recently as 1946 to 1957 (Henny and Wight 1972).

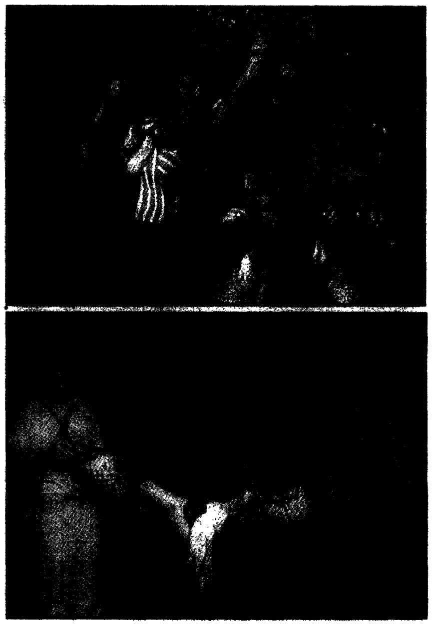

Figure 2. The results of raptor shooting in Pennsylvania in the late 1950s, when Sharp-shinned Hawks (top) remained unprotected, while Ospreys (*Pandion haliaetus*) (bottom), were protected (photographs courtesy Hawk Mountain Sanctuary archives).

The goshawk invasion of the late twenties

As hated as resident accipiters were, migrants from the north were despised even more (Gerstell 1937). Northern Goshawks, in particular, were singled out in this regard. As ornithologists Edward H. Forbush remarked in his *Birds of Massachusetts* "A great flight of goshawks into the United States in fall or winter is followed invariably by a scarcity of Ruffed Grouse [*Bonasa umbellus*]" (Forbush 1929). Thus, when rural inhabitants of Drehersville, at the base of the Kittatinny Ridge in the central Appalachian Mountains of eastern Pennsylvania, reported an "invasion" of northern goshawks during the winters of 1926-1927 and 1927-1928 to the Pennsylvania Game Commission, the commission quickly dispatched state ornithologist George Miksch Sutton to investigate.

Sutton published his initial findings in the *Wilson Bulletin* in 1928 (Sutton 1928b). His report, together with a second paper published three years later (Sutton 1931), initiated a series of events that eventually resulted in the creation of the world's first Sanctuary for birds of prey.

First, by 1929, the Pennsylvania Game Commission was offering a new five-dollar bounty on northern goshawks shot between 1 November and 1 May. Not surprisingly, the bounty substantially increased raptor shooting at the site, as well as throughout Pennsylvania. Second, Sutton's articles alerted the ornithological and conservation communities of both raptor migration and shooting at the site, which the locals called Hawk Mountain.

Earl Poole, then assistant curator at the Reading Public Museum in nearby Reading, Pennsylvania, began visiting Hawk Mountain in 1929. His description of a broad-winged hawk (*B. platypterus*) flight of 2,000 birds on 22 September 1932, represents the first detailed account of raptor migration at the site (Poole 1934). Shortly thereafter, conservationists Henry H. Collins, Jr., and Richard Pough visited the ridge, confirming Sutton and Poole's earlier accounts, and photographing the slaughter that was then underway (Pough 1932; Collins 1933). Pough showed slides of his photographs to a joint meeting of the Hawk and Owl, Linnean, and National Association of Audubon soci-

Figure 3. The view from Hawk Mountain Sanctuary, the world's first refuge for birds of prey, in the 1940s. (Photo: Hawk Mountain Sanctuary archives.)

eties meeting in New York City in autumn 1933. Rosalie Edge, the head of the Emergency Conservation Committee, was in the audience the evening Pough spoke.

The following June, Mrs. Edge, accompanied by Richard Pough and her son Peter, visited Hawk Mountain. Shortly thereafter, Rosalie Edge leased the 565 acres that was to become Hawk Mountain Sanctuary (Bildstein and Compton 2000). Today, the Hawk Mountain Sanctuary Association, the largest and oldest member-based raptor conservation organization in the world, manages the Sanctuary, which has now grown to more than 2,300 acres. Once a shooting gallery where thousands of raptors were killed each autumn, Hawk Mountain's Appalachian lookouts now serve as both learning center and biological field station for more than 80,000 visitors annually (Zalles and Bildstein 2000) (Figure 3).

Unfortunately, the Pennsylvania bounty on goshawks (reduced to two dollars in 1937) remained in place until 1951, when the state legislature approved, and Governor G. H. Earle signed a law protecting all raptors, excepting the three species of "bird-killing" accipiters and two species of owls, the great horned owl and the snowy owl (*Nyctea scandiaca*) (Kosak 1995).

The founding of Hawk Mountain Sanctuary highlighted raptor conservation both within and outside of Pennsylvania as never before. Thirty-five years of conservation and education efforts at the Sanctuary, in part, resulted in statewide, year-round protection for all diurnal birds of prey, including the three "bird-killing" accipiters, in 1969 (Senner 1984). It was not, however, until the Migratory Bird Treaty Act of 1918 was amended in March of 1972, that all raptors, including great horned and snowy owls, were protected in Pennsylvania (Kosak 1995).

A modern era?

Although all raptor species remain officially protected in Pennsylvania, and have for almost thirty years, their status remains controversial. As recently as 1999, the Pennsylvania Game Commission held hearings on a proposal from the President of the Commission (game commissioners are appointed by the Governor) regarding "experimentally" controlling populations of Red-tailed Hawks and Great Horned Owls on several wildlife management areas in an attempt to increase the survivorship of Ring-necked Pheasants (*Phasianus colchicus*) (Riegner 1999). Although a public hearing revealed widespread opposition to the proposal—that was later withdrawn—letters to the editors of local newspapers also suggested a degree of public support for the idea (e.g., Riegel 1999).

Indeed, many birdwatchers, especially those maintaining backyard birdfeeders, continue to call Hawk Mountain Sanctuary to express outrage at the seemingly persistent predatory activities of sharp-shinned and Cooper's hawks at their bird feeders. Both species of accipiters appear to be increasingly willing to enter human-dominated landscapes; most likely in response to reduced human-caused mortality there. Although most callers seem resigned to this situation, particularly once they have been informed that removing a single accipiter from a backyard is likely to be as ineffective as removing a single Gray Squirrel (*Sciurus carolinensis*), others appear determined to "do something themselves" about the situation, including, a few have suggested, shooting the hawk (Bildstein, personal observation).

Research conducted at Hawk Mountain Sanctuary and elsewhere suggests that recent shifts in the migration behavior of eastern populations of sharp-shinned hawks may be related to an increasing tendency to pause at bird feeders along migration routes in the northeastern United States (Viverette et al. 1996; Duncan 1996). Furthermore, data collected by participants in the Cornell University Laboratory of Ornithology's Project Feederwatch indicate that sharp-shinned hawks take more birds at birdfeeders than do feral cats (Dunn and Tassaglia 1994). Whether or not increased numbers of accipiters at bird feeders are impacting regional populations of songbirds and other species feeding at these sites remains unclear, although evidence from England suggests that this may not be occurring (Newton et al. 1997). On the other hand, studies of American kestrels that overwinter in farmlands surrounding Hawk Mountain suggest that recently increased populations of Cooper's and, possibly, sharp-shinned hawks, both of which prey on kestrels, are affecting regional populations of this small falcon (Ardia and Bildstein 1997; Ardia et al. unpublished data).

Conclusion

Raptor populations have increased substantially in the last two decades of the twentieth century (Bednarz et al. 1990; Bildstein 1998), quite possibly to levels similar to those of the latter nineteenth and early twentieth centuries. In Pennsylvania and, indeed throughout most of North America, birds of prey are no longer the endangered boutique predators (i.e. predators whose populations are so low that they do not substantially influence the behavior and ecology of their prey species) that they were as recently as the late 1970s (Bildstein 1998). Now that raptors once again are fairly common and fully functional components of natural and human-dominated landscapes, raptor conservationists are facing many of the same management concerns and human attitudes their predecessors faced more than one hundred years ago.

As a result, keeping common raptors common at the beginning of the twenty-first century may prove to be as much of a challenge for today's raptor conservationists (Garrott et al.1993) as it was for their predecessors at the beginning of the 20th Century. Experience at Hawk Mountain Sanctuary suggests that focused conservation education that extends from primary schools through the general public, longterm population monitoring, and, above all, opportunities for viewing large numbers of these normally secretive birds at migration hawkwatches, are practical and effective ways to build local and regional support for our charismatic birds of prey. With this in mind, the Sanctuary continues to work with local, regional, and national conservationists to foster migration watchsites elsewhere in the world so as to protect long-distance migratory raptors throughout their journeys.

Acknowledgments

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Felid Conservation

Status and Conservation of Endangered Cats Along the U.S.-Mexico Border

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Abstract
*This paper reviews the research and conservation projects associated with the Bordercats Working Group, a group of scientists and advocates concerned with the status of neotropical cats in northern Mexico and the American Southwest. For ocelots (*Leopardus pardalis*), jaguarundis (*Herpailurus yaguarondi*) and jaguars (*Panthera onca*), this region is the northernmost limit of their geographical distribution. We review the results of our field surveys for bordercats in Arizona, Chihuahua, Coahuila, New Mexico, Nuevo Leon, Sonora, Tamaulipas, and Texas; describe a GIS-based habitat mapping project for bordercats; and discuss what conservation-related activities are critical for the long-term survival of the cats in the border regions.*

Introduction
There are three cat species whose northernmost range coincides with the border region of the United States and Mexico. These are North America's neotropical cat species: the ocelot *(Leopardus pardalis)*, jaguar *(Panthera onca)*, and jaguarundi *(Herpailurus yaguarondi)*. Unlike the mountain lion *(Felis concolor)* and bobcat *(Felis rufus)*, these species are endangered and protected by both the U.S. Endangered Species Act (ESA) and by Mexican law (NOM-059-1994), which prohibits hunting of the three species.

The border region presents an array of threats to neotropical cats, such as highway and international bridge construction, land development, and ranching practices that are intolerant of predators. In addition, the border region is a major thoroughfare for illegal aliens. The United States Border Patrol attempts to prevent such crossings by burning or mowing border vegetation, constructing fences, and putting up large floodlights across the border, which for nocturnal cat species are less than ideal. Recovery for neotropical cat populations in the border region requires not only an understanding of the spatial requirements and habitat needs of the cats but an understanding of how the cats can best fit into a complex human matrix consisting of farming and ranching communities, refugees, federal agencies, all converging at the international border. There are a suite of national and international laws, such as the President's National Drug Control Strategy, the National Defense Authorization Act, and U.S.-Mexican immigration policies, which often conflict with Federal environmental laws like the ESA and the National Environmental Policy Act (NEPA). In addition, Texas and states in northern Mexico consist primarily of private land, making a large refuge for the cats particularly difficult. Despite this, outstanding commitment to cat conservation by the U.S. National Wildlife Refuge system, particularly the Lower Rio Grande National Wildlife Refuge (LRGNWR) and Laguna Atascosa National Wildlife Refuge, both in Texas, has elevated public awareness about the cats and the challenges they face.

Addressing conservation needs: The Bordercats Working Group

The Bordercats Working Group was founded in 1998 to promote recovery and conservation for neotropical cats in the border region of the U.S. and Mexico. The group was created because of the great amount of biodiversity in the border region, evidence of few to no neotropical cats in this region in recent times, a gap in knowledge about the detailed distribution of the species in this region, and an existing legal framework in place to protect all three species.

The Bordercats Working Group (BWG), now a member of the World Conservation Union's (IUCN) Cat Specialist Group, has several objectives: to conduct research on all three species in the border region; to demarcate important habitats for the cats in the border region; to educate local communities and organizations about the cats; and to bring individuals, groups, and government agencies together to share in a common mission. This paper will focus on the first two objectives: the distribution of borderland cat populations, and the identification of important cat habitats in this region.

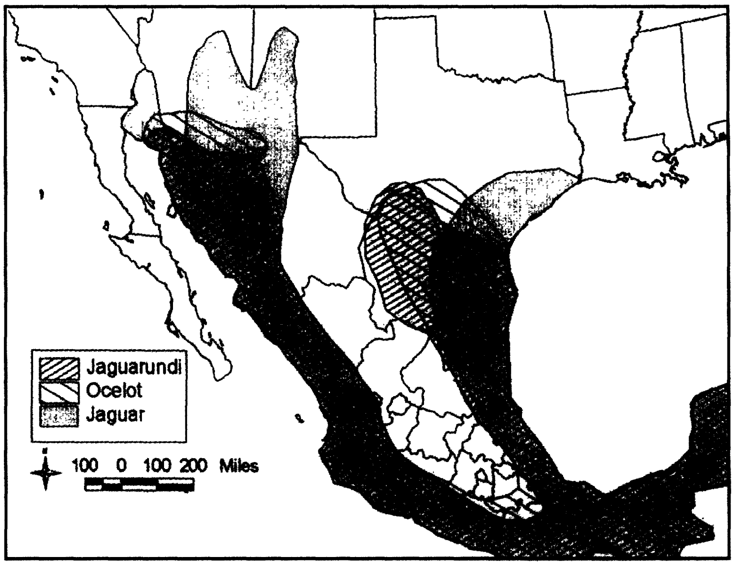

Figure 1. Historical distribution of jaguarundis, jaguars, and ocelots

Distribution of bordercat populations

The historic distribution (Figure 1) for all three cat species in the border region is located in the U.S. Southwest (Arizona, New Mexico, and Texas) and adjacent Mexican States (Chihuahua, Coahuila, Nuevo Leon, Sonora, and Tamaulipas). At present, there are approximately 100 ocelots in the United States and no known populations of jaguars or jaguarundis. From 1982 to 1996, 12 out of 17 radio-collared ocelots in south Texas were killed by automobiles (Tewes et al. 1997). BWG is involved with neotropical cat surveys in four regions: Arizona/New Mexico, Sonora/Chihuahua, Texas/Coahuila, and Tamaulipas/Nuevo Leon.

Arizona/New Mexico

The Chiracahua and Peloncillo Mountains were surveyed for cats for four months during the Spring and Fall of 1999. Field biologists used a combination of techniques to detect cat presence: scat surveys, track surveys, remotely-triggered camera surveys, and formal interviews with biologists and local naturalists. During this survey, no physical evidence of neotropical cats was obtained. There were, however, approximately 12 interviews held with knowledgeable locals, including biologists, who attested to the presence of jaguarundis. More surveys would have to be conducted to verify these reports.

Sonora/Chihuahua

Approximately 215 km south of the Arizona/New Mexico border, in the Mexican State of Sonora, lie important areas for both jaguars and ocelots; jaguarundis may occur just south of this area. Surveys for all three species, which incorporate formal interviews and remotely-triggered cameras, have been ongoing since June 1998. Formal interviews have occurred in 50 municipalities in Sonora and 12 in Chihuahua. Interviewees consisted of cattle association officials, ranchers, and predator hunters. Whenever possible, physical evidence (such as skins, skulls, and photographs) of recently killed cats was collected (Lopez-Gonzalez and Brown in preparation).

There are approximately three extant metapopulations of jaguars in the State of Sonora: Sierra Bacatete, Sahuaripa-Huasabas, and Quiriego-Sinaloa (Lopez-Gonzalez and Brown in preparation). Preliminary evidence suggests that this area supports approximately 1.3 to 1.5 jaguars per 100 km^2. Using a home range estimator (Adaptive Kernel model), a minimum of 6,000 km^2 would be needed to conserve viable metapopulations in this region (Brown and Lopez-Gonzalez 2000). This initial research has found that at least 140 jaguars have been killed in Sonora since 1900; 45 have been killed since 1990 (Lopez-Gonzalez and Brown unpublished report). A minimum of 14 females, three lactating females, and five kittens have been killed since 1989. In 1999 to 2000, sixteen jaguars were killed in the northern range of this distribution (Lopez-Gonzalez and Brown 2000). All of these animals were killed in response to livestock losses and none of the skins were sold internationally (Lopez-Gonzalez and Brown in preparation).

The closest known population of ocelots to Arizona/New Mexico is also in this area. From July 1998 to October 1999, 33 records of hunted ocelots were collected in this region; approximately 79% of these ocelots were killed after 1990 (Lopez-Gonzalez 2000). Although three ocelots have been taken in Sonora within 40 km from the Arizona border, all were male. Due to an average dispersal distance of approximately 20 km, it is unlikely that female ocelots would be found this close to the Arizona border.

Texas/Coahuila

In March 1999, BWG surveyed the Rio Grande River near Big Bend National Park, in Texas and the Mexican State of Coahuila for neotropical cats. Approximately 100 km of river and associated tributaries were surveyed during this trip. Transects walked along each sandy substrate adjacent to the river were scanned for cat tracks and scat. When tracks were found, they were measured and photographed, and plaster castes of each track set were made. A Global Positioning System (GPS) unit was used to measure the location of each track set. In addition, data on substrate, time of day, and general habitat characteristics were recorded for each track set. Scent-stations were set up prior to sunset by sifting sand over one m² areas, placed 10 meters from one another. Hawbawkers Cat Lure was spread over a stone and placed in the center of each one m² sampling unit. Each station was checked for cat tracks the following morning.

During this survey, we observed several cat tracks, including nine mountain lion track sets (seven adults and two cubs), six bobcat track sets, and what appeared to be one jaguarundi track set, all found at the mouth of the western end of the Boquillas Canyon (Figure 2). Big Bend National Park employees have collected numerous jaguarundi sightings from park visitors but currently have no physical evidence in support of this species inhabiting the park.

Tamaulipas/Nuevo Leon

BWG has ongoing research in both Tamaulipas and Nuevo Leon, just south of the Texas border. Survey efforts have focused in areas, within close proximity to the border, which have intact cat habitat and reports of ocelots or jaguarundis (Figure 3). Once suitable areas are identified, a combination of track and scat surveys, remotely-triggered cameras, predator calls, and box traps baited with chickens are used to detect cat presence. We are currently investigating two recent reports of jaguarundis in close proximity to Monterrey, Nuevo Leon, approximately 200 km from the international border (Area #1, Figure 3). We have no evidence of ocelots in this area; however, our surveys are not yet complete. We have surveyed in areas north of Monterrey and have not detected the presence of neotropical cats. Hence, the Monterrey region may be the northern distributional limit for jaguarundis in Tamaulipas/Nuevo Leon. Caso (1994) has been conducting long term research in an area further south of Monterrey, approximately 450 km south of the international border, near Tampico (Area #4, Figure 3). Caso has several ocelots and jaguarundis radio-collared in this study area, and is obtaining important information about their home range sizes and habitat use.

Identification of key bordercat habitats

In order to identify what borderland habitats are most important to neotropical cats, BWG initiated a Geographical Information System (GIS) habitat mapping project in 1999. BWG collected sighting data for each species, spanning the last 100 years. These sightings were plotted on a map containing the following information: vegetation, city and county boundaries, roads, rivers and streams, and elevation. Mapping participants, each familiar with a particular region or conservation unit (CU), were asked important questions about each species sighting map. The questions pertained to both the species and their habitats, major threats to each, the development of conservation corridors between CU's, and the effectiveness of the land-tenure system in place within each CU. The results of this large mapping project will be synthesized and a final map will be produced. The final map will be

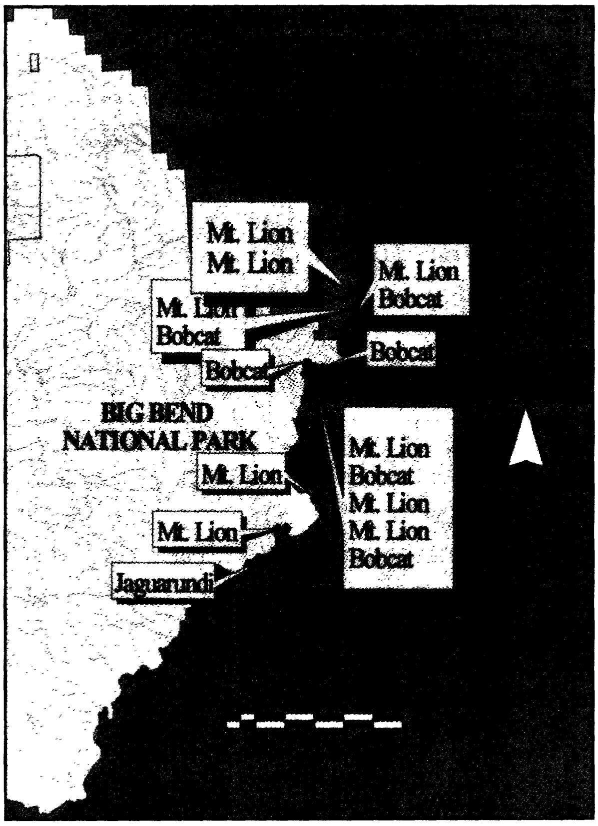

Figure 2. Details of bordercat sightings

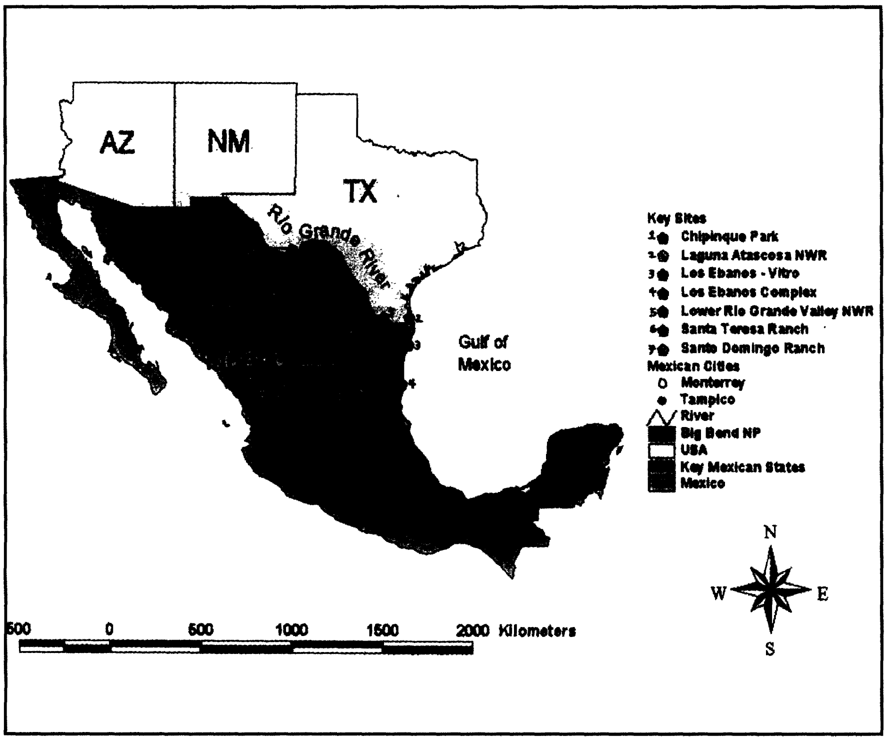

Figure 3. Survey sites for neotropical cats in NE Mexico

completed in Fall 2001 and will serve as a conservation blueprint for future priority research and habitat protection projects related to neotropical cats in the border region.

Conclusions

Obtaining valuable information about the distribution of neotropical cats and what habitats are most important to their conservation is a first step toward promoting their long term persistence in the border region. This research, however, needs to be married with effective environmental education programs that promote carnivore conservation at the community level. BWG is initiating environmental education projects with both ranchers and school children in the border region and developing partnerships with governmental agencies in the U.S. and Mexico to raise awareness about these important species. The results of our future research findings will only enhance these long-term partnerships, as additional information becomes available about the status and distribution of borderland cats.

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We wish to thank all members of the Bordercats Working Group for their ongoing participation, expertise, and commitment to cat conservation along the U.S./Mexico border. We thank Defenders of Wildlife, Round River Conservation Studies and the Walt Disney Company Foundation for their financial assistance, in addition to in-kind support from members of the Bordercats Working Group. The IUCN/Cat Specialist Group, in particular Peter Jackson and Kristin Nowell, is thanked for its guidance and leadership to BWG. Logistical support for field surveys was provided by Raymond Skiles and Betty Alex at Big Bend National Park. Alison Gabel, Gerald Scoville, Dennis Sizemore, Kevin Crooks, and volunteers from Round River Conservation Studies are thanked for conducting field surveys in Arizona and New Mexico. The field participation of Marcela Aranda, Dave and Nancy Foreman, Patricia Manzano-Fischer, Craig Miller, and Scott Royder during the Big Bend surveys was indispensible. John Morrison (World Wildlife Fund-U.S.A.), Eric Sanderson (Wildlife Conservation Society), Robert Thomas (University of California-Davis), and Diana Lawhorn (The Nature Conservancy) are thanked for their involvement in the GIS Habitat Mapping Project. We wish to thank John Morrison and Robert Thomas for producing all maps contained in this manuscript.

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Molecular Scatology as a Conservation tool

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Abstract
The threat of predation on livestock by large carnivores represents a major impediment to the conservation of intact ecosystems throughout the world. Although it has taken nearly a century to learn the truth about predation on livestock in North America, advanced technologies now exist to greatly expedite similar findings elsewhere. By examining the dietary ecology of puma and jaguar on a cattle ranch in the Venezuelan llanos, it is possible to help identify simple, effective methods to reduce livestock predation. In my study, dietary habits of the puma and jaguar were defined from feces (scats), kills, and ranch records. Scats were assigned to predator species through analysis of mitochondrial DNA from feces of wild carnivores. Based on dietary and ecological data of these two large carnivores, recommendations on livestock husbandry are made.

The puma (*Puma concolor*) and the jaguar (*Panthera onca*) are large sympatric carnivores trying to survive in increasingly fragmented habitats of the Neotropics. Much of the range of these endangered animals has been converted to ranchlands, which now hold some of the last remaining natural habitats in Latin America. In much of the Neotropics, the major cause of mortality for large felids is persecution by cattle ranchers for alleged predation on livestock. To conserve viable populations of large cats it is necessary to find some resolution to this problem. This study attempts to accurately identify the predator species involved in livestock depredation, and to provide a clearer understanding of carnivore interactions with livestock.

Dietary habits can be accurately determined noninvasively only by a thorough analysis of scats—the end product of predation. Unfortunately, accurate analysis has eluded scientists until recently. Scats have traditionally been sorted to donor species by diameter. In 1980 thin layer chromatography analysis of bile acids was introduced (Major et al.), and for more than a decade it was the most accurate method for determining donor species. But bile acid assays can give misleading results for closely related carnivores. The identity of puma and jaguar scats is based on the presence of one bile acid that appears in only 71% of jaguar scats, leaving a 29% possibility of mistaking jaguars for pumas (Taber et al. 1997). Nevertheless, diameter of puma and jaguar feces in Taber's study overlapped in almost all size ranges. The need for a more definitive analysis is apparent.

The advent of polymerase chain reaction (PCR) has enabled scientists to acquire information through molecular analysis. Höss et al. (1992) used genetic techniques to analyze mitochondrial DNA (mtDNA) from sloughed off colon cells found in wild scat in order to identify Italian bears. By isolating DNA from scats and checking it against reference samples, it is now possible to accurately determine which species, and sometimes even which individual left the sample (Kohn et al. 1995; Kohn et al. 1999; Ernst et al. 2000).

Molecular scatology can provide new insights into the carnivore diet, and help lead to solutions for a range of conservation problems (Kohn and Wayne 1997). With the use of scat-sniffing dogs, non-biased recovery of samples from multiple target species is now feasible (Wasser et al. 1999), making population estimates for similar-sized sympatric species possible. By analyzing the prey content of scats from undisturbed habitats, we can determine the preferred prey of carnivores. If we manage this prey appropriately, attacks on livestock by hungry carnivores can be reduced.

The molecular method is completely noninvasive, though removing all scats may confuse animals by removing territorial markers, which also signal reproductive status (Brown et al. 1994). The field collection methods for this study were simple and effective; samples were collected, air-dried, and then stored at room temperature. Using these collection and storage methods, a recent lab run produced positive identifications from scats that were seven to ten years old (in preparation). Samples were transported without freezing or excess bulk, hazardous chemicals, or cumbersome permit restrictions. Scats are the only item from Appendix I species exempt from CITES controls (Gerloff et al. 1995), although other restrictions, such as those imposed by the USDA, may apply.

Carnivore scats were collected opportunistically from roads and trails on Hato Piñero, a cattle ranch in the llanos, the seasonally flooding savanna of western Venezuela. Samples were dried and a small portion of each scat was stored for genetic analysis (Farrell et al. 2000). Samples were then analyzed for prey contents; remains were visually identified to the lowest taxon possible by examining teeth, claws, bones, fur, feathers, and scales. For this study, all mammals less than one kilogram were considered small—mostly rodents, but including small marsupial mouse opossums (*Marmosa sp.).* Medium mammal prey ranged from 1 to 15 kilograms, including armadillo (*Dasypus novemcinctus*), rabbit, sloth (*Bradypus variegatus*), opossum (*Didelphis marsupialis*), juvenile peccary, and newborn calves (*Bos bos*). Large mammals were anything over 15 kilograms, such as adult capybara (*Hydrochaeris hydrochaeris*) and peccary, deer, giant anteater (*Myrmecophaga tridactyla*), and juvenile and adult cows.

In the llanos, the ocelot (*Leopardus pardalis*) and crab-eating fox (*Cerdocyon thous*) each average about six to 10 kg in body weight, the puma about 40 to 50 kg, and the jaguar up to 100 kg (Figure 1). But, body size is not necessarily indicative of scat size. Puma scats have been identified down to 19mm in diameter, and ocelot scats up to 27 mm. Because of this overlap, it was necessary to define prey contents of almost all size predator scats to accurately define the dietary habits of puma and jaguar. Frequency of occurrence of prey types were separated into small and large predator categories—first based on size of scat (traditional method) and then based on DNA identification.

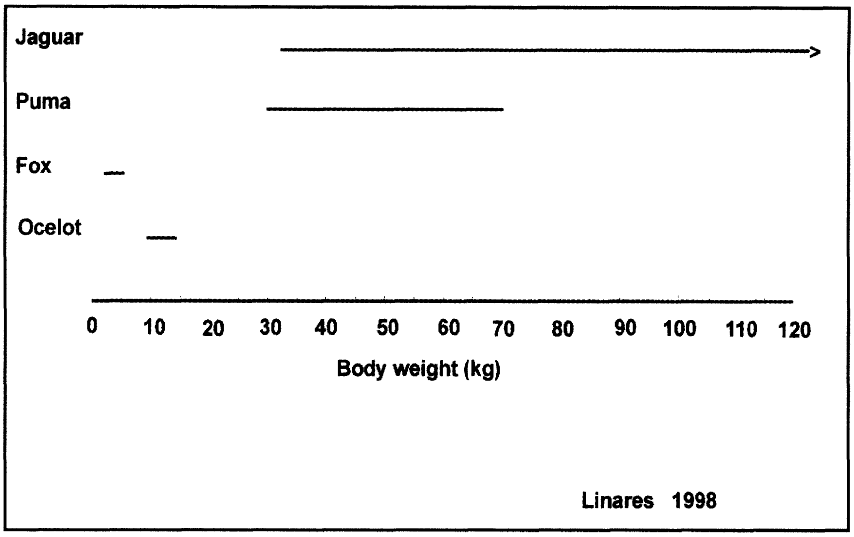

Figure 1. Overlap in body size of the four main carnivores on Hato Piñero, Venezuela (Linares 1998).

For the preliminary analysis by scat size, scats were split into large, assumed to be puma and jaguar, and small, assumed to be ocelot and fox. Based on previous studies (L. Emmons, personal communication; Fernandez et al. 1997), 25 mm dry diameter was chosen as an arbitrary breaking point large enough so that it was beyond the range of most fox and ocelots.

For the molecular analysis, a portion of the mitochondrial cytochrome b gene was chosen as the DNA marker to assign unknown scats to predator species. Samples were washed, and DNA extracted from the isolate and amplified using PCR (Farrell et al. 2000). The resulting strands of DNA were sequenced and compared to samples extracted from blood of the carnivore species at the study site. Even this test wasn't perfect at first—one sample that could not be matched to a known carnivore was compared to sequences posted in GenBank using the BLAST program (National Center for Biotechnology Information (NCBI), www.ncbi.nlm.nih.gov) to find a cytochrome b sequence with the closest match. When that came back as a fruitfly (*Drosophila melanogaster*; likely apprehended while the scat was collected), primers that targeted carnivores were designed. The sample was reamplified with these new primers and found to be a perfect match with our ocelot reference. Targeting a shortened fragment of 147 base pairs also increased the percentage of successful samples, because fecal DNA is often degraded. Twenty of 34 samples (59%) were successfully sequenced.

Three jaguar and two fox scats were identical to control samples, as were five puma scats (one puma scat differed by one base pair). The sequences from 10 ocelots showed more variation, which makes sense in light of a study by Eizerick et al. (1998) showing a relatively high degree of intraspecific variation among the smaller Neotropical felids.

Puma, ocelot, and fox scat sizes overlap in a narrow range around 25mm diameter, and puma and jaguar overlap at larger diameters between 32 and 37 mm (Figure 2). One individual can leave scats of various sizes, and the range of body

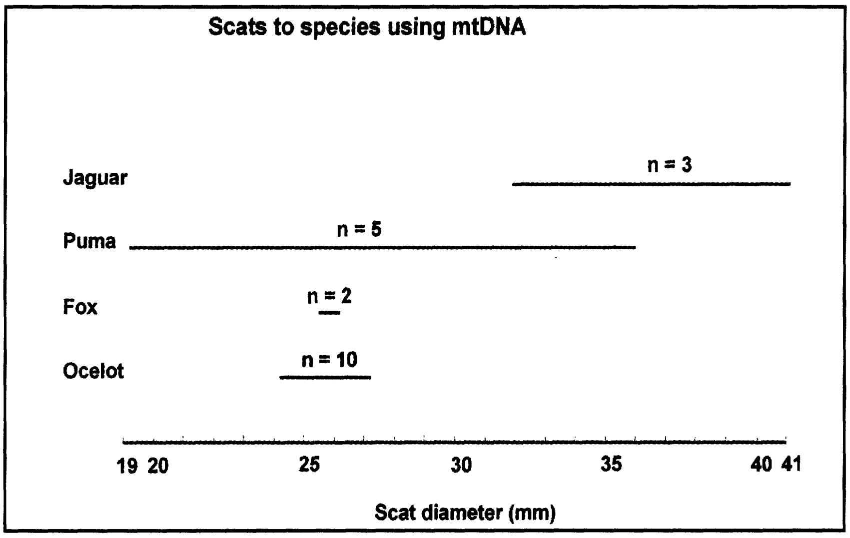

Figure 2. Overlap in scat sizes of the four main carnivores when the scat identity is confirmed by molecular analysis.

sizes can vary greatly within species. For example, male jaguars in the llanos average about 100 kilograms, whereas males in the Central American rain forest average 55 kg.

Using size to define dietary analyses, smaller carnivores appear to ingest solely mammalian prey (Table 1, left side). They even appear to take large prey, though the thought of an ocelot or fox subduing and killing an 18 kg collared peccary (*Tayassu tajacu*) may seem unusual. Large predators appear to take prey from all categories, predominantly small mammals.

DNA analysis of samples shows that almost the inverse is true. Large carnivores prey exclusively on medium and large mammals, and small carnivores take a wide range of prey, including everything except large mammals (Table 1, right side). Analysis of one scat with a size indicative of a small ocelot (19.25mm), clarified that it was a puma who ate the peccary.

McNemar's test (Harrison 1996) showed the difference between identification of scat donor by size and by genetic analysis to be highly significant (X_ = 7.36, $p<0.01$, 1df). With differentiation by size, 83% of ocelot and fox scats were misclassified as large predator scats, and 12% of puma and jaguar scats were misclassified as small predator scats. These findings are preliminary and the sample size is small, dietary results may change when scats from the last two years of the three year project are analyzed.

With DNA analysis, predators are discernable to the species level (Table 2). The biomass of each prey type is then calculated to learn its relative importance in each carnivore's diet (Farrell et al. 2000). Dietary resources are divided more between large and small carnivores

Table 1. A comparison of prey frequencies with scats separated by size and molecular analysis. One jaguar scat was disqualified from the dietary analysis for containing bait (from Farrell et al. 2000). Small predators = Fox and ocelot. Large predators = Puma and jaguar.

Frequency (%) of Prey Types				
	Scat size		mtDNA	
	Small predators ≤25mm	Large Predators ≥25mm	Small Predators	Large Predators
Small mammals	75	46	64	-
Medium mammals	12.5	12	3	50
Large mammals	12.5	9	-	50
Reptiles	-	9	9	-
Birds	-	12	12	-
Fish	-	3	3	-
Crabs	-	9	9	-
	n=3	n=16	n=12	n=7

Table 2. Frequency of prey biomass with predator scats identified using molecular analysis (from Farrell 1999).

	Jaguar n = 2	Puma n = 5	Ocelot n = 10	Fox n = 2
Small mammals	-	-	26	34
Medium mammals	28	45	22	-
Large mammals	72	54	-	-
Reptiles	-	-	43	-
Birds	-	-	7	23
Fish	-	-	-	31
Crabs	-	-	2	12

Table 3. Predation on cattle (calves and adults) by puma and jaguar on Hato Piñero from January 1987 through October 1996, as reported by ranch workers (Hato Piñero, unpublished data; from Farrell 1999). *There was found to be a discrepancy of 27% between causes of livestock mortality reported by ranch workers and evidence at the site when reported incidences of predation were investigated in 1999.

	Total	Puma	Jaguar
Number of mortalities attributed to predation*	310	275	35
% Predation		89	11
% of total cattle mortalities from all causes (n=3672)	8.4	7.5	<1

than between same size sympatric carnivores; puma and jaguar versus fox and ocelot. Jaguar scats contained a peccary and a cottontail rabbit (*Sylvilagus floridanus*). A third was disqualified from the dietary analysis because it contained a wild pig set as bait while trapping cats for radio-telemetry. Five puma scats contained a couple of juvenile peccaries, an armadillo, one deer (*Odocoileus virginianus*), and a domestic guard dog (*Canis familiarus*), confirming the suspicion of ranch workers that their missing tag-along dog had been consumed by a puma. The samples containing dog and pig bait illustrate that this method can be used to pinpoint problem predators.

Dietary data derived from the genetic analysis, along with information from kills recovered during this study and ranch records, can offer suggestions to help decrease livestock predation. Livestock and wild kills were examined when located (n=15) and habitat conditions were recorded to discern where cats kill, and what conditions are conducive to predation. Seven verifiable incidences of livestock predation were examined, and six of these confirmed as pumas (Farrell 1999). Frequency and location of predation by cats on cattle was determined from 10 years of ranch records that reported causes of livestock mortality (Table 3; Hato Piñero, unpublished data). When observations of livestock killed during my study are combined with evidence from ranch records and molecular data, the preference for different sizes of prey between the two cats becomes clearer. Jaguars select for very large adult prey. They took only a very small percentage of the total cattle lost (<1% according to ranch records), yet these cows were typically full grown and a greater financial loss per head. As the more endangered of the two felids, perhaps jaguars should be allowed to keep practicing business as usual, and a compensation program for ranchers be initiated. Pumas are pinpointed as a greater problem; they are more general in prey selection and concentrate on medium and large size prey—often in the form of newborn and juvenile calves. Their preference for calves and juveniles when preying on livestock corroborates data from genetic identification of puma scats that revealed no prey item larger than a juvenile calf.

While jaguars prefer moister habitats of lower elevations; the pumas preference for dry habitat types is significant. Pumas move to dryer, nonflooding ranges for the wet season. Unfortunately cattle are also moved to these areas at the same time. The birthing of inseminated cows (each representing an investment of over $100 US) is timed as well for the beginning of the wet season in pastures along the edges of the ranch. In these areas humans harvest much of the natural prey, so it is no surprise that these pastures exhibit the greatest incidences of livestock predation. Pumas arriving thin and hungry from desolate ranges at the end of the dry season are greeted by a landscape flush with naive, newborn prey that are just learning to walk.

It is important to incorporate different types of data and the views of different parties to create solutions that will be effective in helping both predators and local ranchers. This study verified that pumas are a greater problem than jaguars on Hato Piñero. By determining which prey carnivores favor through scat analysis, management of the preferred prey can help reduce predation problems. To test deterrence, electric fencing was used in one of the hardest hit maternity pastures during a later phase of this study, and found to curtail predation by 100% (Scognamillo et al. 1999). One noteworthy finding of my study is that when water buffalo (*Bubalis bubalis*) were placed with cattle, predation mortalities in one long-suffering maternity pasture decreased close to zero (Andres Rodriguez, personal communication). Water buffalo are already being raised on this and other ranches throughout the llanos for meat and cheese and mix well with cattle, while apparently affording them some protection from predation (Farrell 1998).

Vast portions of puma and jaguar ranges are dedicated to livestock production—often on ranches of 80,000 to 120,000 hectares that contain large areas of natural habitat. Owners who are confident that their stock is safe are less likely to go hunting for these elusive cats. Reducing the threat of felid predation on livestock will have broad implications for future conservation of jaguar and puma in the Neotropics.

Acknowledgements

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Linking Snow Leopard Conservation and People-Wildlife Conflict Resolution: Grassroots Measures to Protect the Endangered Snow Leopard from Herder Retribution

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Abstract
*Livestock depredation has become a significant problem across the snow leopard's (*Panthera uncia*) range in Central Asia, being most severe in and near protected areas. Such predation, especially incidents of "surplus killing," in which five to 100 or more sheep and goats are lost in a single night, almost inevitably leads herders to retaliate by killing rare or endangered carnivores like snow leopard, wolf, and lynx. Ironically, such loss can be avoided by making the night-time enclosures predator-proof, improving animal husbandry techniques, educating herders on wildlife conservation and the importance of protecting the natural prey base, and by providing economic incentives like handicrafts skills training and marketing, along with carefully planned ecotourism trekking and guiding. The author explores innovative conservation initiatives in the Himalaya (Ladakh and Tibet) and Mongolia, which also build local capacity, self-reliance, and stewardship for nature using Appreciative Participatory Planning and Action, or APPA, techniques. The most sound conservation investments are those contingent upon establishing direct linkages with biodiversity protection, ensuring co-financing and reciprocal responsibility for project activities, encouraging the full participation of all stakeholders, and assuring regular monitoring and evaluation of the village-based agreements (embodied in Action Plans).*

Introduction
Wildlife damage is a major source for conflict between local communities and protected areas managers in the Himalaya (Kharel 1997; Jackson et al. 1996; Oli et al. 1994). In India's Kibber Wildlife Sanctuary, Mishra (1997) noted that 18% of livestock holdings were killed by snow leopard (*Uncia uncia*) and wolf (*Canis lupus*) for an estimated total value of US $128 per household per annum—a very significant economic impact given per annual cash incomes of $200 to $400. Villagers claimed predation rates increased after sanctuary establishment, while surveys indicated dramatic increases in livestock numbers accompanying changes in animal husbandry systems (Mishra 2000).

A similar situation in Hemis National Park, Ladakh, Jammu and Kashmir State, led to the establishment of a compensation scheme, but within two years the sponsoring Ladakh Wildlife Department found itself committing 60% of its annual $26,000 budget to the program. Payment takes up to two years, with claimants being paid only 10 to 30% of their animal's market value. Understandably relations between local people and the park have deteriorated, with retaliatory killing constituting a major threat to both snow leopard and wolf. Because local livelihoods are intimately bound with long-standing patterns of agro-pastoralism, relocation of people or the exclusion of livestock is not a feasible solution. Rather, local people's willingness to co-exist with predators hinges upon reducing depredation to an acceptable level while improving incomes to help offset unavoidable losses of livestock.

This field note describes grassroots initiatives being undertaken in Hemis National Park to alleviate livestock loss to predation and to encourage herders to become effective stewards of the snow leopard, its prey and its habitat.

Community-based conflict alleviation initiatives in Hemis National Park
Established in 1981, this park covers 3,350 square kilometers in the TransHimalayan Range (Fox and Nurbu 1990). Besides offering excellent snow leopard habitat, the park's four species of wild sheep

and goats give it international biodiversity importance. About 1,600 people live in 16 small settlements scattered in three valleys. They grow barley and a few vegetables, and own more than 4,000 head of livestock, of which 81% are sheep and goats, and 11% are yaks, cattle and crossbreeds. Tourism provides an important source of supplementary income. Ladakh was opened to tourism in 1974, and the Markha Valley circuit through Hemis National Park remains the most popular trekking route, with about 5,000 visitors per year.

We surveyed 79 households living within or immediately adjacent to the park to determine livestock ownership patterns, document depredation losses and map the depredation-prone areas or "hotspots" (Bhatnagar et al. 1999). Over half the households interviewed lost one to 15% or more of their domestic stock to predators, or 492 animals valued at USD $23,500 over a 14-month period. Snow leopard and wolf were associated with 55% and 31% of the presumed depredation incidents respectively, with sheep and goats constituting 75% of the stock lost, followed by yak-cattle (13%) and horses (8%). Three settlements incurred 54% of the depredation. Losses incurred from snow leopards entering poorly constructed corrals accounted for 14% of all incidents (N=210), but nearly 50% of all livestock lost—understandably arousing considerable anger among the livestock owners.

Along with poorly constructed livestock pens, investigations into the root causes of depredation implicated lax daytime livestock guarding practices. Stock was allowed to forage in areas with well-broken terrain and cliffs, prime habitat for snow leopard (Jackson et al. 1996). The fact that domestic livestock now substantially outnumbers natural prey and biomass only invites loss to wild predators. Historically there has been better emphasis on daytime guarding, and problem predators were controlled through trapping and other traditional control methods. With more children going to school and youths reticent to assume this hard livelihood, even highly vulnerable small-bodied livestock are left to graze unattended. While baseline documentation is lacking, predator numbers appear to have increased due to park regulations prohibiting hunting and patrolling by park guards. As Figure 1 suggests, depredation rates vary with locality, presumably reflecting differences in predator densities, habitat suitability and herding patterns.

The household survey was followed by a workshop held in Markha village in association with the Ladakh Wildlife Department, national and international non-governmental agencies (NGOs). The primary objectives were to (1) identify cost-effective and ecologically compatible measures for reducing livestock losses; (2) Train park staff, NGOs and villagers in wildlife damage alleviation techniques; and (3) Promote community-based wildlife stewardship and enhance awareness of the opportunities to conserve even "problem" species.

Using a highly participatory

Snow leopard (*Uncia uncia*). Photograph by M. Elsbeth McPhee

process known as Appreciative Participatory Planning and Action (APPA), workshop participants and villagers examined root causes of depredation and identified a series of measures aimed at reducing depredation loss, improving household incomes and promoting wildlife conservation. APPA combines concepts from Appreciative Inquiry (used in business leadership training) and Participatory Learning and Action (PLA, Pretty et al. 1995), in a collective inquiry and planning process aimed at fostering effective group action. It operates under two complimentary premises: (1) What you seek is what you find—"if you look for problems, then you will find more problems" or conversely, "if you look for successes, you will find more successes;" and (2) What you believe is what matters most—"if you have faith in your vision or ideas for the future, and if these are believable, you can achieve success without waiting for government or outside agents to take you there."

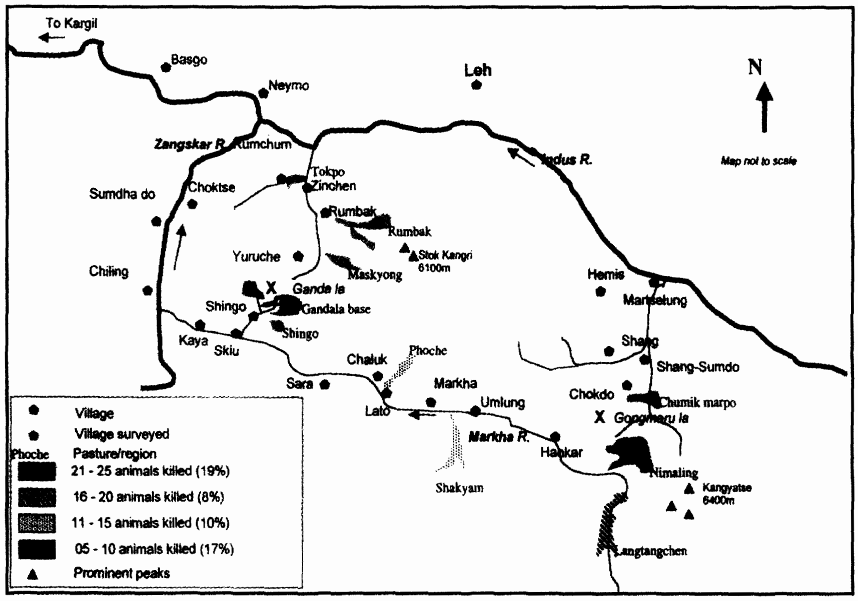

Figure 1. Depredation hotspots

APPA is practiced through an iterative process that seeks to (1) *discover* the community's strengths and its valued resources; (2) *envision* short-term and long-term futures if resources were mobilized and the community acted in concert; (3) *design* a basic action plan for guiding development and nature protection in ways that substantially limit long-term dependency upon outside financial sources or technical "know-how;" and (4) motivate participants to initiate community-improvement actions *immediately*, and largely on their own.

Outside donor support was only offered if the following provisions were met:

1) Conservation—Biodiversity conservation is the primary motivation behind external investment, and therefore all project activities must be implicitly linked with clearly defined conservation objectives.

2) Participation—the active and equitable involvement of each stakeholder group is promoted throughout the project to ensure all affected households will be benefitted and to encourage participation irrespective of gender, age or economic status.

3) Reciprocity—All stakeholders, whether outside donor, local NGO, government, or villagers are expected to make a reciprocal contribution within their means (e.g., cash, materials, labor, or in-kind service).

4) Responsibility—The beneficiary community must be willing to assume responsibility for meeting the conservation objectives and for maintaining any infrastructural development. There should be clear penalties for infringement by any of the participants.

5) Monitoring—Stakeholders should employ simple but realistic indicators for monitoring project impact and performance, described in the Action Plan prepared jointly and signed by the key parties.

External expertise is blended with local knowledge in designing remedial actions that were environmentally responsible (i.e., compliant with park regulations and species/habitat management requirements); economically sustainable within the local context; socially responsible (e.g. building upon proven traditions and cultural values which protect nature); and which are implemented under a mutually agreed-to (and signed) work-plan that sets forth the responsibilities, contributions and obligations of each partner.

Markha villagers concluded that their best option lay in replacing the existing four winter corrals with three larger predator-proof structures placed side-by-side and sharing inner walls. They donated their labor and provided on-site materials (stones and mud), while external donors provided off-site materials (wire mesh, roofing poles, and secure doors). Construction was scheduled for spring, but was delayed due to frozen ground. Also, corrals had to be 15 feet longer than the plans indicated because the villagers deliberately underestimated their livestock holdings, fearing they would be taxed more by the government for reporting actual herd sizes. Unfortunately, they used the corral before it was fully predator-proofed, and lost 29 animals to a snow leopard. The outside donors felt some responsibility for the loss and called a community meeting. The villagers, however, assumed full responsibility for what had happened. Their reasoning was as follows: there had been a death in one of the families, just before the depredation. As the other six affected households had only lost one or two animals, they all agreed that a traditional Mountain Spirit had been responsible for the snow leopard's visit. We believe that by predator-proofing a village's corrals we are removing as many as five to 10 snow leopards from risk of retaliatory killing.

Conclusions

At the broader level, the future of these protected areas hinges on the degree to which the basic concerns, needs and aspirations of the local people are addressed. Over the long-term, the most cost-effective approach for cash-strapped developing countries may lie in promoting a set of carefully designed and monitored community-based stewardship initiatives in which local people benefit from the presence of wildlife, including predators. While our initial effort focused on reducing loss of livestock to predators, we are now concentrating on measures aimed at helping local people capture more benefits from tourism. For example, women are being offered skills training to enhance their summer tea-house operations by improving menus, ensuring hygienic conditions, and building campground facilities. A key next-step will be to use the "parachute cafes" or tea-houses as focal points for providing tourists and local communities with wildlife conservation education.

We believe this approach is highly effective in mobilizing rural communities toward greater self-reliance and thus a more harmonious long-term relationship with the National Park in which they live, and on whose resources they depend so heavily. APPA builds pride by highlighting positive community attributes and building upon traditional values and successes. NGOs are the most obvious vehicle for facilitating community-based integration of conservation and development; however, the sponsoring agency must be willing to make a long-term commitment to their rural stakeholders (Sanjayan et al. 1997).

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This work was initiated in the senior author's capacity as Conservation Director of the International Snow Leopard Trust, now being continued under the Snow Leopard Conservancy. Special thanks to the J&K Wildlife Department and LEDeG for co-sponsoring the workshop in Markha. The senior author is indebted to Rinchen Wangchuk and the residents of Hemis National Park for their commitment to finding long-term solutions to this vexing problem.

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Canid Conservation

The Status of the Wolf Population in Post-Soviet Kyrgyzstan

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Abstract

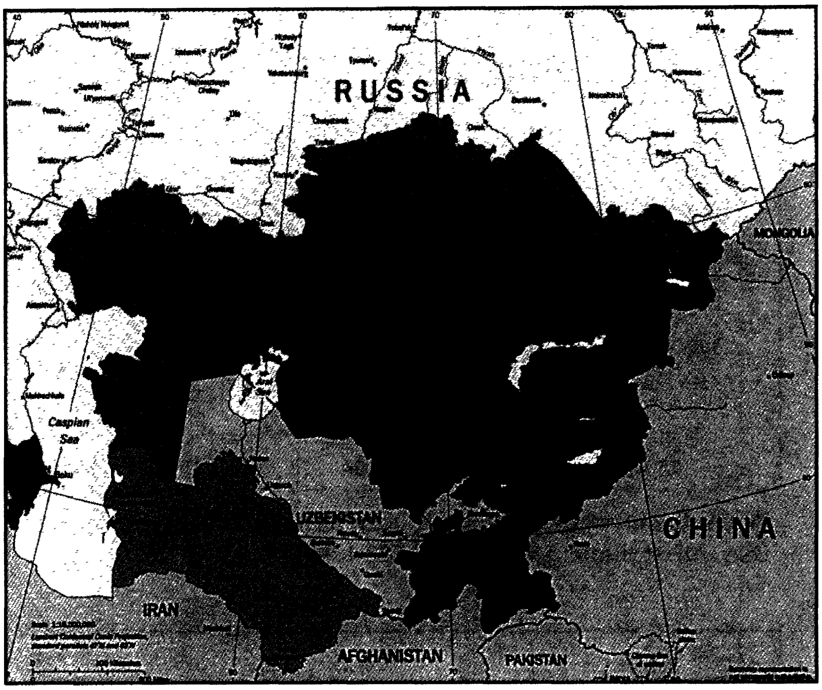
*The countries that made up the former Soviet Union are home to the largest existing wild gray wolf (*Canis lupus*) population. This makes the status of wolves in these areas of particular importance. Kyrgyzstan is a former Soviet republic located in Central Asia that became independent in 1991. Russia and other former Soviet republics with wolves reported in 1999 that their wolf populations were either stable or increasing. Kyrgyz officials also consider their wolf populations to be increasing since independence. At the same time, officials report that populations of deer, wild boar, and other wildlife are decreasing drastically. Between January and May 1999, I made seven field trips in southern Kyrgyzstan to look for evidence to support official reports. I visited two zapovedniks (strictly protected areas), one national park and two national forests (less protected), and two rural areas with no wildlife protection to look for field signs and to collect anecdotal information from local villagers. I found abundant wolf sign at only two sites and abundant deer sign at only one site. Villagers tended to say that there were many wolves, but they were 20 km away. They also said that deer, boar, and other wildlife were difficult to find now. This evidence supported official reports that wildlife populations were decreasing but contradicted reports that wolf populations were increasing. Government records show that most species of game animals and sheep, the primary livestock animal, have decreased by close to 50% in the south since independence. Data also showed that wolf populations in the south have dropped by 43%. This evidence suggests that official reports that wolf populations in Kyrgyzstan are stable or increasing are inaccurate and wolf populations may actually be declining.*

Introduction

Gray wolves (*Canis lupus*) used to roam over most of the Northern Hemisphere. Human persecution over the last few centuries has resulted in very small, isolated populations or extirpation in much of North America and Western Europe. The area encompassed by the former Soviet Union is the home of the largest remaining populations of gray wolves (Theberge 1991). The Soviets have always considered wolves to be a pest species and have traditionally supported strong, government-sponsored wolf control programs. Historically, during economic downturns, wolf populations have increased due to less money being available for wolf control (Bibikov 1993).

In 1999, Wolf International, the magazine of the International Wolf Center, reported on the status of wolf populations worldwide, including some areas of the former Soviet Union. Of the Commonwealth of Independent States (CIS) countries reporting, most list their populations as increasing or stable with wolves being unprotected or considered game species (Route and Aylsworth 1999). Since the break up of the Soviet Union from 1989 to 1991, most of that region has been experiencing severe economic decline (Anderson and Pomfret 2000). Given the changes currently taking place, both political and economic, an assessment of the impact these changes are having on wolf populations is important.

The Kyrgyz Republic is a former Soviet republic in Central Asia that became independent in 1991. It is a small country of just under 200,000 km^2 with a population of about 4.6 million people. Over 90% of the terrain is mountainous with elevations ranging up to 7,439 meters. Almost 40% of the land area is considered to be uninhabitable due to elevation, glaciers, or the presence of bare rock. Less than 4% of the land area is currently forested. It has an arid, alpine climate and much of its economy is agriculturally based with livestock grazing, particularly



Commonwealth of Independent States (CIS): Central Asian States

sheep, a primary activity (Ministry of Environmental Protection 1998).

Evaluating the current status of wolf populations

Information on the condition of wildlife in Kyrgyzstan that was collected as part of a biodiversity assessment completed in 1999 indicated that, in general, wildlife populations were declining (Ministry of Environmental Protection 1998). In contrast, the general perception on the part of the public and government officials was that wolf populations were actually increasing. People in villages were concerned about livestock losses to wolf depredation. The perception that wolves were an increasing threat to the economic well being of villagers contributed to the government policy of paying a substantial bounty for killing wolves. The bounty in 1998 was 1000 som ($20) for an adult male and 1500 som ($30) for an adult female (Kurmanaliev, personal communication). This is a substantial amount of money in a country where over half the population makes less than 4,500 som a year (World Bank 1999).

From January to May 1999, I conducted a study to determine if the perception that wolves were increasing in abundance in spite of decreasing wildlife populations could be substantiated. I conducted seven field surveys in the southern part of the country by going out with local guides to look for wolf signs as well as signs of potential wild prey species. These surveys were not meant to be exhaustive or systematic. I asked the guides to show me the most likely places that wolf sign would be found. I hoped to survey the areas that were more likely to have higher wolf populations than elsewhere near the village. This way, the results were more likely to indicate an abundance of wolves and result in a conservative measure of possible declines. The surveys included two zapovedniks (protected areas with no recreational use or hunting allowed), two national forests and one national park (protected, but some uses allowed), and two rural areas with no protected status. These areas also represented a variety of ecosystem types including juniper forests, walnut forests, and alpine meadows at elevations ranging from 2,000 to 3,000 meters. The major commonality for these areas was that all of them were historically rich in wildlife. I also collected anecdotal information from local people on wolves and the abundance of deer and other wildlife. In particular, I asked about changes they had noticed since Kyrgyzstan became independent. In addition, I looked at government records from the Hunting Institution and the Agricultural Ministry of Osh Oblast (province) in southern Kyrgyzstan to determine the official estimates of both wildlife and livestock populations and the changes from pre- to post-independence.

Results

Out of the seven locations surveyed, there was only one that had abundant signs of both wolves and potential wild prey species. This was in one of the rural, unprotected areas. Wolf tracks were abundant both on the mountain ridges and down into the river valley near the villages. Roe deer (*Capreolus capreolus*) were also abundant in the area, though there were no signs of wild boar (*Sus scrofa*), which used to be common in the area. We did locate one deer kill site about one km from a village and also a marking post with scat on a mountain ridge 12 km from the village. The scat consisted entirely of wool, indicating that these wolves had been eating sheep.

The other unprotected rural location also had abundant signs of wolves coming down from the mountain ridges into the river valley near the villages. The only wildlife species that appeared abundant in the river valley, however, was red marmot (*Marmota caudata*). There were no signs of deer or other larger mammal species. The wildlife signs in the other sites surveyed ranged from a few to no wolf signs with few signs of other large mammal species.

One of the zapovedniks, Sara Chilik, had some signs of wolves, deer, and wild boar, but the official wildlife counts indicated that the numbers of all were lower than they had been a decade earlier, prior to independence. In Besh Aral Zapovednik, there were few signs of wolves in spite of high official estimates. There were no deer or boar signs and marmots were the only species that appeared abundant. There were no long-term population surveys available for Besh Aral so a comparison of abundance from before to after independence was not possible. Anecdotal evidence indicated that in most areas, most wildlife species were declining in numbers. Villagers were having to range further away when hunting to find game. Even wild boar, which had not been hunted much in the past due to the Muslim culture of the Kyrgyz, seemed to be in decline. In most of the villages near the field survey sites, the locals felt that wolves were still abundant, but that most of the wolves were further away from the villages than they had been before. Most cases of wolf depredation on livestock that were related to me were from "the next village" rather than from the village I was in.

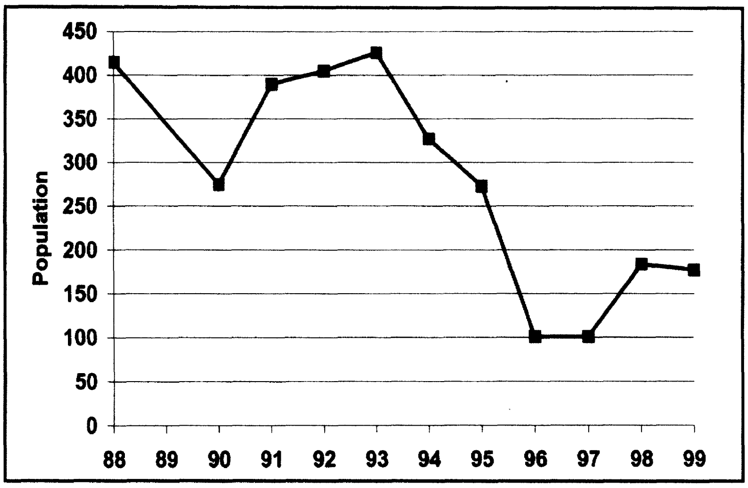


Figure 1. Survey of roe deer populations in Osh Oblast (data from Osh Oblast Hunting Institution 1988-1999).

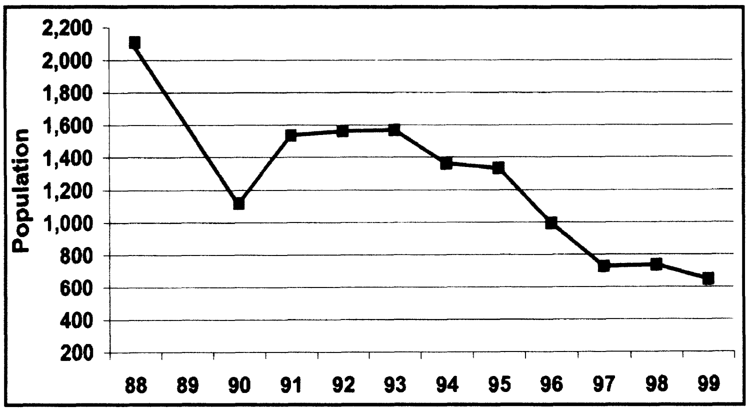


Figure 2. Survey of wild boar populations in Osh Oblast (data from Osh Oblast Hunting Institution 1988-1999).

Official estimates

The Government Hunting Institutions of Kyrgyzstan are responsible for keeping yearly estimates of all games species and other species of interest throughout the country. While the numbers from the official surveys may not be accurate as to the actual population size for the animals surveyed, they are collected using the same technique from year to year. Therefore, while not relying on the numbers to represent true population sizes, I felt that they were still useful to indicate trends. I compared the results of government estimates of the populations of wolves, roe deer, and wild boar in Osh Oblast from 1988 to 1999 (Osh Oblast Hunting Institution 1988-1999). These estimates showed a substantial decrease in the number of deer and boar over this time period. Deer populations declined by 39.4% while boar populations declined by 55.7% from a few years before the breakup of the Soviet Union to the present with the declines starting around 1992, just after independence (Figures 1 and 2). These numbers coincided with the results of the field surveys and with the anecdotal information from both official and local sources.

Since livestock plays such an important role in the Kyrgyz economy and can serve as an alternative prey base for wolves, I also examined the agricultural census data for Osh Oblast from 1987 to 1997 (Osh Oblast Subsistence Farm Administration 1987-1997). This was to determine if there had been changes in the total number of livestock that might be available to wolves. According to the official count, the numbers of horses and cattle have experienced only small declines over the 10-year period. However, sheep, which is the dominant domestic species in the country, have steadily declined since independence. There were almost two million sheep in Osh Oblast before the breakup of the Soviet Union and by 1997 the number had declined to less than 900,000—a drop of 55.6%.

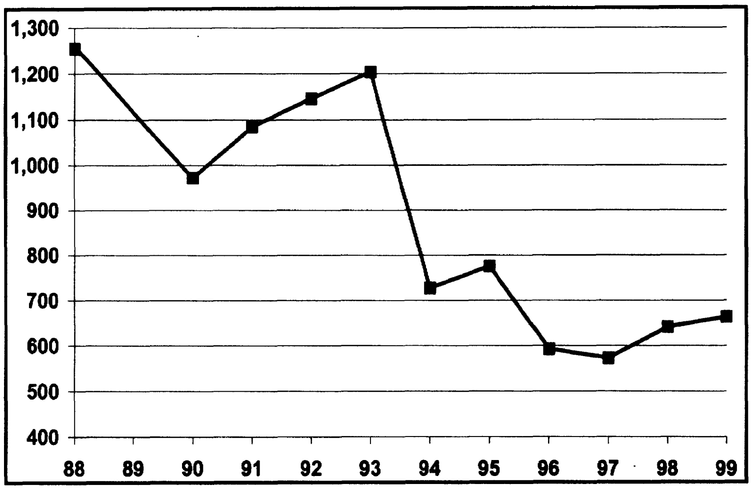


Figure 3. Survey of gray wolf populations in Osh Oblast (data from Osh Oblast Hunting Institution 1988-1999).

In addition to showing declining populations for wild prey species, the Government Hunting Institution survey data also showed a marked decline in wolf abundance from approximately 1100 wolves in Osh Oblast to just over 600 —a decline of 44.3% over the same time period (Figure 3) (Osh Oblast Hunting Institution 1988-1999). This would seem to indicate that even the survey information collected by the government does not support the official stance that wolf populations have increased substantially since independence. The field survey results, as well as much of the anecdotal information, also provide evidence that wolf populations have actually declined over the last decade.

Discussion

There are two questions that need to be addressed. One is why, during the current economic hard times, are wolf populations declining rather than increasing, as they have in the past during Soviet times? The other is why the current perception is that wolf populations are increasing when populations may actually be declining? In order to answer these questions, we must understand how the economy and lifestyle of the people have changed with the breakup of the Soviet Union and the coming of independence.

Since the breakup of the Soviet Union, Kyrgyzstan, like most of the former republics, has been experiencing a severe economic decline. Overall the country has experienced declines in productivity and GDP, resulting in a sharp increase in the unemployment rate (World Bank 1999). During the early years of independence, there was a serious decline in income for most people (Anderson and Pomfret 2000). Pensions are no longer sufficient for the elderly to live on, and even people who did have jobs frequently were not paid for months at a time (personal observation). This decline in income and employment coincided with the dismantling of much of the social safety net that was in place during the Soviet times. Medical care and education are no longer free and subsidies for food and utilities have either been eliminated or sharply reduced (Anderson and Pomfret 2000). This has resulted in close to 80% of the population of Kyrgyzstan living in poverty (Ministry of Environmental Protection 1998).

The increase in poverty and unemployment and the changes in government subsidies for social programs are only part of the story. In addition, there have been substantial changes in the structure of the agricultural sector of the economy. This is significant because the economy is largely agriculturally-based (World Bank, 1999). The large state-owned collective farms of the Soviet Union have almost all been broken up and the assets distributed as part of the privatization process (Karimbiekov, personal communication). As a result, most livestock is now privately held in small family farms with private herds being relatively small.

The increase in poverty has also played a role in the overall decrease of wildlife populations. More people are exploiting natural resources for their daily subsistence. Poaching has increased significantly. Habitat destruction is also an increasing problem with the need to use natural resources for both food and fuel due to a lack of alternatives that were previously available. In addition, the changes in the structure of livestock herds has brought about a change in grazing patterns, resulting in increased competition for resources between wild and domestic species (Ministry of Environmental Protection 1998). The reason that this economic decline hasn't resulted in an increase in wolf populations is that the nature of the economy itself has changed. During Soviet times, the major threat to wolves was the large-scale government sponsored wolf control programs. When the economy experienced a recession, there was less money to spend on these programs and wolf populations were able to rebound (Bibikov 1993). Currently, the major threat is a lack of natural resources. The prey base for wolves has declined significantly in less than 10 years due to habitat destruction, increased human exploitation of game species, and a decline in domestic livestock. In addition, while there are currently no government employees being paid to hunt wolves, there are still bounties in place that encourage private hunting of wolves.

Over 90% of the countryside of Kyrgystan is mountainous (photograph by CJ Hazell).

The reason that people perceive wolf populations as increasing rather than decreasing may be due to the changes in both the economy and in the lifestyle of people. Economically, before independence, individuals didn't experience significant changes in living conditions during a recession because of the regulated, centralized economy that existed during the Soviet times. The social safety net that existed guaranteed access to employment and basic amenities. In addition, most livestock was gathered into large state-owned herds. Wolf depredation resulted in the taking of a few head of sheep from very large flocks so proportionately the losses were not as significant.

Since independence, the state-owned herds have been broken up and distributed to individuals, resulting in many small, privately owned herds. In addition, the economic downturn has resulted in a significant increase in poverty and a drop in the standard of living for most people. When sheep were taken from a state-owned herd, it didn't result in individual farm workers experiencing economic hardship. This is no longer true. The increased economic hardship for farmers resulting from wolf depredation is coupled with a decline in wild prey species. The wolves have fewer alternatives to livestock. This may be resulting in wolves coming closer to villages as well as a possible increase in depredation. The latter is difficult to substantiate due to the lack of documentation of depredation rates. These conditions could easily result in the perception that wolves are increasing in number simply because their visibility and economic impact may be increasing.

Conclusion

The former Soviet Union comprised a very large geographic area. Since its breakup, the former republics are all experiencing profound economic and social upheaval. These changes are not only affecting people, but are also having a significant impact on wildlife. Most of the resulting declines in wildlife populations are being acknowledged. Wolves, however, have not been recognized as a species that may also be experiencing negative impacts from the current situation. The perception that wolf populations are stable or increasing appears to be common in these regions. This perception in Kyrgyzstan, coupled with the increased economic impact of wolf depredation due to the increase in poverty and private ownership of livestock, appears to be exerting increasing pressure on the government to re-institute wolf control programs (Kurmanaliev, personal communication). This could be devastating if wolf populations are already in decline.

The Kyrgyz Republic is only one small part of the former Soviet Union, but most former republics are experiencing similar social and economic problems. We need to more fully investigate how these changes are affecting wildlife communities. Most of this region has reported that wolf populations are stable. We should be skeptical of this.

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Gray Wolf Restoration in the Northwestern United States

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Abstract

Gray wolf (Canis lupus) *populations were eliminated from Montana, Idaho, and Wyoming, as well as adjacent southwestern Canada by the 1930s. After human-caused mortality of wolves in southwestern Canada began to be regulated in the 1960s, populations began expanding southward. Dispersing individuals occasionally reached the northern Rocky Mountains of the United States, but lacked legal protection there until 1974, after passage of the Endangered Species Act of 1973. In 1986, wolves from Canada successfully raised a litter of pups in Glacier National Park, Montana, and a small population was soon established. In 1995 and 1996, wolves from*

western Canada were reintroduced to remote public lands in central Idaho and Yellowstone National Park. These wolves were designated as nonessential experimental populations to increase management flexibility and address local and state concerns. Wolf restoration is rapidly occurring in Montana, Idaho, and Wyoming, and there were at least 28 breeding pairs in December 2000. There are now about 63 adult wolves in northwestern Montana, 192 in central Idaho, and 177 in the Greater Yellowstone area. Dispersal of wolves between Canada, Montana, Idaho, and Wyoming has been documented. Occasional lone wolves may disperse into adjacent states, but population establishment outside of Montana, Idaho, and Wyoming is probably not imminent. The gray wolf population in the northwestern U.S. should be recovered and, depending on the completion of state and tribal wolf conservation plans, could be proposed to be removed from Act protection within three years. Wolf restoration has proceeded more quickly and with more benefits, such as public viewing than predicted. Problems, including confirmed livestock depredations, have been lower than estimated. The Service led interagency recovery program focuses its efforts on achieving wolf recovery while addressing the concerns of people who live near wolves. Wolves have restored an important ecological process to several large wild areas in the northern Rocky Mountains of the U.S. The program has been widely publicized and is generally viewed as highly successful.

Wolves in northwestern Montana

Sixty years after being nearly exterminated from the lower 48 states, the gray wolf *(Canis lupus)* was listed under the Endangered Species Act (ESA) in 1974 and was eventually restored to Montana, Idaho, and Wyoming. Wolves were once common throughout North America but were deliberately exterminated in the lower 48 states (except northeastern Minnesota). Wolves remained abundant in much of Canada and Alaska. Recovery began in northwestern (NW) Montana in the late 1970s by natural dispersal from nearby expanding Canadian wolf populations (Pletscher et al. 1997). Wolves first denned in NW Montana in Glacier National Park in 1986 (Ream et al. 1989). Wolf numbers steadily increased until 1996, when there were a minimum of 70 wolves in seven different packs that lived solely in NW Montana. An unusually severe winter in 1996-97 caused a 30 to 50% decline in the white-tailed deer (*Odocoileus virginanus*) populations, the primary prey of those wolves. The number of wolves dropped to just over 50 in five packs in 1997, largely as a result of agency wolf control actions in response to high livestock depredations and subsequent poor pup production (Bangs et al. 1998). Wolf numbers have only slightly increased since 1997. In 2000, there were an estimated 63 wolves in about a dozen groups, but only six of those successfully reproduced. Most wolves in NW Montana live in a mix of private and public land west of the Continental Divide.

Wolf reintroduction in Yellowstone National Park and central Idaho

In 1988 and 1990, Congress directed the National Park Service to prepare a series of reports on the potential effects of reintroducing wolves to Yellowstone National Park (YNP 1990). Wolf depredation on livestock, wolf predation on wildlife, land-use restrictions, tourism, other predators including grizzly bears (*Ursus arctos*), diseases, and a wide variety of other issues were evaluated. In 1990, Congress established a Wolf Management Committee, consisting of federal, state, and private special interest groups to try to forge a political compromise on the issue of wolf reintroduction in both Yellowstone and central Idaho. Their report was completed in May 1991, but Congress chose not to act on the Committee's recommendation, which included wolf reintroduction and more flexible wolf management than was normally allowed under the ESA. All these reports, and all subsequent investigations, made it clear that reintroducing wolves in Yellowstone National Park and central Idaho was feasible and would ultimately result in wolves attempting to recolonize areas throughout Montana, Idaho, and Wyoming and far outside the reintroduction areas.

In late 1991, Congress directed the Fish and Wildlife Service (FWS) to lead preparation of an environmental impact statement (EIS) to examine the effect of reintroducing wolves to Yellowstone National Park and central Idaho (FWS 1994). The planning and public involvement effort took two years to complete. By the time it was finished the Service had distributed over 750,000 documents, conducted over 130 public meetings and hearings, and reviewed 170,000 public comments. The decision was to reintroduce wolves to both Yellowstone and central Idaho as nonessential experimental populations, the most flexible classification for species listed under the ESA. The decision was approved in spring 1994 by both the Secretary of the Interior (FWS, National Park Service and Bureau of Land Management) and the Secretary of Agriculture (Wildlife Services and Forest Service).

The EIS predicted that a recovered wolf population (a minimum of 10 breeding pairs, estimated to be about 100 adult-sized wolves) in the Yellowstone area would kill an average of 19 cattle (*Bos* sp.), 68 sheep (*Ovis aries*), and up to 1,200 ungulates (primarily elk) annually. This would not affect hunter harvest of male ungulates, but could reduce hunter harvest of female elk (*Cervus elaphus*), deer (*Odocoileus* sp.), and moose (*Alces alces*) in some herds. Hunter harvests or populations of bighorn sheep (*Ovis Canadensis*), mountain goats (*Oreamnos americanus*), or antelope (*Antilopra americana*) would not be affected. Bison (*Bison bison*) would not be preferred prey. Wolf predation may reduce populations of elk five to 30%, deer three to 19%, moose seven percent, and bison up to 15%. The presence of wolves would not change uses of public or private land except for potential use of M-44 cyanide devices, used to control coyote (*Canis latrans*) damage, in areas occupied by wolves. Visitor use was predicted to increase five to 10%. At wolf recovery, annual economic losses were estimated to be $187,000 to $465,000 in hunter benefits (what hunters said hunting female elk was worth to them), $207,000 to $414,000 in potential reduced hunter expenditures (what hunters of female elk said they would have spent hunting), and $1,888 to $30,470 in potential livestock losses. Annual increased visitor expenditures were estimated at $23 million and the existence value of wolves was estimated at $8.3 million (what people believed having wolves in the Yellowstone area was worth to them). Similar predictions were made for the central Idaho area. Depending upon their distribution, more than 100 adult-sized wolves would proportionally increase impacts above those predicted in the EIS. To date, at least the trends in these predictions appear to have been fairly accurate. It will take time before wolf numbers and distribution stabilize and the true effect of having wolves back in these areas can be ascertained.

The restoration of wolves to public lands in the western United States, particularly Yellowstone National Park, was proposed as early as the 1940s. After years of direct involvement by Congress and exhaustive public involvement and planning, 35 wolves were reintroduced via hard (immediate) release to wilderness areas in central Idaho, and 31 were soft released in Yellowstone National Park, Wyoming in January 1995 and January 1996 (Fritts et al. 1997; Bangs et al. 1998). Those wolves, originally from Canada, were designated as nonessential experimental populations to increase management flexibility over what is normally allowable for species listed under the ESA. Examples of this flexibility are: landowners could harass wolves at any time; livestock producers could shoot wolves seen attacking livestock; wolves could be relocated if they significantly impacted wild ungulate herds (as defined in approved state wolf management plans); there would be virtually no land-use restrictions; the Service could use special permits to take wolves for various management reasons; and funding was offered for state and tribal leadership in wolf recovery actions (Bangs and Fritts 1996). Currently wolves in Wyoming and Montana are primarily managed by the FWS, National Park Service (in Parks), and USDA Wildlife Services. In Idaho, wolves are primarily managed by the Nez Perce Tribe and Wildlife Services, under a cooperative agreement with the FWS.

Reintroduced wolves adapted better than predicted and only two years of reintroduction were required rather than the three to five years that were predicted (Fritts et al. 1997). In December 2000, the population estimate was 177 wolves in 13 breeding groups in the Yellowstone area and 192 wolves in 9 breeding groups in Idaho. To date, wolves have settled primarily on remote public lands, but that will change as the population expands and more wolves disperse beyond where wolf packs currently exist. Dispersing wolves will increasingly try to occupy private lands used for livestock production; this will increase the rate of livestock depredations and agency control. Except for a few temporary closures to protect wolf viewing opportunities around active dens in Yellowstone National Park, and restricting some M-44 use, the wolf restoration program has caused no land-use restrictions that might disrupt traditional human activities such as logging, mining, livestock grazing, hunting, trapping, or wildland recreation. Over 70,000 visitors to Yellowstone National Park have seen wolves and public interest in them is extremely high.

Wolf research

Between 1979 and the late 1990s, extensive research on wolves in NW Montana was supported by a host of state and federal resource management agencies. Field work and data analysis were carried out largely by graduate students and the University of Montana. Those studies investigated the relationships between wolves and other wildlife, including white-tailed deer, elk, and moose, other predators such as mountain lions (*Felis concolor*) and coyotes, and livestock (Kunkel and Pletscher 1999; Kunkel et al. 1999; Kunkel

1997). This research indicated wolves were just another predator on wild ungulates, neither much less nor much more effective than other native predators, such as mountain lions, black (*Ursus americanus*) and grizzly bears, or coyotes. Wild predators, including wolves, typically killed more of the most vulnerable of ungulates (injured, sick, or very young and very old individuals) than did human hunters. Wolf predation in combination with other factors such as winter weather, human hunting, other predators, and habitat conditions, contributed to a decline in white-tailed deer and elk in the North Fork of the Flathead River. Moose populations apparently were not as affected by these same circumstances. As a result of that prey decline, wolf numbers in that area dramatically declined, from nearly 30 wolves in three packs during the most intensive research in the early 1990s to a few individuals that did not produce pups in 1999 or 2000. Wolves often trailed mountain lions to take over their kills and killed a few lions. Direct competition for the types of ungulates that are most vulnerable to predation was likely the main impact that wolves would have on lions. Wolves also killed a few coyotes. While wolves displaced lions and coyotes from ungulate carcasses, wolf kills were often usurped by grizzly bears. Studies of wolf genetics and dispersal indicated that genetic diversity was high and likely not a management concern, as long as opportunity for occasional dispersal from wolf populations in Canada and other U.S. recovery areas in Idaho and Wyoming was maintained (Boyd and Pletscher 1999).

Research indicated that although wolves often lived near livestock (primarily cattle) and other domestic animals, conflicts were uncommon. Dogs, almost exclusively hunting hounds and livestock guard and herding dogs, were apparently killed as competitors rather than prey. Wolves commonly fed on carrion of both livestock (carcass dumps) and wild ungulates (road and train kills, unretrieved hunter-killed game, and gut piles). In some instances, abundance of natural prey and relative vulnerability of livestock affected how often wolves attempted to attack livestock. Sick or wounded livestock or small livestock, such as calves or sheep, appeared particularly vulnerable to wolf predation. But often, wolves appeared to attack livestock without any predisposing factors and nearly all wolf packs with regular exposure to livestock sporadically caused depredations.

A large number of studies and research are currently being conducted on wolves in the Yellowstone and central Idaho experimental areas so that accurate information can be used to better manage wolf populations and expand the level of knowledge about wolves. Wolf predation studies indicated elk were more than 90% of the prey killed by wolves in Yellowstone. Kill rates were about 15 elk per wolf per year. In Idaho, wolves also preyed mainly on elk, but wolves there killed a higher proportion of mule deer (*Odocoileus hemionus*). Wolf kills were more likely to be in open habitats and the remains scattered, compared to mountain lion kills that were often covered and hidden in thick cover. This gave a visual impression that wolves killed more deer and elk than mountain lions, but a lion actually kills more ungulates per year than does a wolf. Annual wolf kill rates typically average about 20 adult deer or 12 adult female elk per year, while adult lion kill rates can be twice as high. Both wolves and lions tended to prey on the most vulnerable wild ungulates such as calves and very old females. Calf elk killed by lions in Idaho were in better condition than calf elk killed by wolves. Bison are difficult to kill and few wolves have learned to do so effectively (Smith et al. 2000). Somewhat surprisingly, to date no Bighorn Sheep have been confirmed killed by wolves in either area.

Carcasses of elk killed by wolves were utilized by a wide variety of other wildlife species and provided a year-long food source that would likely increase overall wildlife diversity. Coyote numbers in some areas may have been reduced by half because of wolves killing coyotes. Mountain lions and wolves tend to kill the same types of prey, but lions are usually confined to more rugged steep and vegetated terrain, while wolves preferred flatter terrain and made more use of open habitat. Grizzly bears often usurped wolf killed ungulates. Studies are investigating the effect of wolves on elk distribution on winter feeding grounds in Wyoming, but tentative results suggest little effect other than elk appear to be more wary and may prefer larger groups and more open habitat when wolves are present. Earlier studies in Montana indicated that wolves did not change ungulate distribution on natural winter ranges, but apparently caused ungulates to be more wary and to temporarily retreat to thicker cover when wolves were present.

A recent study funded and initiated by the Nez Perce Tribe and a host of federal agencies and local livestock producers found that confirmed livestock losses may be a fraction of actual losses under some circumstances. That study determined the cause of death and detection rate of 220 radio-tagged livestock calves of about 700 on large, very remote, and heavily forested USDA Forest Service grazing allotments. After two years, pneumonia killed the most marked calves, but wolf predation was the second leading cause of death. Sample sizes were very small, but as many as 5.7 calves may have died from wolf predation for every one discovered by normal livestock herding practices. Wolves killed calves that were the lowest weight, least guarded by people, nearest to an active wolf den, and in the heaviest forest cover, suggesting that wolves tested and hunted cattle like wild prey and attacked the most vulnerable animals.

Livestock depredations

Since 1987, annual confirmed minimum livestock losses in NW Montana totaled 82 cattle, 68 sheep and seven dogs. As a result, 41 wolves were killed and 32 were moved. Depredations averaged 5.8 cattle, 4.8 sheep, and less than one dog annually. Agency control killed an average of three wolves per year. On average, less than six percent of the wolf population is annually affected by agency wolf control actions (Bangs et al. 1995). Minimum confirmed livestock losses have annually averaged about 3.6 cattle, 27.8 sheep, and 3.8 dogs in the Yellowstone area, and 9.2 cattle, 29.4 sheep, and 1.8 dogs in central Idaho. In addition, one newborn horse (*Equus* sp.) was killed in the Yellowstone area. In total there have been 146 cattle, 356 sheep and 35 dogs confirmed killed by wolves from 1987 until January 2001. Since 1987, the Service and USDA Wildlife Services have killed 41 wolves in NW Montana, 18 in central Idaho, and 26 in the Yellowstone area because of conflicts with livestock. The rate of confirmed wolf-caused livestock losses and the number of wolves that have been removed in agency control actions is one-third to one-half of the levels predicted in the EIS. Despite lower than expected losses and less wolf control than predicted, wolf depredations and control remain inordinately controversial. Even the most routine wolf depredation and control actions still result in major local news coverage. To the general public, this probably greatly exaggerates both the role of wolves as livestock predators and the level of agency control. Since 1987, livestock producers who experienced confirmed or highly probable wolf-caused losses in Montana, Idaho, and Wyoming have been compensated about $155,000 by a private compensation fund administered by the Defenders of Wildlife, who support wolf recovery and management efforts.

Minimizing livestock conflicts

The Service is evaluating a wide variety of alternative methods to prevent or reduce conflicts with livestock in addition to relocating or killing problem wolves. The experimental population rules and the recently proposed special rule for wolves listed as threatened would allow for harassment and killing of problem wolves. In cooperation with USDA Wildlife Services and private conservation organizations we have: used light and siren devices, including models triggered by the signals from individual radio-collared wolves; established barriers to wolves using guard animals, flagging and fencing; provided extra surveillance of livestock with herders or agency personnel; harassed and moved and/or provided supplemental food to wolves that established dens and rendezvous sites in livestock grazing pastures; initiated research using electronic dog training collars to teach wolves not to attack livestock; provided livestock producers radio telemetry receivers so they could closely monitor wolves near their livestock; and helped provide alternative pasture to reduce livestock and wolf encounters. We have permitted livestock producers to shoot wolves actually seen attacking livestock, and in a few chronic cases of depredation on private property, to shoot wolves on-sight. We have allowed landowners to non-injuriously harass wolves at any time. We have trained and then issued cracker shells and less-than-lethal munitions (12-gauge bean-bag or rubber bullet shells) to private landowners so they could injuriously harass any wolves near their livestock or property.

Litigation

Several lawsuits were filed over the reintroduction program, by a wide variety of groups, including the Sierra Club Legal Defense Fund who supported and the American Farm Bureau Federation who opposed wolf restoration. The lawsuits were pooled into a single case that questioned whether the Service's use of an experimental population designation for reintroduced wolves illegally reduced protection of wolves that might naturally wander into the experimental areas. To date, no naturally dispersing wolves have been found in the Yellowstone area, but at least three wolves from NW Montana have dispersed into the central Idaho area. The Wyoming District Court eventually ruled against the Service's position in December 1998 and ordered all the reintroduced wolves removed, but stayed its own decision pending appeal. That case was then reviewed by the Tenth Circuit Court of Appeals in Denver, Colorado. Their ruling in January 2000 overturned the Wyoming lower court ruling. The Tenth Circuit endorsed and validated the legality of the Service's authority and the wolf reintroduction program. None of the losing parties appealed to the Supreme Court perhaps because several months earlier, in a closely related case that involved the illegal killing of an reintroduced wolf, the Ninth Circuit Court of Appeals in California had also ruled strongly in favor of the Service's authority and the Supreme Court had refused to hear an appeal. The only unresolved litigation involves a Wyoming rancher who suffered several confirmed wolf-caused livestock and pet depredations, and suspects he had many other unconfirmed losses. He claims that the government reintroduction program resulted in an uncompensated "taking" of his private property and lifestyle. Other litigation on a wide variety of wolf management issues is almost certain because of the strong symbolism of wolves to various special interest groups and the public, both at the local and national level.

The Service-led interagency wolf recovery program focuses its efforts on achieving the wolf recovery goal while addressing the concerns of people who live near wolves. Over 85% of all known wolf mortalities are caused by people, and the majority of those are a result of agency wolf control actions (Bangs et al. 1998). The key to successfully completing wolf restoration efforts will depend on maintaining some connectivity between the few remaining areas of large wild habitat remaining in the western U.S. and tolerance of wolves by the local rural residents (Fritts and Carbyn 1995).

Wolf recovery

Wolf populations should be fully recovered (30 breeding pairs with equitable distribution throughout the three recovery areas for three successive years) and will no longer need protection under the ESA by 2003. As a result of dispersal by wolves from Canada and the combination of reintroduction from two areas in Canada, genetics should not be a factor in wolf population viability as long as some connectivity is maintained (Boyd and Pletscher 1999). Once the recovery goal is achieved, wolves could be proposed to be delisted. After extensive public and professional review of the Service's delisting proposal, including assurance that state wolf management plans would conserve wolves above recovery levels, the affected states and tribes could manage wolves without federal oversight, except for the five-year post recovery monitoring period that is required by the ESA. State and tribal management programs will likely allow wolves to be killed in defense of life and property and in regulated public harvest programs, just as other large predators in these states are managed. Ultimately, wolf numbers (above minimum recovery levels) and wolf pack distribution will be determined by state and tribal wildlife management agencies. Once recovery goals have been achieved, delisting and a return to sole state and tribal management will signal the final success of the ESA at recovering the once imperiled gray wolf in the northern Rocky Mountains of the U.S.

Acknowledgments

Literally hundreds of people have contributed in a multitude of ways to wolf restoration. It is impossible for us to individually recognize and thank all of them in this paper, but without strong interagency cooperation and public involvement and tolerance, the successes so evident today would not have been possible.

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The Feasibility of Gray Wolf Reintroduction to the Grand Canyon Ecoregion

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Abstract
*As part of a regional conservation planning initiative, this study is being undertaken to determine the biophysical and socioeconomic feasibility of reestablishing a top carnivore, the gray wolf (*Canis lupus*), in the Grand Canyon Ecoregion (GCE). The GCE is a roughly 1.5 million km² area located on the southern Colorado Plateau. The last remaining gray wolves were probably eradicated in the 1920s and 1930s. Because of an interest in restoring extirpated native species to this ecoregion, and the desire to increase the size of the gray wolf metapopulation in the Southwest, there is need for an objective and spatially explicit landscape-scale model of potential gray wolf habitat. The first phase of this conservation GIS analysis involves utilizing six habitat characteristics or factors—vegetation cover, surface water availability, prey density, human population density, road density, and land ownership—to identify and describe potential reintroduction sites in the Arizona section of the Grand Canyon Ecoregion. Initial results show that there are at least two localities in northern Arizona suitable for reintroduction of around 100 wolves. This paper is a preliminary report on observations, results, and some recommendations deriving from the feasibility study.*

Introduction
Conservation biologists have shown that large or top carnivores are often keystone species whose removal jeopardizes the maintenance of ecological integrity in large-scale ecosystems (Soule and Noss 1998; Terborgh et al. 1999). Therefore, conservation planners interested in restoring and protecting large ecosystems or ecoregions emphasize recovery of top predators. The primary goal of this study—which is supported by the Grand Canyon Wildlands Council, Defenders of Wildlife, and Prescott College—is to determine the capability and suitability (together, the feasibility) of reintroducing one top carnivore, the gray wolf (*Canis lupus*), to the Grand Canyon Ecoregion (GCE) (Figure 1).

Ultimately, this study will address 26 factors or aspects, grouped into two dimensions—biophysical and human—that are expected to affect the feasibility of wolf recovery in the entire GCE (Sneed and Crumbo 1998). This paper, however, will focus on current, preliminary results from research done on a limited number of factors in the northern Arizona section of the ecoregion (see Figure 1).

Historic occurrence and taxonomic position
To accurately reconstruct the historic distribution of gray wolves in the GCE is challenging for a variety of reasons. Nineteenth century writers often accidentally or purposefully misidentified coyotes (*Canis latrans*), wolves, and wolf-dog hybrids (Gipson et al. 1998). Wolf hunters and trappers sometimes exaggerated the number of wolves in an area to enhance their job security and occasionally misrepresented where a wolf was killed in order to claim a local bounty. Furthermore, the widespread use of poisons meant that many animals, including wolves, were dispatched without any record of their death.

Regardless of inaccuracies in the historical record, a partial picture of where wolves occurred prior to their extermination in the Southwest can still be pieced together. These records show that at least small populations of wolves were found throughout the woodlands and forests of northern Arizona (Brown 1984). For example, from these records we know that there were at least 30 wolves on or near the North Kaibab because of the number reported killed between 1907 and 1926 (Russo 1964). Brown (1984) claimed that "the last wolf in this part of northern Arizona was taken on the Paria Plateau about 1928", but a former Civilian Conservation Corps worker recently reported that he saw wolves on three different occasions in 1935 on the North Rim of Grand Canyon National Park (GCNP) (Leslie, personal communication). Moreover, "as recently as March 3, 1948, assistant chief ranger A.L. Brown reported wolf tracks in fresh snow in the area of Bright Angel

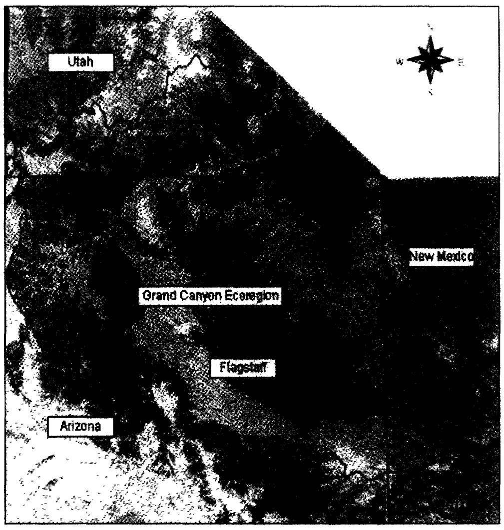

Figure 1. Grand Canyon ecoregion

Point" on the North Rim of GCNP (Hoffmeister 1971). Finally, the last wolf inhabiting the Mogollon Rim area in the southern part of the GCE was reportedly taken in 1942 (Hoffmeister 1986). Clearly, gray wolves occurred within the Grand Canyon Ecoregion well into the twentieth century, although their exact numbers and range will probably never be known with certainty.

Due to the taxonomic splitting approach of the time, Young and Goldman (1944) identified 23 subspecies of North American gray wolves (based on skull measurements, pelage color, and size) and mapped their geographic distribution. Two of these 23 nominal subspecies—*C.l. mogollonensis* (the Arizona wolf), *C.l. youngii* (the Great Basin or Intermountain wolf), and, possibly, *C.l. baileyii* (the Mexican wolf)—inhabited the Grand Canyon Ecoregion (Brown 1984; FWS 1996). Development of similar classification schemes continued into the 1970s (e.g., Hall and Kelson 1959) until some taxonomists began questioning the splitting tradition of wolf taxonomy.

Modern lumping systems of wolf taxonomy are based on multivariate statistical analysis of large sample sizes and confirmed by the results of contemporary molecular genetics (Wayne et al. 1992). Adopting this approach, Nowak (1995) lumps the two previously identified GCE wolf subspecies, (*C. l. mogollonensis* and *C. l. youngii*), in with the geographically widespread subspecies *C. l. nubilus*. Nowak (1995) also affirms the validity of a truly Southwestern subspecies, the Mexican wolf (*C. l. baileyii*), which may have occupied or dispersed into the southeastern part of the GCE.

Habitat capability and suitability mapping

Restoration of viable large carnivore populations is probably among society's greatest challenges, requiring extraordinary innovation and cooperative management on an ecoregional scale (Paquet and Hackman 1995). Furthermore, solutions to large predator conservation are economic, sociological and political (human dimension issues), as well as biological and ecological (biophysical factors) (Clark et al. 1996). The feasibility of wolf recovery depends on the capability and suitability of habitat for sustaining wolf populations. Although many factors can and should be considered, the ultimate determinants are ungulate prey and human impact (Fuller et al. 1992) or, put another way, sustenance and security.

Course screen landscape-scale habitat mapping for the Arizona portion of the GCE (see Figure 1) has been done following other similar studies (Mladenoff et al. 1995; Quinby et al. 1999; Ratti et al. 1999; Wydeven et al. 1998). Various biophysical factors can be considered in evaluating the capability of habitat to support wolves, but this study focuses on vegetation cover, surface water availability, and, most importantly, ungulate prey abundance. In addition to adequate food supplies, security from human disturbance and persecution are important factors affecting the suitability of a landscape for wolf recovery. At this stage in the research, three critical human dimension aspects are considered: human population density, road density, and land status.

Biophysical factors

Several reintroduction studies (e.g., Mladenoff et al. 1995) suggest that gray wolves, at least those living south of the Arctic, tend to prefer forested landscapes. Historically, in the Southwest, wolves were most commonly found associated with woodlands and montane forests (Groebner et al. 1995; FWS 1996). When observed elsewhere, such as in grasslands, they were probably simply passing through as they moved between their preferred habitat of forested highlands. Figure 2 maps the distribution of these two vegetation types, as well as others such as shrublands and grasslands. This figure plainly illustrates a broad band of forestlands-woodlands extending north-south from the Kaibab Plateau, through the Flagstaff area to the Mogollon Rim, interrupted only by the Grand Canyon and urbanized areas such as Flagstaff. Other areas of woodland/forest vegetation types are found in isolated mountain areas of the Arizona Strip as well as the Hualapai and Navajo Indian Reservations.

Because wolves require large amounts of water to aid digestion (Lopez 1978; Mech 1970), several studies of wolves in the Southwest (Groebner et al. 1995; FWS 1996) and elsewhere (Quinby et al. 1999) have suggested that the availability of free water is an important determinant of gray wolf abundance and distribution. Figure 3 illustrates the distribution of currently mapped lakes, springs, and streams in the Ari-

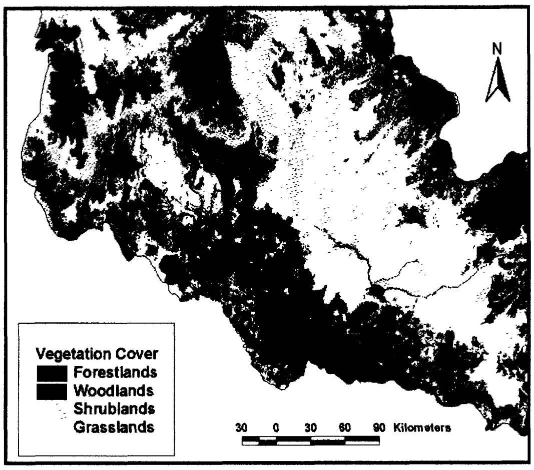

Figure 2. Vegetation cover

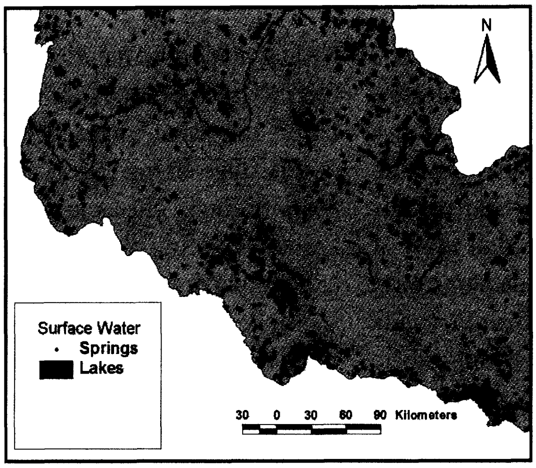

Figure 3. Surface water

zona portion of the Grand Canyon Ecoregion. Although the digital data available is very incomplete, this figure shows that there are more than enough sources of surface water on the Kaibab Plateau, in the Flagstaff area, and along the Mogollon Rim. This conclusion is supported by the observed presence of relatively high numbers of other large predators (e.g., mountain lions (*Felis concolor*) and prey species in these locales.

Clearly, one of the most important determinants of suitable wolf habitat is the abundance of ungulate prey species. The primary prey species for wolves in the Grand Canyon Ecoregion are mule deer (*Odocoileus hemionus*), followed in order of importance by elk (*Cervus elaphus*), pronghorn (*Antilocarpa americana*), and bighorn sheep (*Ovis canadensis*). Information about abundance (density) and distribution of these wildlife species in the Arizona part of the ecoregion was obtained from the Arizona Game and Fish Department. Figures 4 and 5 display the approximate density of mule deer and elk populations in Arizona GCE.

Other reintroduction studies (e.g., FWS 1996) indicate that a density of approximately two to six deer per km^2 would be required to support a Mexican wolf population and, presumably, similar numbers would be adequate for wolves in the GCE. Figure 4 demonstrates that much of the Kaibab Plateau enjoys a very high density of mule deer (eight to 13 animals per km^2), while the remainder of the area has an adequate density of three to eight deer per km^2. The Coconino Plateau around Flagstaff also supports quite dense populations (three to eight per km^2). Furthermore, similar densities probably exist on parts of the Hualapai and Navajo reservations in the GCE, but no data is readily available to confirm this supposition. Even if the current mule deer population density is one-half of what these figures indicate, as some Arizona Game and Fish Department personnel suggest (e.g., Goodwin, personal communication), there are still more than sufficient deer densities to support gray wolves. Figure 5 shows that elk densities, while somewhat lower on average than mule deer, are quite high (i.e. two to three animals per km^2) around Flagstaff and southeast along the Mogollon Rim. Of course, elk also average three times the biomass of deer. Figures 4 and 5 combined map an adequate ungulate prey base extending north-south from the Kaibab Plateau, through the Flagstaff area, and southeast along the Mogollon Rim.

Human dimensions

An important determinant of habitat suitability for gray wolves and other large carnivores such as grizzly bears (Merrill et al. 1999) seems to be human population density. Studies (e.g., Mladenoff et al. 1995; Ratti et al. 1999) have shown that lands with a human population density greater than 12 to 13 persons per square kilometer will not be suitable wolf habitat. The map displayed in Figure 6 indicates that most of the Arizona section of the Grand Canyon Ecoregion has population densities less than 13 persons per km^2. Except for the Flagstaff-Sedona urban zone, the entire north-south corridor from the Kaibab Plateau to the eastern, slightly urbanized, part of the Mogollon Rim has a human population density of less than four people per km^2. Not surprisingly, this same corridor has high prey species densities and seems capable of supporting wolves.

Wolves are usually not threatened by roads, except when they are struck by motor vehicles (Mech 1977). Nonetheless, roads can provide access to generally undisturbed areas where humans may harass or kill wolves. Studies of road density and wolf distribution relationships by Thiel (1985) and Mech et al. (1988) suggest a road density threshold value of between 0.6 and 0.8 kilometers of road per square kilometer of area. Higher road density values generally result in unsuccessful breeding attempts. Mladenoff et al. (1995), using radio collar data on recolonizing wolves in northern Wisconsin, discovered that road density and fractal dimension—reflecting the degree of habitat fragmentation (often the result of road building)—were the most important predictors of favorable wolf habitat. Figure 7 shows that most of the north-south corridor, extending from the Kaibab Plateau through to the Mogollon Rim southeast of Flagstaff, has road densities higher than 0.68 km per km^2, but generally lower than 1.4 km per km^2. Road density in many parts of the GCE is somewhat higher than recommended in other studies, but most of the numerous roads in the ecoregion are tertiary or unimproved roads that could be eliminated on public lands with a vigorous road-closing program. Furthermore, the low human density numbers (Figure 6) might indicate that these areas are favorable wolf recovery habitat despite the existence of relatively high unimproved road densities.

Favorable land status, defined here as lands in public ownership and, especially, designated protected areas, can help make a landscape suitable for gray wolf reintroduction (Southern Rockies Ecosystem Project, 1998). Identifying, describing, and mapping proposed and designated wilderness areas and other areas designed to protect ecological processes or wildlife, such as the Grand Staircase/Escalante National Monument and the Grand Canyon Game Preserve in the Kaibab Forest (Miller 1996) is especially important. Figure 8 maps distribution of public lands, both state and federal, exclusive of Indian reservations. This reveals that a wide band of federal public lands (including large tracts of protected areas) runs north-south from the Kaibab Plateau through the Flagstaff area and southeast along the Mogollon Plateau (again, corresponding with the distribution of important biophysical factors). State lands, even though currently interspersed in a "checkerboard" fashion (see Figure 8) with private and federal lands, could be consolidated through land trades and purchases to create wildlife corridors between federal public lands such as the Coconino and Kaibab National Forests.

The landscape-scale habitat mapping, included in this progress report, is admittedly somewhat incomplete at this stage in the research. Nonetheless, the mapped variables of both biophysical and human dimensions point strongly towards the probability that at least two areas—the Kaibab Plateau and much of the Mogollon Rim south and east of Flagstaff—are capable of supporting viable wolf populations and suitable for reintroduction of gray wolves

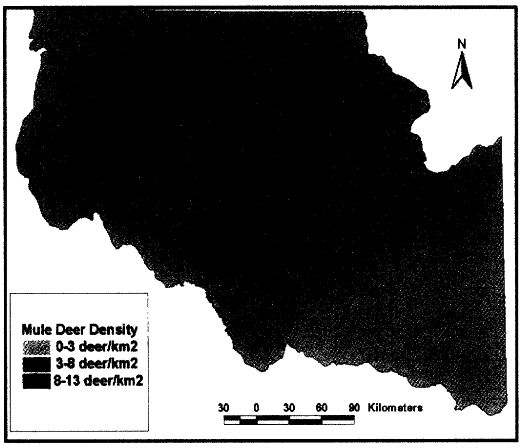

Figure 4. Mule deer density

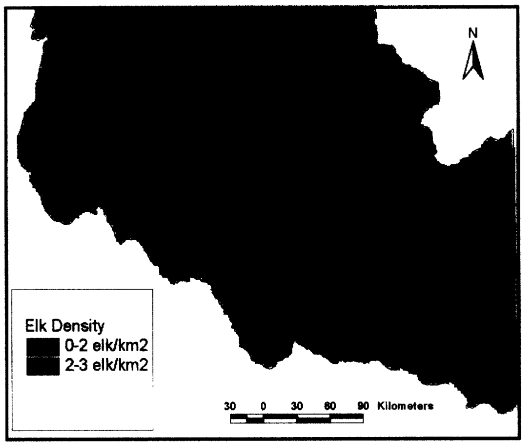

Figure 5. Elk density

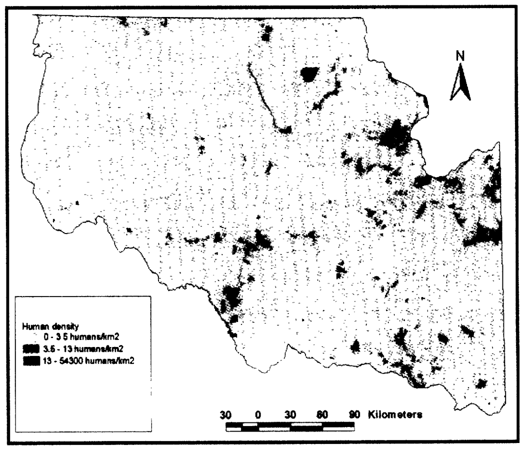

Figure 6. Human density

Projected wolf densities

Assuming that wolf reintroduction is feasible, it is reasonable to ask how many wolves might the Arizona portion of the Grand Canyon Ecoregion support. Utilizing the existing deer and elk density distribution maps (Figures 4 and 5), and following Fuller (1989), very preliminary calculations of predicted wolf density were done using these equations:

$$W = 3.4 + 3.7D$$

and

$$W = 3.4 + 3.7(3E)$$

where W is predicted wolf density (per 1000 km^2), D is estimated mule deer density (per km^2), and $3E$ is estimated elk density (per km^2) times a relative biomass value (elk biomass is 3 x 1 deer). Both low and high ungulate density estimates were utilized in calculating a range of predicted wolf numbers shown in Table 1.

Although this facet of the study is far from finished, historical records and initial carrying capacity research suggest that to reintroduce at least 100 gray wolves into the Arizona portion of the Grand Canyon Ecoregion would be feasible.

Potential stock for wolf reintroduction

When the time comes to make a decision about reintroducing gray

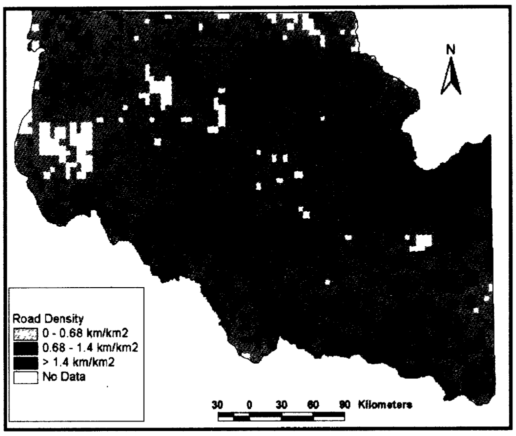

Figure 7. Road denisty

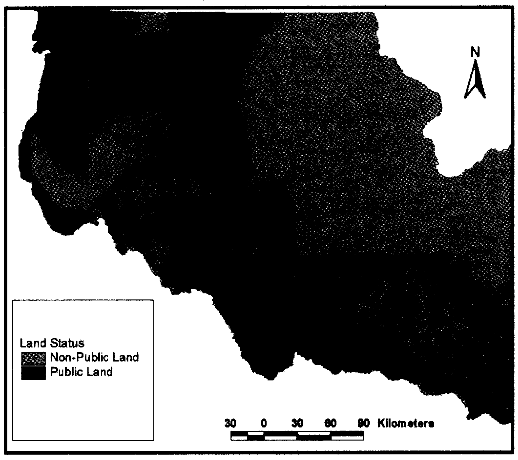

Figure 8. Land status

wolves to the Grand Canyon Ecoregion, we should "...consider behavioral or demographic factors to be more important than maintenance of the genetic purity of putative wolf subspecies..." (Wayne et al. 1992). If Nowak's (1995) recent revision of wolf taxonomy is accepted, it seems biologically appropriate that stock for reintroduction could be taken from anywhere in the historic range of *C. l. nubilus*. While finding areas of surplus wolf populations with habitat exactly comparable to the GCE will be difficult, regions such as the Great Lakes, currently supporting *C.l. nubilus* populations, do exhibit analogous forested ecosystems (albeit different forest types) and have similar ungulate prey species (i.e., deer and elk). Wild wolves translocated from the Great Lakes region, for example, would at least be habituated to forest habitats (as opposed to tundra) and experienced in hunting the types of ungulate species that are most abundant in the GCE.

Alternatively, captive or wild-bred Mexican wolves (*C.l. baileyii*) could be utilized for reintroduction. Given the difficulties experienced with captive bred stock in the current Mexican wolf recovery effort, however, it seems best to wait for the availability of surplus wild-raised stock GCE (Parsons, personal communication). Also, when the Mexican wolf population reaches a viable size in the wild, dispersers from eastern Arizona and western New Mexico will likely attempt to colonize the southeastern part of the GCE. Thus, this recovery opportunity in the GCE could help extend the geographic range and metapopulation of the currently recovering, but still endangered, Mexican wolf.

Conclusions

The first phase of this landscape-scale analysis involved utilizing six factors of the biophysical and human dimensions to identify and describe potential reintroduction sites in the Arizona section of the Grand Canyon Ecoregion. Initial results show that there are at least two localities in northern Arizona available for reintroduction of around 100 wolves. Source stock for wolf recovery in the GCE could come from existing large *C. l. nubilus* populations in the Great Lakes and/or a recovered *C. l. baileyii* population in the Southwest. Clearly, the future extension of wolf recovery into northern Arizona and other parts of the GCE will have to be done under the legal mandate of the ESA and will most likely be sponsored by a federal agency such as the Fish and Wildlife Service and/or National Park Service.

Further investigation and study will continue to refine the habitat capability and suitability analyses, as well as to help determine the most appropriate subspecies for wolf reintroduction in the ecoregion. In the end, however, the most important consideration is how to best assist nature in restoring gray wolves to the Grand Canyon Ecoregion and thereby help in the national effort to conserve this magnificent and ecologically essential carnivore.

Acknowledgements

This project was primarily funded by the Grand Canyon Wildlands Coun-

Table 1. Minimum and maximum number of wolves predicted to occupy Arizona part of Grand Canyon Ecoregion based on combined mule deer and elk densities and areas.

Location	Area (square kms)	Wolves (pop. range)
North Kaibab	1472	35 to 62
South Colorado Plateau	3849	80 to 125
Total Arizona GCE	5321	115 to 187

cil, Defenders of Wildlife, and Prescott College. Although I am solely responsible for any errors or shortcomings, I could not have made the progress I have without the help of these people: Kelly Burke, Larry Stevens, and Kim Crumbo of the Grand Canyon Wildlands Council; Melissa Grigione and Dan Smith, Defenders of Wildlife; Mike Siedman, Phoenix Zoo; Mike Senn, Rick Miller, John Goodwin, Ray Lee, Matt Alderson, and Amber Munoz, Arizona Game and Fish Department; Elaine Leslie, Grand Canyon National Park; Rick Moore and Aimee Porcaro, Grand Canyon Trust; Dave Hanna and Craig Martinsen, Prescott College; and, Jim Riedman, Arizona Land Resources Information System.

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Canis Soupus: Eastern Wolf Genetics and Its Implications for Wolf Recovery in the Northeast United States

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Abstract
*Efforts to restore wolves to the northeastern United States have been confounded by a new taxonomic proposal: that the wolf historically inhabiting this region was not, as previously thought, a subspecies of gray wolf commonly called the eastern timber wolf (*Canis lupus lycaon*), but rather a separate species closely related to the red wolf (*Canis rufus*) of the southeast United States. This hypothesis raises numerous biological, legal, policy, and management questions about potential wolf restoration. While restoring wolves could complete a broken food chain by providing a natural predator for moose in the northern forest ecosystem, the process of wolf restoration in the Northeast is in its infancy. Further studies must address biological, sociological, and economic impact questions, as well as answer the basic question of what wolf originally inhabited the northeastern forests?*

Introduction
Efforts to restore wolves to the northeastern United States have been confounded by a new taxonomic proposal: that the wolf historically inhabiting this region was not, as previously thought, a subspecies of gray wolf commonly called the eastern timber wolf (*Canis lupus lycaon*), but rather a separate species closely related to the red wolf (*Canis rufus*) of the southeast United States (Wilson et al. 2000). This hypothesis raises numerous legal, policy, and management questions about potential wolf restoration. In addition, wildlife managers now have a basic biological question to consider when debating the merits of wolf reintroduction to New England and upstate New York: what wolf should be restored?

Wolves were extirpated from the Northeast by the end of the nineteenth century (Fowler 1974). In 1974, a year after passage of the Endangered Species Act (ESA), the U.S. Fish and Wildlife Service (FWS) listed eastern timber wolves as endangered, except in Minnesota where a remnant population was listed as threatened. In 1978, the Service developed a recovery plan for the eastern timber wolf. At that time, scientists believed that the eastern timber wolf had historically ranged throughout the Northeast and west to the Great Lakes region. The recovery plan identified several areas in the Northeast as potential wolf habitat, including northwest Maine and the Adirondack Mountains of New York. These areas remained in the recovery plan when it was revised in 1992 but the Service did not actively pursue northeast wolf restoration.

A recent proposal by the Service to declare a Distinct Population Segment for wolves in the Northeast (Federal Register 2000) has triggered renewed interest in wolf recovery in this region. If enacted, this designation would separate the Northeast administratively under the ESA from wolf populations in the Great Lakes states and require the Service to develop a new recovery plan for New England and upstate New York. Recent studies indicate there are adequate habitat and prey to sustain a healthy wolf population in this region (Harrison and Chapin 1997; Mladenoff and Sickley 1998), and several surveys indicate strong public support for wolf restoration (Responsive Management 1996; Downs and Smith 1998). However, the unresolved issue of taxonomy, while certainly not the only impediment to northeast wolf restoration, is complicating the prospect.

Previous taxonomic classification
In this article, we use the term eastern wolf to refer to the wolf that, by

Nowak's description (1995), currently resides in southeastern Canada and formerly inhabited the northeastern United States. We also use the term western gray wolf to refer to what Nowak (1995) describes as *Canis lupus nubilus*, the larger wolf that formerly inhabited much of the western United States and much of Canada. Goldman (1937) classified the eastern wolf as *Canis lupus lycaon*, a subspecies of gray wolf, and for years its historic range was thought to be the northeastern United States as far west as the Great Lake states and north into southern Ontario and Quebec (Goldman 1944; FWS 1992; Nowak 1995). The red wolf is classified as *Canis rufus*, a distinct species (Goldman 1937), and its historic range has long been considered to be the southeastern United States as far west as Texas and as far north as Pennsylvania (Nowak 1995). By these boundary definitions, the ranges of the red wolf and eastern wolf would have originally overlapped in the mid-Atlantic states (Nowak 1995), including Pennsylvania, West Virginia, and Virginia (see Figure 1).

Over the years, scientists noted similarities between the two canids based on morphology and skull measurements (Lawrence and Bossert 1975; Alexander 1983). The two animals are apparently so similar that one report in the literature 25 years ago refers to a red wolf in Algonquin Park, Ontario (P. Wilson, personal communication), even though scientists did not think they ranged that far north.

Recipe for canid soup

Scientists noted another trait common to both red wolves and eastern wolves—the tendency to hybridize with coyotes, *Canis latrans* (Wilson et al. 2000). Coyotes, historically absent from the east, reached Ontario in the early 1900s as habitat alteration and fragmentation favored their eastward expansion (Clarke 1970; Wayne and Lehman 1992; Roy et al. 1994). In the 1920s, coyotes moved east across southern Ontario and eastern Quebec and by the 1930s had reached New York and New England (Parker 1995). The southeastern United States similarly underwent landscape changes that enabled coyotes to expand to this region as gray and red wolves were extirpated in the last century (Jenks and Wayne 1992; Parker 1995).

As coyotes expanded eastward, they encountered dwindling populations of both eastern and red wolves. There has been much discussion in the literature about the propensity for and degree of interbreeding between red wolves and coyotes, both earlier this century and since their reintroduction in North Carolina (Wayne and Jenks 1991; Nowak 1992; Kelly et al. 1999). At present, the FWS is struggling to keep reintroduced red wolves from hybridizing with coyotes. If the current rate of interbreeding is not halted, red wolf genes will be completely diluted within a few generations in a process known as genetic swamping (Kelly et al. 1999).

Genetic testing of the relatively large coyotes from the Adirondack Park and central New York similarly indicates a history of interbreeding with wolves. The degree of wolf genetic material varies across these samples, with some being more "wolf-like" than others (Chambers 2000). Genetic testing on northern New England canids shows interbreeding as well, though more sampling is needed. In the Frontenac Axis region of Ontario, southeast of Algonquin Park, a similar, though slightly larger canid is commonly called the Tweed wolf, and is probably a hybrid containing more wolf genes than coyote genes (Edwins et al. 2000; P. Wilson personal communication). A coyote-wolf mix is commonly found west of Algonquin Park, whereas the park itself maintains the most wolf-like form of eastern wolf (Wilson et al. 2000).

The result of these genetic analyses is the discovery of a canid gradient in the northeastern United States and southeastern Canada, containing a mix of eastern wolf, western gray wolf and coyote genes. This phenomenon is often lightheartedly referred to as "canid soup." According to Wilson et al. (2000), in the northeastern United States, coyote genes dominate this mix. In southeastern Ontario, eastern wolf genes are more common, and in northeastern and northwestern Ontario an eastern wolf/western gray wolf animal may be predominant. The extent to which eastern wolves interbreed with western gray wolves is currently unknown. The regions in far northern Ontario are predicted to be predominantly western gray wolf, but at present the boundary between the eastern wolf and western gray wolf is not well established (P. Wilson, personal communication).

Genetic hypothesis

The tendency for eastern and red wolves to hybridize with coyotes is not observed in western gray wolves (Edwins et al. 2000; Wilson et al. 2000). Additionally, there are morphological characteristics shared by red and eastern wolves but not western gray wolves, such as the smaller size of wolves in the East (Goldman 1944). These commonalities led researchers to examine more closely the relationship between red wolves and eastern wolves. They hypothesized that red and eastern wolves were more closely related to each other than either was to western gray wolves (Wilson et al. 2000). In the past decade, researchers from the genetics labs at Trent and McMaster's universities in Ontario have conducted genetic analyses of canids

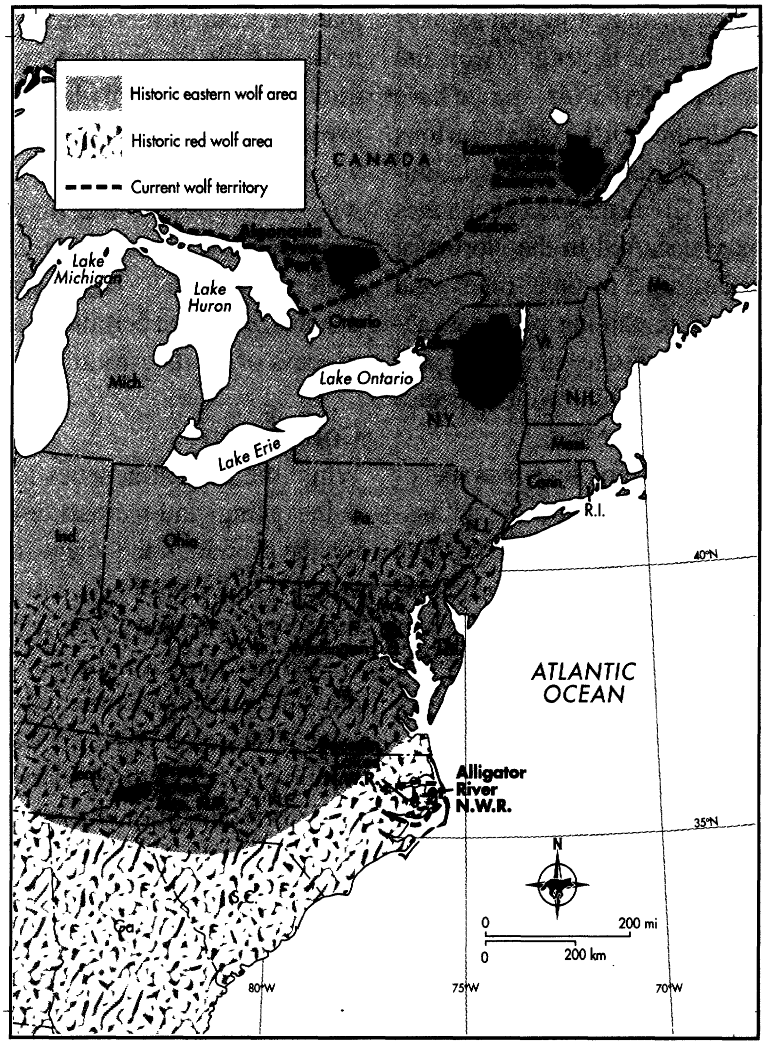

Figure 1. Historic and current ranges of the red and eastern wolf

from throughout the red and eastern wolf range, using various criteria to compare relatedness. Their data support the hypothesis that the red wolf and eastern wolf have a common North American origin separate from that of the western gray wolf. In fact, these researchers suggest that the red wolf and eastern wolf are actually the same species, and they propose changing the scientific name of both to *Canis lycaon*, with a common name of red wolf (Wilson et al. 2000). The recommended name is based on historical precedence in early wolf taxonomy (Brewster and Fritts 1995).

According to Wilson et al. (2000), North America was inhabited by a common canid ancestor one to two million years ago. At some point, some of these animals traveled to Eurasia over the Bering land bridge and evolved into the gray wolf. The remaining canids evolved wholly in North America. Between 150,000 and 300,000 years ago they diverged into the coyote, which adapted to preying on smaller mammals in the arid southwest, and the eastern/red wolf, which adapted to preying on white-tailed deer (*Odocoileus virginianus*) in eastern forests. Gray wolves returned to the North American continent approximately 300,000 years ago, adapting to preying on large ungulates throughout the western United States and Canada. According to this hypothesis, coyotes are more closely related to the eastern/red wolf than to the western gray wolf, hence the propensity for interbreeding (Wilson et al. 2000).

Discussion

If the issue of eastern canid taxonomy is the subject of debate for academicians, it is nothing less than confounding to the public, legislators and even wildlife managers, who want to make informed decisions about wolf restoration in the Northeast. The concept that the eastern wolf and red wolf are the same species complicates an already complex topic, presenting numerous questions to all those interested in examining the potential for wolf recovery in New England and New York. What canid originally occupied the Northeast and what canid is there now? What role is the present canid filling and what role would a restored, larger canid fill? What wolf population would scientists use as source animals, and would they displace or interbreed with the Northeast's resident coyotes?

Because of the tendency of the eastern wolf to hybridize with coyotes, it is a fair assumption that if it were reintroduced in the Northeast it would be vulnerable to interbreeding with eastern coyotes. Therefore, despite efforts to reintroduce a separate species in the Northeast, biologists might just be regenerating an animal that is already present in the form of a large, hybrid coyote. It is important to note, however, that the core population of eastern wolves within Algonquin Park is not readily hybridizing with coyotes, although the reason for this is unclear (Edwins et al. 2000). Habitat saturation by established wolf packs might be one explanation. Protected wolves that are able to maintain long-term territories are able to prevent the encroachment of coyotes into core wolf areas (J. Theberge, personal communication). Canadian wolf researcher John Theberge has documented short-term invasion by coyotes following the breakup of a resident wolf pack within Algonquin's core wolf zone (J. Theberge, personal communication).

Understanding this relationship could prove useful in the Northeast to determine what conditions are necessary to discourage interbreeding.

From an ecological perspective, wildlife managers will have to determine what the most appropriate species of wolf is for the modern day moose-dominated North Woods. Should a species' historical presence be the determining factor, or should the most suitable candidate be selected to match the ecological conditions existing today or likely to exist in the future? If so, the wolves in the Laurentides area of Quebec might provide a better source population for the northeastern United States. The close proximity of the two regions might allow some genetic exchange. Preliminary testing indicates that genetically, the Laurentides animals may be a mix of gray and eastern wolf (P. Wilson, personal communication), a finding which is further supported by their large size of up to 100 pounds. The Laurentides wolves prey primarily on moose (*Alces alces*), a plentiful prey item in Maine. Most importantly perhaps, these animals may be less likely to hybridize with eastern coyotes than the Algonquin-type wolf, though this crucial relationship remains to be tested (P. Wilson, personal communication). In the world of survival of the fittest, it is unlikely that natural selection would favor a moose-eating canid that compromises its size by breeding with the much smaller eastern coyote.

This theorizing, however, begs the question: did this larger gray wolf ever exist in the Northeast? For now, scientists can only speculate, but it is generally accepted that the northeastern United States was primarily a moose-caribou ecosystem before European settlement (D. Harrison, personal communication). It is questionable whether the deer-adapted eastern wolf would have thrived in this environment, indicating that perhaps both canids—the larger gray wolf and the smaller eastern wolf—might have inhabited the northeast at various points (P. Wilson, personal communication). Given the radical changes that have occurred in the Northeast ecosystem since colonial times, and the lack of remaining physical evidence of the presence of wolves, it is difficult to determine which species may have been present.

Similarly, some scientists speculate that the eastern wolf did not historically occur north of the Canadian border (P. Wilson, personal communication). Southeastern Canada was home to moose, woodland caribou (*Rangifer tarandus caribou*) and elk (*Cervus elaphus*) and would have more likely contained the larger gray wolf. Overhunting and habitat alteration likely contributed to the decline of some of these species, including the wolf (Nelson 1997). As intensive logging encroached into southeastern Canada, white-tailed deer populations expanded northward from the United States, thriving in the second growth forests that resulted. Eastern wolf populations may have likewise moved north in an expansion similar to their primary prey species. Today, the range of the eastern wolf may extend as far west as Saskatchewan and include northwestern Ontario and Minnesota. These northern areas may contain both eastern wolves and western gray wolves, but more research is needed before this can be determined (Wilson et al. 2000).

The sociological implications of the new taxonomic proposal are as challenging as the biological. Wolf restoration is always politically divisive. Groups and individuals opposing wolf restoration have already latched onto the current debate, declaring that the role of the eastern wolf is filled by the large, resident eastern coyote, which many call the "brush wolf." If scientists determine that the Laurentides animal is the most suitable for Northeast reintroduction, but cannot definitively prove its earlier existence there, would the public accept a species whose original range may have stopped short of the proposed reintroduction area? These types of debates will most certainly slow the process of wolf restoration.

Conclusion

While the current genetics research is intriguing, and increasingly gaining the support of the scientific community (Kelly, personal communication), further research and discussion is essential before conclusions can be drawn. In the meantime, other biological factors are ripe for wolf restoration. The northern forests have made a remarkable recovery in the last fifty years and now comprise more than 26 million acres—ample habitat to support a population of wolves (Harrison and Chapin 1998; Mladenoff and Sickley 1998). With the expansion of moose and beaver (*Castor canadensis*) and the occurrence of white-tailed deer throughout the north woods, there is ample prey. Most scientists agree that there is an ecological role for a larger canid in the northeast's North Woods (Harrison quoted in Fascione and Kendrot, 2000). Even though coyotes occasionally form packs, they do not normally prey on moose. Restoring wolves could complete a broken food chain by providing a natural predator for moose in the northern forest ecosystem. Since the 1995 wolf reintroduction in the northern Rockies, the Greater Yellowstone ecosystem has undergone significant ecological changes as a result of the restoration of this top predator. Recent research in Yellowstone National Park suggests that the effect wolves have on their prey can benefit vegetative structure and overall species biodiversity (Ripple and Larsen, in press).

The process of wolf restoration in the Northeast is in its infancy, however, and further studies must address biological, sociological and economic impact questions (Fascione et al. 2000). While the proposed taxonomic revision highlights the need to identify the connections and potential interactions between eastern wolves, red wolves, gray wolves and eastern coyotes in the Northeast, a clearer picture of these complex relationships is not likely to simplify the management, policy, legal, and moral issues that follow. Although the questions raised by this taxonomic discussion are unique to the Northeast, controversy itself is nothing new to wolf restoration. Ultimately, genetics are just one more piece of information that wildlife managers, advocates, and the public will have to distill and blend with the ecological and sociological realities of not only today's environment, but tomorrow's as well.

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Captivity, Inbreeding, Cross-Lineage Matings, and Body Size in Mexican Wolves

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Abstract
*The Mexican gray wolf (*Canis lupus baileyi*) appears to be extinct in the wild and exists now only in captivity and in a single, small, reintroduced population. A recent study of captive animals found no evidence for inbreeding depression in juvenile viability or litter size. We examined the relationship between inbreeding and body weight in captive wolves. We found that captive wolves with little or no known inbreeding had lower body size than wild caught wolves. In addition, captive wolves with higher inbreeding had lower body size than captive wolves with little or no inbreeding. The captive population was descended from three founders until two other lineages, each descended from two founders, were recently added to the population. Consequently we examined the potential statistical power associated with future comparisons of body weights between inbred wolves and the offspring of cross-lineage matings. In this case it appears likely that in the next few years there will be an adequate sample size to statistically evaluate differences in body size between these groups.*

Introduction
Minimizing inbreeding and the loss of genetic variation are major concerns of captive breeding programs for endangered species. Increases in inbreeding have been correlated with decreases in juvenile survival in captivity in a number of different species (Ralls et al. 1988). In addition, a number of recent studies in wild populations have demonstrated that a reduction in genetic variation resulting from inbreeding and/or genetic drift has been associated with the decay of phenotypic attributes related to fitness in wild populations (see references in Hedrick and Kalinowski 2000).

Mexican wolves *(Canis lupus baileyi)*, the southern-most subspecies of gray wolf, currently exist primarily as a captive population numbering about 200 individuals. Other than a small, recently reintroduced population in Arizona and New Mexico, there have been no confirmed sightings of wild Mexican wolves in more than 20 years. The current population of wolves originated from three independent captive lineages with each having only two or three founders captured during the 1960s and 1970s from northern Mexico and southern Arizona. Because of the few founders, each of the lineages has become inbred in captivity. Following a genetic evaluation of the three lineages (Hedrick et al., 1997) in which no evidence of coyote, (*Canis latrans*), or dog, (*Canis familiaris*), ancestry was found, the McBride, Ghost Ranch, and Aragon lineages were recently combined to form a single population with seven founders.

The most common phenotypic trait used to evaluate the effects of fitness in captive populations of endangered species has been juvenile survival; secondarily, reproductive traits, such as litter size, have been used. Kalinowski et al. (1999), however, recently found no effect of inbreeding on juvenile survival and litter size in captive Mexican wolves. The only other trait for which a substantial amount of data currently exists for captive Mexican wolves is adult body size measured by mass. Some body size data is also available from historical wild-caught wolves (see below). For these reasons and others, body size has been proposed as a phenotypic indicator with which to monitor the effects of merging the three lineages. Because until recently there had been no breeding between the three lineages, offspring from crosses between lineages will be free of inbreeding. If inbreeding has negatively affected body size or other fitness related traits then individuals free of inbreeding may show changes in these traits. Except for a few offspring that have resulted from recent crosses between the lineages, all Mexican wolves alive today are inbred to some extent.

For wolves, changes in body size resulting from inbreeding in captivity may result in reduced fitnesses of individuals reintroduced into the wild. This may consequently result in a lower viability of wild, reestablished populations because body size is important in successful prey capture. Further, if body size has been affected by inbreeding, other fitness-related traits may also be adversely affected. Besides inbreeding, the captive environment and its related husbandry may also potentially result in changes in fitness-related traits by exerting selective forces different from the natural environment.

Here we look for evidence of an effect of inbreeding on adult body size in captive Mexican wolves and evaluate the utility of this trait for detecting potential future changes in size in offspring from cross-lineage matings. Specifically we ask (1) whether historic wild-caught adult wolves were larger than captive adult wolves with little or no inbreeding and (2) whether wolves in the McBride captive lineage with little or no inbreeding were larger than adults later in the lineage with greater amounts of inbreeding. In addition, we examine the statistical power associated with future comparisons when larger samples of inbred and cross-lineage wolves become available to assess the likelihood of detecting differences associated with inbreeding and outbreeding, should they occur.

Materials and methods

For comparisons involving historic, wild wolves we used the masses of 28 wild Mexican wolves killed in Chihuahua and Durango, Mexico during the 1970s as part of control efforts (McBride, 1980, as reported in Brown, 1983). Full masses were reported for only eight of the 20 wolves; the remaining wolves had their viscera removed before weighing. Sizes of gutted wolves were adjusted to approximate their full weight (Table 1). For comparisons involving captive wolves, we used 71 masses from 26 male and 24 female adult wolves from the McBride captive lineage. Two of these wolves were wild-caught, and one was a lineage founder. Only masses from wolves 1.5 to 9.5 years old that were taken from August through January were used. Of the three lineages, only the McBride lineage was managed from its inception as part of a captive breeding program; the weight data used here was taken during the course of these activities. Because the other lineages were not similarly managed until more recently, similar data for the Ghost Ranch and Aragon lineages do not exist.

Because the level of inbreeding in the captive wolves varied we split the wolves into two groups based on the level of inbreeding in individuals. Wolves with no inbreeding in captivity (inbreeding coefficient $f = 0$) and wolves comprising the first generation of inbred individuals ($f = 0.125$) were combined to form a group with little or no inbreeding, and wolves with greater levels of inbreeding ($f > 0.125$) were combined to form the second group (Table 1). The maximum inbreeding coefficient in our sample was 0.25. Because body size varies between the sexes, male and female wolves were considered separately in all analyses. Comparisons between groups were made using t and nonparametric Mann-Whitney U tests.

Results and discussion

Comparisons using existing data

For wild and captive wolves combined, female body sizes ranged from 20.2 to 37.4 kg, while male sizes ranged from 21.8 to 41.3 kg. Historic wild male and female wolves were significantly larger than wolves in the McBride lineage with little or no inbreeding (females: $t = 4.01$, df $=24$, $P < 0.001$; males: $t = 4.24$, df $= 23$, $P < 0.001$). Even with the limited data currently available, the mean size of inbred female wolves ($f > 0.125$) was significantly smaller than those of females with little or no inbreeding (Mann-Whitney $U=112.00$, $P < 0.01$), however, there was not a significant difference between males ($t=1.18$, df$=24$, $P=0.129$) (Table 1). The mean difference in body size for females in the two groups is only 3.0 kg (about an 11% size reduction), a value that becomes significant primarily because of the low variance in body mass in inbred females (2.4). The comparison between males had only a 30% chance of detecting a difference at the $\alpha=$

Table 1. Sample sizes and mean body sizes (kg) for historic, wild Mexican wolves and captive wolves by level of inbreeding.

		Inbreeding	Sample size	Mean size
Wild	Females		16	31.9
	Males		12	37.6
Captive	Females	$f \leq 0.125$	10	26.4
		$f > 0.125$	14	23.3
	Males	$f \leq 0.125$	13	30.9
		$f > 0.125$	13	28.7

0.05 level if a difference truly exists, given the means, variances and sample sizes of the two groups.

Although inbreeding did not appear to have an effect on viability and litter size in Mexican wolves (Kalinowski et al., 1999), our results provide some evidence of changes in body size that may be manifestations of inbreeding depression. Conclusions from these small samples, however, may or may not be supported when larger samples from inbred wolves in the McBride lineage become available. Such samples would be helpful in determining more definitively whether there is any evidence of a decline in size among inbred wolves. Masses from additional wolves with little or no inbreeding in the McBride lineage will not be available because most of these individuals are dead and no such matings are now possible.

Comparisons using future data

Analyses of statistical power suggest that differences between males with little inbreeding and those with greater inbreeding will probably not be detectable with the sample sizes attainable in the next few years without a lower mean mass or variance in the larger sample of inbred wolves. Differences, however, between inbred wolves and the offspring from cross-lineage matings will likely be detectable, should they occur, with the samples sizes likely available in the next few years. In other words, within the next few years, assuming several litters of cross-lineage wolves are produced per year, enough data should soon be available to evaluate the effect of merging the lineages on body size. It will be interesting to determine if crossing to other lineages will genetically restore fitness-related traits in Mexican wolves, as found in recent studies of other species.

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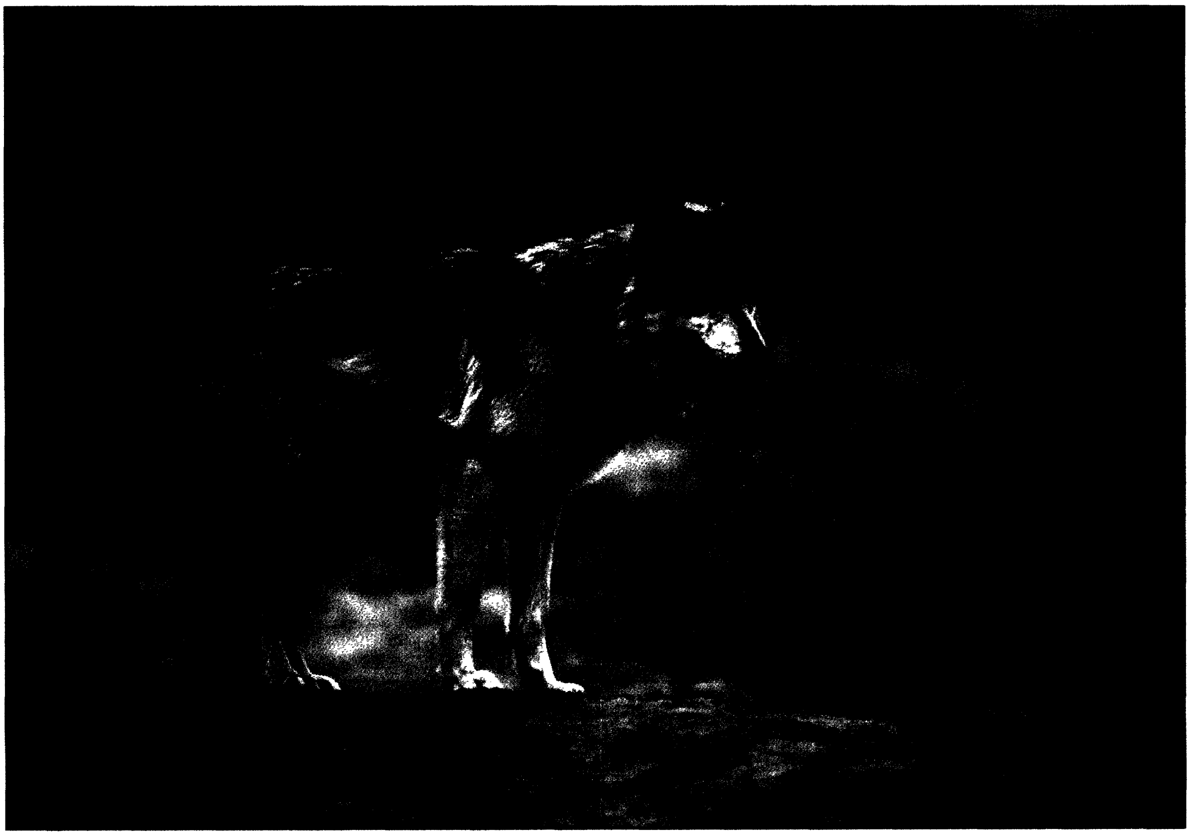

Mexican wolf (*Canis lupus baileyi*). Photograph by B. Moose Peterson/WRP

First Swift fox, *Vulpes velox,* Reintroduction in the USA: Results of the First Two Years

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Abstract
The swift fox, Vulpes velox, *is native only to the great plains of North America. The species was completely eradicated from its Canadian range by 1938, and from over 90% of its range in the USA by 1993. There have been two attempts at reintroducing the swift fox in to its historic range, one in Canada (1983 to 1997) and one on Blackfeet Tribal Land in the USA (1998 to present). The reintroduction of the swift fox to the Blackfeet Tribal Lands in Montana began in August 1998. This is the first swift fox reintroduction to be undertaken in the USA. The project is a five-year partnership between the Blackfeet Fish and Wildlife Department, Cochrane Ecological Institute and Defenders of Wildlife. The reintroduction methodology used was based on that developed by the Cochrane Ecological Institute. A total of 76 swift fox have been released to date, and two years remain in the program. Survival of released animals is still being investigated; survival of radio collared swift fox released in 1999 was 75% over a one-year period. Cubs have been produced in all years of the program. More than 24 cubs were observed in the field in 1999 and 2000.*

Introduction
Two species of fox, the kit fox (*Vulpes macrotis*) and the swift fox (*Vulpes velox*), are native to the great plains of North America. Historically, the swift fox range coincided with that of the North American bison, *Bison bison.* The species was once commonly found over a vast area, ranging from the eastern slopes of the Rocky Mountains in the west to the Souris hills of Manitoba in the east, north to the banks of Canada's North Saskatchewan River and south to the Texas panhandle and the New Mexico border in the U.S.A (Mackenzie 1911; Carlington 1980; MacGregor 1998).

The swift fox is North America's smallest canid, weighing between 2.3 and 3.2 kg. They are opportunistic feeders, eating seeds, berries, grass, insects, amphibians, reptiles, and small animals and birds (Uresk and Sharps 1986; Bremner 1997). If water is available they require 210 g of food per day, in the absence of water they absorb their needed liquid from their food and require 330 g of food per day (Flaherty and Plaake 1986). Swift foxes are preyed upon by both terrestrial and avian predators and are extremely den dependent. Swift foxes are sedentary, extremely social and largely monogamous. They rely on the burrows of other species for survival, making use of them as escape terrain and den sites. Swift fox are, in general, more active at night than during the day but this activity pattern varies with the season (Pruss 1994; Teeling 1996).

A subspecies of the swift fox, the northern swift fox, *V. v. hebes,* described by Merriam (1902) was briefly listed as endangered in the U.S. The species was delisted on the basis that a valid subspecific variation did not exist, supported by Hall's statement that *Vulpes velox* was conspecific with *Vulpes macrotis* (Hall 1981). By 1980, the northern swift fox had vanished throughout its Canadian range and from much of its northern range in the U.S. reducing the possibility of subspecies verification. Definitive research, combining morphological and molecular systematic data (Stromberg and Boyce 1986; Wayne1998) has resulted in the acceptance of distinct taxonomic descriptions for kit fox and swift fox.

In 1978, the Committee On the Status of Endangered Wildlife In Canada (COSEWIC) classified the swift fox as extirpated in Canada. By 1995, the species was considered extirpated from 90% of its historic range in the USA (United States Federal Register Vol. 60, No. 116, 1995). The causes of the extirpation of this species were attributed to the rapid and radical change of the great plains ecosystem from native grasslands to cultivated farmland, and the inevitable hunting, trapping and poisoning programs which accompany such habitat transformation (Carlington 1980; Weagle and Smeeton 1995).

Captive breeding of the swift fox for reintroduction into their original Canadian range was initiated in 1972, at the Wildlife Reserve of Western Canada (now the Cochrane Ecological Institute, CEI), using four founder foxes from the United States. Over the period of the program CEI has continued to acquire swift foxes to add to the captive colony from the USA. No Canadian animals were available for addition to the swift fox captive breeding group as the species had been extirpated from that country. The captive breeding program was initiated solely to provide swift foxes for reintroduction and not for exhibit, sale or trade. M.R. Smeeton, of the CEI, started the swift fox studbook in 1972 and in 1986 the studbook information was transferred to ISIS software. That studbook has now been consistently maintained for over 29 years. The ISIS software (ISIS 1989) is used to ensure maximum genetic heterozygosity in new breeding pairs (inbreeding coefficient <0.05) and, also, that animals of the same bloodlines are not repeatedly reintroduced into the same geographic area. In mid-1980, on the advice of the newly formed Swift fox Propagation Committee, swift foxes from the CEI captive breeding colony were sent to three zoos (Calgary, Edmonton, and Moose Jaw) as breeding and educational exhibit animals. By 1997, none of the zoos participating in the breeding program had swift foxes, having either returned their swift fox stock to the CEI, released, or euthanized them.

After founding the CEI's captive breeding colony in 1972, Miles & Beryl Smeeton went on to sign a cooperative agreement with Dr. Steven Herrero, then Dean of the Faculty of Environmental Design at the University of Calgary, in 1977. This agreement outlined a series of research projects to be conducted as M.Sc. theses (Carlington 1980; Reynolds 1983; Schroeder 1987) on potential release sites and methodology. The Canadian Wildlife Service (CWS) a branch of the Federal Government of Canada, became involved for the first time in the program in 1978, when the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) designated the swift fox as "extirpated" in Canada. The Canadian federal government's jurisdictional involvement in the swift fox reintroduction program was joined by provincial government involvement when the provinces of Alberta (1983) and Saskatchewan (1985) permitted swift fox releases within their jurisdiction. The last swift fox reintroduction took place in Canada in 1997, and in 1998 the species was downlisted from extirpated in Canada to endangered in Canada by COSEWIC (COSEWIC1998)

In the summer of 1996 a rancher from Montana visited the CEI. Upon his return to the U.S., he contacted the Blackfeet Fish & Wildlife Department (BFWD), to discuss the possibilities of reintroducing swift fox into their original, historic range on the Blackfeet Tribal Lands. Swift fox were considered extirpated from the Blackfeet tribal lands by the 1950s, and extirpated from the State of Montana by the 1960s (Knowles 1991). The Blackfeet Tribe showed interest in the project, provided that the proposed reintroduction sites appeared suitable, and offered to provide release sites, protection and monitoring for the projected swift fox program.

At the Swift Fox Symposium in Saskatoon, Saskatchewan, February 1998 the project was further developed and, as result of these discussions, Defenders of Wildlife were contacted to explore project funding.

In June 1998, The BFWD invited the Directors of CEI to the Blackfeet tribal lands to examine possible swift fox reintroduction sites (Figure 1). After this meeting and the examination of the potential release sites BFWD initiated the project by requesting CEI to provide swift fox to the Blackfeet Nation for a reintroduction program.

In July of 1998 an 11 day baseline survey of potential swift fox habitat, escape terrain, and prey abundance and predator abundance was conducted to find suitable swift fox release sites on the AMS Ranch on the Blackfeet tribal lands. This survey identified seven release sites to be used for the initial release, that occurred in late August of 1998.

Project design

The project was designed as a partnership of three groups: Blackfeet

Figure 1. Land use in the release areas on the Blackfeet Tribal lands, Montana

Fish and Wildlife Department who provided the lands for the release, protection of the reintroduced animals and some post release monitoring; Cochrane Ecological Institute who maintain the captive colony, and provided swift foxes for release, as well as the logistical planning for the releases, including liaison with government agencies in USA and Canada, development of nonintrusive survey methodology and research from graduate students; and Defenders of Wildlife who provided a significant proportion of the funding and help obtaining the requisite additional funding needed for the project.

The protocol used to release the swift foxes on Blackfeet land followed that described in Smeeton and Weagle (2000). After identifying potential release sites through a pre-release survey, Portable Protective Shelters (PPS) were set out in the release sites. One set of observers and one PPS was used per sibling group. Each sibling group was released in the vicinity of the PPS and, for a period of two days to one month, the released animals were kept under close observation. At the end of 10 days the PPS was removed from the site. The use of PPS encouraged the animals to remain in the area where they were released. Longterm monitoring of the population after the first month of intensive observation was conducted and coordinated by Blackfeet Fish and Wildlife and the CEI. Subsequent swift fox releases took place in areas where pre-release surveys indicated swift fox presence.

Results to date

Table 1 summarizes the swift fox released to date on the Blackfeet Tribal lands. In 1998, 30 juvenile swift fox were released on the AMS Ranch on the Blackfeet tribal lands. This was the first attempt at reintroducing swift fox to be undertaken in the USA. Throughout the first winter following the release (1998/99) the reintroduced animals were monitored, using tracks and scat, by officers of the Blackfeet Fish & Wildlife Department. A spring survey by CEI found abundant swift fox sign, three active dens, one pair with a litter of four cubs and one other den with a possible litter of three. A later survey, in early August, discovered a den containing 2 cubs. Two known mortalities occurred during the first winter, the cause of which was vehicle impact, as the swift foxes were hunting on or near roads.

Table 1. Summary of data on the swift fox released on Blackfeet Tribal Land, Montana from 1998 to 2000.

Year	Swift Fox Released (Fall)	Radio Collared	Cubs Found (Spring)	Recorded Deaths
1998	30	0	N/A	2
1999	15	8	4 (possibly 9)	2
2000	31	16	20+	3 (to mid October)

In 1999, 15 mixed adult and juvenile swift foxes were released on the AMS Ranch, eight of these, all adult animals, were radio collared. A radio telemetry program, undertaken by the Blackfeet Tribal Fish & Wildlife Department in the spring of 2000, found 5 of the collared swift fox and additional searches in August found a sixth radio collared swift fox. Although the analysis of the data is not complete first indications are that 75% of the captive bred, radio-collared swift foxes survived the winter.

In 2000, a spring survey undertaken by the CEI and the Blackfeet Fish and Wildlife Branch discovered five swift fox natal den sites containing more than 21 cubs. These cubs were all born to swift foxes without radio collars.

In general, (although they were not trapped and handled to confirm) the cubs sighted on the Blackfeet land in 2000 appeared to be older than the cubs born to new pairs at the CEI in 2000. This indicated either:

(1) that the difference in latitude between the more northerly CEI and Blackfeet lands to the south resulted in an earlier mating season, or

(2) that these cubs had been born to established pairs, rather than new pairs, either to the adult pairs released in 1999 or juvenile pairs, which had bred in 1999 and again in 2000. Established pairs do not go through the extended mating ritual that newly paired animals do, as a result their cubs are bigger because they are born earlier in the year (Smeeton et al. in press).

In August, 2000 31 swift fox were released on the AMS Ranch. Sixteen of these animals were radio collared. To date there have been three mortalities, two of the mortalities occurred in the first two weeks after the release.

Monitoring of the radio collared animals released on Blackfeet tribal lands continues over the winter of 2000 to 2001.

Conclusion

The Blackfeet swift fox reintroduction project seems to have benefited from the lessons learned in the Canadian Swift Fox Reintroduction Program (1983 to 1997). The use of the PPS release methodology, developed by the CEI over the latter part of the

Canadian reintroduction program, encouraged the animals to remain in the release sites and reduced mortality. The full support for the program by the Blackfeet people and the protection provided to the released animals by the Blackfeet Fish & Wildlife Department has ensured that none of the animals released on Blackfeet lands, unlike those released in the Canadian program, were killed by being accidentally poisoned, trapped, or shot. Pre-release surveys to identify suitable release sites, undertaken by the CEI on the Blackfeet Tribal Lands, indicated that the area was suitable and that there was sufficient escape terrain and a large enough prey base to support reintroduction. The survival and successful reproduction of the swift foxes released in Montana on the Blackfeet tribal lands appears to confirm that assessment. Unlike the Canadian reintroduction program, only captive bred swift foxes have been reintroduced in the Blackfeet program and these animals have survived, bred, and successfully raised their young on the release sites. Radio telemetry data has shown 75% survivorship over one year from those radio collared captive bred swift foxes that have been reintroduced onto the Blackfeet tribal lands.

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Interspecific Interactions Among Wild Canids: Implications for the Conservation of Endangered San Joaquin Kit Foxes

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Abstract

*Interspecific interactions among wild canids have significant implications for the conservation and recovery of endangered San Joaquin kit foxes (*Vulpes macrotis mutica*). Coyotes (*Canis latrans*) and non-native red foxes (*V. vulpes*) both engage in interference and exploitative competition with kit foxes. Several behavioral and ecological adaptations of kit foxes ameliorate such competition with coyotes and facilitate their coexistence. These adaptations include habitat partitioning, food partitioning, opportunistic foraging patterns, and year-round use of multiple dens. These adaptations are less effective against red foxes due to greater food and habitat overlap, the ability to pursue kit foxes into dens, and high potential for disease transmission. Thus, non-native red foxes pose a serious threat to kit foxes. Interactions between coyotes and red foxes may benefit kit foxes. In particular, interference competition by coyotes may limit the abundance and distribution of red foxes in the San Joaquin Valley. These interactions should be considered when evaluating management options (e.g., predator control).*

Introduction

Kit foxes are relatively small canids (1.7 to 3.0 kg) that occur in arid and semi-arid habitats of the southwestern United States and northern Mexico. The San Joaquin kit fox (*Vulpes macrotis mutica*) is a genetically distinct subspecies that historically occurred in the San Joaquin, Salinas, and Cuyama Valleys of central California. The abundance and range of this taxon have been significantly reduced, primarily due to habitat loss and degradation associated with agricultural, industrial, and urban development (FWS 1998). Fur harvests, predator control programs, and rodent control programs also may have contributed to observed declines. San Joaquin kit foxes persist as a metapopulation comprising three larger "core" populations and a number of smaller "satellite" populations. Current threats include continuing habitat conversion, rodenticide use, and interspecific competition. Much of the remaining habitat is fragmented, disturbed, and subject to competing land uses such as hydrocarbon production and water banking (FWS 1998). The San Joaquin kit fox was listed as Federally Endangered in 1967 and California Threatened in 1971.

Interspecific competition from other mammalian predators is an important factor affecting the remaining San Joaquin kit fox populations (Cypher et al. in press). Coyotes (*Canis latrans*), bobcats (*Lynx rufus*), gray foxes (*Urocyon cineareoargenteus*), badgers (*Taxidea taxus*), feral cats (*Felis domesticus*), and non-native red foxes (*V. vulpes*) all engage in interference and/or exploitative competition with kit foxes. Interference competition consists of direct mortality, harass-

San Joaquin kit fox (*Vulpes macrotis mutica*). Photograph by B. Moose Peterson/WRP.

ment, and spatial exclusion. Exploitative competition consists of overlap in use of potentially limited resources such as food items and dens. Competitive interactions with coyotes and red foxes have the greatest implications for San Joaquin kit foxes. Our objectives for this paper are to summarize the competitive interactions that occur between kit foxes, coyotes, and red foxes, and to assess the potential implications of these interactions for kit fox conservation and recovery.

Competitive interactions

Coyotes engage in both interference and exploitative competition with kit foxes. Coyotes have long been recognized as a significant cause of mortality for kit foxes (Seton 1925). At various study sites throughout the range of the San Joaquin kit fox, coyotes are the primary source of kit fox mortality for which the cause of death is identifiable (Hall 1983; Briden et al. 1992; Standley et al. 1992; Ralls and White 1995; Spiegel and Disney 1996; Cypher et al. 2000). This mortality indeed appears to be the result of competition rather than predation. Coyotes commonly do not consume the kit foxes they kill (Spiegel and Disney 1996; Cypher and Spencer 1998), although over half of kit foxes killed at one location were consumed during a period of low food availability (Ralls and White 1995).

Another effect of interference competition is spatial exclusion in which the presence of one species in an area results in decreased use of that area by another species. On a regional scale, kit foxes usually are absent or less abundant in more rugged terrain (areas with average slopes >5%). Kit foxes may have more difficulty eluding predators in rugged terrain. At one study site, kit foxes were abundant in rugged terrain when regional coyote abundance was low, but kit fox numbers quickly declined in this terrain as coyote abundance increased (Warrick and Cypher 1998). On a local scale, White et al. (1994) did not detect any temporal segregation among telemetered kit foxes and coyotes, indicating that areas were being used concurrently by the two species.

With regards to exploitative competition, coyotes consume some of the same food items consumed by kit foxes. Items commonly used by both species include black-tailed hares (*Lepus californicus*), desert cottontails (*Sylvilagus audubonii*), kangaroo rats (*Dipodomys* spp.), pocket mice (*Chaetodipus californicus*, *Perognathus inornatus*), California ground squirrels (*Spermophilus beechyi*), pocket gophers (*Thomomys bottae*), grasshoppers (Acrididae), Jerusalem crickets (Gryllacrididae), and beetles (*Eleodes* spp.). Overlap in resource use varies with the relative availability of different food items (White et al. 1995; Cypher and Spencer 1998), and therefore may not be a significant factor in all areas or all years.

The overall effect of competitive interactions between coyotes and kit foxes is not clear. Declines in kit fox abundance in some areas have been associated with increases in coyote abundance (Cypher et al. 2000; White et al. 2000). Coyote predation, however, does not appear to be the primary factor driving kit fox population trends. Instead, food availability as mediated by annual environmental conditions (particularly precipitation) appears to be the primary factor influencing kit fox population dynamics (White et al. 1996; Cypher et al. 2000). Predation by coyotes potentially could have a more significant impact on kit fox populations when food availability is low or when kit fox populations are small (particularly introduced populations).

Coyotes have historically occurred throughout the range of the kit fox, and kit foxes have evolved adaptive strategies for coexisting with coyotes. One such strategy is year-round use of dens. Kit foxes may have over 100 dens scattered throughout their home range, although on average about 12 dens are used by a given fox annually (Koopman et al. 1998). These dens facilitate escape from predators (White et al. 1994; Cypher and Spencer 1998). In addition, some amount of resource partitioning occurs between coyotes and kit foxes. Although food habits overlap, coyotes take higher proportions of leporids while kit foxes usually consume higher proportions of nocturnal rodents, particularly kangaroo rats and pocket mice (White et al. 1995; Cypher and Spencer 1998). Furthermore, there is some evidence that some degree of habitat partitioning may occur with kit foxes preferring more open areas with reduced shrub cover (White et al. 1995; Warrick and Cypher 1998).

Red foxes also engage in both interference and exploitative competition with kit foxes. Historically, native red foxes only occurred at high elevations in the Sierra Nevada and Cascade ranges in California, and were not found in any portion of the range of San Joaquin kit foxes (Grinnell et al. 1937). In more recent times, red foxes from the eastern United States have been introduced into lower elevation areas of California by humans for hunting and trapping, and have escaped from fur farms (Jurek 1992; Lewis et al. 1993). These highly adaptable, non-native red foxes have spread rapidly and have colonized many regions of California, including the San Joaquin Valley. The abundance of anthropogenic water sources in the San Joaquin Valley (e.g., canals, irrigated agriculture, stock ponds, and urban areas) likely has facilitated colonization by red foxes.

Red foxes are larger (3 to 8 kg) than kit foxes, and therefore dominate in interference competition. Red foxes are known to have killed radiocollared kit foxes on at least three occasions (Ralls and White 1995; Clark 2001). In addition, red foxes may be competitively displacing kit foxes in some locations (White et al. 2000). At one location, kit foxes primarily used areas not occupied by red foxes (Clark 2001). Finally, red foxes and kit foxes are closely related taxonomically which may increase the potential for disease transmission.

Being relatively close in size to kit foxes, exploitative competition may be more intense between the two species. Considerable overlap in use of foods was documented at one location in the San Joaquin Valley (Warrick and Clark unpublished data). Red foxes will use kit fox dens, excluding use by kit foxes. Red foxes have been observed in dens formerly used by kit foxes on a number of occasions at three different locations (White et al. 2000; Cypher et al. in press; Warrick and Clark unpublished data).

Red foxes constitute a serious threat to San Joaquin kit foxes. Adaptations of kit foxes that reduce competition with coyotes are less effective against red foxes. Red foxes can enter most dens used by kit foxes, making escape and avoidance more difficult. Resource partitioning may be minimal although some degree of habitat partitioning may occur between the two species. Red foxes may have difficulty colonizing kit fox habitat lacking in nearby water sources. Most red fox observations in the San Joaquin Valley are in relatively close proximity to sources of water.

Interactions between coyotes and red foxes can affect kit foxes. The presence of both coyotes and red foxes in a given area may act in an additive manner with regards to reducing food availability for kit foxes. Thus, exploitation competition between coyotes and red foxes may detrimentally affect kit foxes.

Red fox (*Vulpes vulpes*). Photograph by B. Moose Peterson/WRP.

Interference competition between coyotes and red foxes may also benefit kit foxes. Reduced abundance of red foxes attributable to coyotes has been documented in a number of locations (Dekker 1983; Voigt and Earle 1983; Major and Sherburne 1987; Sargeant et al. 1987). This reduction is a consequence of both direct mortality and exclusion. In a study in the San Joaquin Valley of California, 11 radiocollared red foxes were recovered dead. Of the nine mortalities for which the cause of death could be identified, all nine were killed by coyotes (Clark 2001). Coyotes have been suggested as a biological control strategy for red foxes in coastal areas of California where the foxes are preying on endangered California least terns (*Sterna antillarum browni*) and California light-footed clapper rails (*Rallus longirostris levipes*) (Jurek 1992). Coyotes also have been proposed as a means of reducing red foxes in the prairie pothole region of North America, thereby reducing red fox predation on duck nests (Sargeant and Arnold 1984).

Conservation implications

Due to anthropogenic ecosystem modification, non-native red foxes are expanding their range and increasing in abundance in California (Jurek 1992; Lewis et al. 1993). As described above, this species poses a potentially serious threat to kit foxes. Implementing effective control programs for red foxes would be extremely difficult for a number of reasons. In general, predator control programs can be costly as they usually must be implemented over large areas for multiple years (sometimes indefinitely) to achieve effective control, and are not popular with the general public even when conducted for the conservation of a rare, native species (Goodrich and Buskirk 1995). Poisons can not be used due to the threat to kit foxes as well as other non-target species. Trapping red foxes without also capturing kit foxes would be difficult due to the relatively close weights of the two species. The use of many types of trapping devices was banned in California in 1998. Shooting, possibly in conjunction with predator calling, may be possible, but could be difficult and costly to implement over large areas or near human-inhabited areas where red foxes commonly occur.

Coyote control is often suggested as an action that would benefit San Joaquin kit foxes. Similar to control efforts for red foxes, coyote control can be both difficult and controversial (Cypher and Scrivner 1992; Connolly and Longhurst 1975). There may be certain situations where limited coyote control might benefit kit foxes (e.g., reintroduction sites, smaller preserves and habitat blocks during periods of low food availability). Given the potential beneficial role of coyotes in limiting non-native red foxes, however, the implementation of coyote control should be carefully considered. Any reduction or limitation of red fox abundance achieved naturally through competitive pressure from native predators could significantly benefit kit foxes and would require no effort on the part of humans. Red foxes are rarely observed in areas where coyotes are abundant, even though kit foxes persist in these areas (Ralls and White 1995; Spiegel and Disney 1996; Cypher et al. 2000).

Ultimately, the strategy with the greatest potential for effectively conserving and recovering San Joaquin kit foxes will be to conserve and properly manage large blocks of habitat that are connected by movement corridors. This will facilitate larger kit fox populations that are more robust to losses from interspecific competition, and that are able to naturally recolonize areas where local extirpations of kit foxes may have occurred. Also, the fragmentation of kit fox habitat by anthropogenic disturbances promotes increased abundance of red foxes, which negatively impacts kit foxes. The conservation of large blocks of habitat is a paramount goal of the recovery plan for San Joaquin kit foxes and a number of other rare species that occur sympatrically with kit foxes (FWS 1998).

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Predation Management

Chemical Repellents and Other Aversive Strategies in Predation Management

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Abstract
*Chemical repellents and other aversive strategies are the core of non-lethal wildlife management. These strategies typically depend on irritation (pain), conditioning, or fear for their effectiveness, and none is universally successful. Thus, conditioned food aversions deter browsing and foraging by deer (*Odocoileus virginianus, O. hemionus*), but are less useful with predators, because killing, not consumption, is the behavior of interest. Broadly speaking, the utility of non-lethal strategies is affected by number and density of wildlife species, availability of alternative foods, palatability and novelty of treated items, and intensity of pain, sickness, or fear used to establish avoidance. Some of the most promising areas for successful predation management are those involving a combination of strategies tailored to a specific problem. For example, behavioral-contingent auditory and visual stimuli coupled with presentations of electric shock or momentary vibration (via telemetry collars) could provide an effective and unambiguous cue for withdrawal. Non-lethal methods, however, are rarely stand-alone technologies. More often, integrated strategies, involving both lethal and non-lethal methods, are required for effective predation management.*

Introduction
The survival or restoration of threatened and endangered species can depend on protection from predators (Witmer and Fall 1995; Witmer et al. 1996; Hecht and Nickerson 1999). Most of the relevant data for managing predation stem from research on the protection of livestock, crops, and commodities (Campbell et. al 1998; Fall and Jackson 1998). Deterring predators from prey is even more complex than protecting crops or other commodities because more is involved than food consumption (Fall 1990; Knowlton et al. 1999). Especially challenging is the development of non-lethal approaches. Demand for these strategies is increasing despite the fact that effective options remain virtually nonexistent. Repellents and other aversive techniques provide cases in point. If wildlife numbers are sufficiently high, or alternative foods are sufficiently scarce, repellents usually fail as a deterrent. Few demonstrably effective alternatives exist, and practical obstacles to the development of new materials are considerable. The present discussion will cover these topics by considering: mechanisms underlying the effectiveness of repellents and aversive agents, regulatory constraints that govern implementation of new methods, and the importance of employing multiple sensory modalities (i.e. visual and auditory cues, chemical and color cues) whenever non-lethal strategies are implemented.

Chemical repellents
Vertebrate chemical repellents are effective because they are irritating, cause sickness, or stimulate fear (Mason and Clark 1997). As a rule, these substances are most useful when they are applied directly to inert materials (e.g., prepared foods, fruits, grains, electrical wiring, irrigation hose; Werner et al. 1998). There is no good evidence that predators or other wildlife will avoid areas protected solely with border treatments. To illustrate the point, Renardine is commercially available for use with red fox (*Vulpes vulpes*) in the United Kingdom and

is being evaluated for use with coyotes (*Canis latrans*) in Canada (Martin and O'Brien 2000). (Mention of trade names and manufacturers is for identification only and does not imply endorsement by the authors or the U.S. Department of Agriculture.) The substance is bone tar oil dissolved in kerosene. The label advises that it should be applied liberally to pasture borders (on fence posts, etc.) to prevent predators from entering and attacking livestock. In testing with captive coyotes in the U.S., not only did Renardine fail to prevent entries into areas, but food adulterated with the material was eaten as rapidly as unadulterated food (Zemlicka and Mason 2000). This probably reflects the fact that sulfurous compounds in bone tar oil are attractive to coyotes (see **Fear** below).

Tastes, per se, are rarely effective feeding deterrents. While bitter and acidic substances can initially reduce the consumption of treated materials slightly (cf. Nolte et al.1994b), intake typically returns to baseline within a short period of time. Products that claim effectiveness solely because of a "bad" taste are doing so largely because humans find the taste repulsive. Some species of interest, including obligate carnivores such as the Felidae, have taste sensitivities that are greatly different than humans, including insensitivity to salt and sweet (Beauchamp and Mason 1991). In particular, products that contain denatonium derivatives (compounds very bitter to humans) are ineffective repellents, almost regardless of species (although bears, *Ursus horribilis* and *Ursus americanus*, may avoid denatonium in foods; G. Witmer, National Wildlife Research Center, personal communication) or method of application (e.g., topical spray, incorporated into products). Canids, in particular coyotes, are markedly insensitive to denatonium benzoate (Mason and McConnell 1997). Despite this lack of demonstrated utility, new veterinary and wildlife control products containing denatonium derivatives as the active ingredient are regularly offered for sale.

Irritation

Among the three types of chemical repellents, substances that cause sensory irritation or pain (the same neural receptors are involved) usually are most effective. This is because sensory pain leads to immediate withdrawal, independent of learning. Such avoidance does not habituate (diminish) for as long as the irritating stimulus is present. Moreover, taxonomic differences in irritant sensitivity between birds and mammals (Clark 1998a) permit the development of repellents with some degree of selectivity (e.g., Norman et al. 1992).

For mammals, strong irritants include capsaicin and capsicum oleo resins (i.e. the active ingredients in 'hot sauce' preparations; Norman et al. 1992), volatile chemicals like mustard oil (allyl isothiocyanate) and ammonia (Budavari et al. 1996), and non-volatile substances including astringent tannins such as quebracho (Swihart 1990). None of these substances repel birds (Mason and Clark 1997). Unfortunately, when irritant chemicals dissipate (e.g, by evaporation or photolysis), there is usually an immediate resumption of the unwanted behavior (Mason et al. 1985). A more important drawback is that intrataxonomic differences in irritant sensitivity are small. There are no known irritants that affect only some mammalian species but not others (e.g., coyotes but not sheep, *Ovis aries,* or humans).

This is not to say that irritants are completely ineffective deterrents to predation. Irritants can be effective when prey are completely infused. This strategy is relatively common among insects (e.g., Wickler 1968), amphibians, reptiles (e.g., Schmidt et al. 1989), and occasionally birds. One example is the Pitohui bird (*Pitohui dichrous*) that stores and uses toxicants from insects it ingests (Dumbacher et al. 1992). However, in the absence of complete infusion, repellency is easily circumvented. This explains the ineffectiveness of topically applied irritants as deterrents to predation. For example, sheep fitted with collars containing capsicum oleo resin were killed as readily as sheep without collars, despite the fact that attacking coyotes punctured collars and were exposed to high concentrations of the irritant (Burns and Mason 1996).

Fear

Sulfur compounds and volatile ammonium soaps of higher fatty acids induce what humans describe as fear in herbivores (Milunas et al. 1994). These substances underlie the effectiveness of predator urines and many commercial preparations used to repel browsing deer, rabbits (*Sylvilagus floridanus*), and rodents (Nolte et al. 1994a; Lewison et al. 1995; Mason et al. 1999). Typically, substances that frighten herbivores attract obligate carnivores and many omnivores (Kimball et al. 2000). There are no published data consistent with the belief that urine samples from one predator are actively avoided by other predators.

A disadvantage of fear-inducing chemicals is that animals readily habituate to their presence. The rate of habituation is largely dependent on the degree to which the chemical cue is associated with a risk of predation. When risk is low, habituation is rapid. Cues may even become attractive. For example, wolf (*Canis lupis*) urine applied as a repellent along roadways in winter can attract moose (*Alces*

alces) and other ungulates that learn to associate the odor with the presence of road salt (T. Sullivan, personal communication).

Sickness (conditioned or learned avoidance)

When the ingestion of novel flavors or tastes by mammals or distinctively colored foods by birds is followed by sickness, a learned avoidance usually results (Beauchamp and Mason 1991). This effect is variously called conditioned (or learned) taste, food, or flavor avoidance (CA). CA can occur after a single aversive experience, particularly when the intensity of sickness is great and the taste, food, or flavor is new (Pelchat et al. 1983). As with other chemical repellents, substances that elicit CA are classified as pesticides by regulatory agencies, which typically require extensive data sets for registration prior to commercial use.

An extensive literature on theory, use and applications of CA is available (e.g., Riley and Tuck 1985). CA is the mechanism underlying the utility of commercial bird repellents containing methiocarb or anthraquinone (Conover 1982; Reidinger and Mason 1983), and commercial deer, rabbit, and rodent repellents containing thiram or ziram (Thomson 1995). CA using lithium chloride or estrogens to induce sickness has been investigated as a way to: reduce depredation by coyotes, resolve nuisance feeding by black bears (*Ursus americanus*; Ternent and Garshelis 1999), and curtail egg predation by raccoons (*Procyon lotor*), skunks (*Mephitis mephitis*), mongooses (*Herpestes nyula*), and ravens and crows (*Corvus spp.*; e.g., Nicolaus and Nellis 1987; Nicolaus et al. 1982, 1983; Semel and Nicolaus 1992). While evidence suggests that CA can be used to successfully manage nuisance complaints and egg depredation under some conditions, no lithium chloride- or estrogen-based method has been registered by the U.S. Environmental Protection Agency.

Gustavson (1974) conducted the first studies of CA as a management strategy with coyotes. His initial data were promising, generating considerable interest in the approach. Some investigators have reported success in preventing predation (Gustavson et al.1974; Ellins and Martin 1981; Gustavson et al.1982; Forthman-Quick et al. 1985a, 1985b), while others have reported failure (Conover et al. 1977; Burns 1980; Burns and Connolly 1980; Bourne and Dorrance 1982; Burns 1983). Two large field trials conducted in Canada generated opposite results (Bourne and Dorrance 1982; Gustavson et al. 1982). Ten years after the most extensive field trial (Gustavson et al. 1982; Jelinski et al. 1983), survey responses of 52 participating ranchers indicated that while 54% initially considered lithium-chloride baiting "successful" or "somewhat successful," only one participant continued to use it (Conover and Kessler 1994). While no explanation for differences among studies is completely accepted, most arguments have focused on methods and experimental design (Bekoff 1975; Gustavson et al. 1975; Sterner and Shumake 1978; Horn 1983; Forthman-Quick et al.1985b; Conover 1997). Although CA may be a useful tool in some situations, its utility in predation management appears quite limited. Gustavson acknowledged that coyotes often resumed killing sheep shortly after conditioning, and ". . .once a coyote becomes a confirmed sheep-killer, perhaps it will be necessary to remove it from the population" (Gustavson et al. 1978). This could reflect the possibility that while CA may affect consumption of prey, the generalization of learning from consumption to killing is weak. There are no data on the use of CA to protect big game from predators, but considerable data relating to attempts at livestock protection provide little promise of potential utility.

Mechanical and electronic devices

There are many parallels between chemical repellents and mechanical or electronic devices that provoke fear or deliver painful or irritating stimulation. Scarecrows and their modern analogues have been widely examined with both birds and mammals. Studies typically report rapid habituation and variability among species and settings (Koehler et al. 1990; Bomford and O'Brien 1990). Nevertheless, at least for depredation management, there are promising results in certain situations. For example, Linhart et al. (1984) found that combinations of battery-operated strobe lights, sirens, and high frequency horns, placed on the edges of sheep pastures or bedgrounds and activated for short irregular intervals during night and early morning, stopped predation for 27 to 136 nights. Coyotes apparently remained active around the peripheries of the test pastures, but habituation to the devices was delayed by the irregular patterns of activation. More recently, using animal-activated or demand-performance frightening devices (Stevens et al. 2000), Shivik and Martin (in press) showed that motion-activated sirens were more effective than random sirens in delaying habituation by captive coyotes. Limited field trials of behavior-contingent, multistimulus (light and sound), systems to deter wolf predation are ongoing and appear promising (Shivik and Martin in press). Devices are activated when radio-collared wolves approach livestock production areas.

Application of a brief, non-lethal electric current has been widely studied as a means of deterring predators (Linhart et al. 1982; Sargeant and Arnold 1984). Linhart et al. (1976) found that coyotes fitted with collars that provided a contingent electric shock stopped attacks on rabbits for several months. More recently, commercial electronic dog training collars that deliver a mild static electrical discharge have been successfully tested as a deterrent to captive coyotes attacking sheep (Andelt et al. 1999). Manual activation of collars stopped attacks in progress and greatly reduced the probability of subsequent attacks. After one to three training bouts, coyotes avoided or retreated from sheep in tests four months after initial sessions. Shivik and Martin (in press) are testing similar collars, triggered by radio signals, with wolves. Animals wearing modified radio telemetry collars self-activate the static discharge when they approach within biting distance of a calf, providing an unambiguous cue for withdrawal. Collars utilizing momentary vibration (a sensation humans perceive as similar to static discharge) are also commercially available for dog training and could have similar application. Shivik and Martin (in press) describe efforts to develop auto-attaching collaring systems that utilize breakaway snare technology, which, if successful, would substantially reduce the cost of the method.

Ecological and behavioral consequences

At issue is effective adaptation of agricultural methods to endangered species protection, while avoiding negative ecological consequences (e.g., affecting wildlife other than target species) often associated with pest control efforts. Much attention has focused in the past on unintended consequences on non-target species of broad-spectrum pesticide use. However, all proposed methods of pest control, including those presumed to be non-lethal, must be carefully examined for effectiveness in specific situations, selectivity, and potential environmental and behavioral effects. For example, fences placed to exclude a predatory species may interrupt movement patterns or block migration routes of another. Selectivity for particular problem species, or individual animals causing predation, and avoidance of problems with primary or secondary effects on non-target animals are desirable features for all animal control tactics, especially those involving chemical applications. Classic examples of past successes in agriculture in finding such alternative approaches include: (1) replacing dynamiting of vampire bat (*Diphylla ecaudata*) caves to control paralytic cattle rabies with vampire bat-selective toxicant treatments (Mitchell 1986; Lewin 1986); and (2) replacing poisoned carcass bait stations as a method of coyote predation control with selective methods, such as Livestock Protection Collars (Connolly et al. 1978; Connolly 1993), den hunting (Till and Knowlton 1983), and aerial hunting (Connolly and O'Gara 1988; Wagner and Conover 1999; and Mason et al. in press).

Similarly, control programs aimed at protecting endangered species from predation must be planned strategically for specific areas to assure they achieve desired objectives. For example, Conner et al. (1998) found no relationship between annual coyote removal and levels of coyote predation on sheep on a California agricultural experiment station where non-lethal methods and non-selective coyote removal had not achieved desired reductions in predation after several years of effort. Sacks et al. (1999) found that adult territorial coyotes responsible for sheep killing in the area were less vulnerable to these control tactics than coyotes not involved in livestock predation. Ultimately, removal of specific depredating individuals by shooting and Livestock Protection Collars greatly reduced predation (Blejwas et al. in press). Livestock Protection Collars, however, have recently been banned in California by a ballot initiative, creating a need for development of effective alternative methods. In another situation, probably common in both agriculture and conservation predation management applications, feral cat (*Felis catus*) control efforts had to be implemented following a highly successful rodent control effort to protect nesting Dark-rumped petrels (*Pterodroma phaeopygia*) from predation in the Galapagos Islands (Cruz and Cruz 1987). When black rats (*Rattus rattus*) were removed as the primary predator, cats, which had been subsisting on rats, switched prey, diminishing initial increases in nesting success achieved by the rat control program. Although non-lethal tactics, if they become available, would be expected to be more benign and specific in such situations, both legal and ethical considerations require their careful assessment before implementation takes place.

Regulatory constraints

Obstacles to development of new materials and methods are considerable. These include a variety of constraints imposed by regulatory agencies. Even when new repellent technologies are uncovered, the path to commercial availability is long and can be very expensive (Fagerstone and Schafer 1998). For example, methyl anthranilate is one of two new bird repellents to become commercially available in the United States during the past 25 years. This natural substance is GRAS-listed (generally recognized as safe) with the U.S. Food and Drug Administration and it has been widely used as a grape flavoring in human and animal feeds since the turn of the century. Despite these facts, registration efforts to permit spraying of methyl anthranilate on turf to deter grazing Canada geese (*Branta canadensis*) took five years and cost $5 million (P.J. Voigt, R. J. Advantage, Inc., personal communication). While research on non-lethal methods for agricultural applications has been relatively well-supported, support needed to develop and evaluate methods for endangered species applications has been more elusive, usually coming in the form of small grants that cannot cover long term costs of developmental research to meet regulatory requirements.

Combinations of stimuli

Beauchamp (1997), among others, has suggested that the most effective strategy in development of repellents may be use of combinations of stimuli. The evidence is consistent with this suggestion (Clark 1998b). Thus, a mixture of capsaicin (irritation), thiram (sickness-based conditioned avoidance), and Big Game Repellent (sulfur-based fear) is a more effective deterrent to browsing white-tailed deer (*Odocoileus virginianus*) than any of these substances alone (M. Richmond, U.S. Geological Survey, Cornell University, personnel communication). Likewise, mixtures of methiocarb (sickness-based conditioned avoidance) and methyl anthranilate (irritation) are more effective than either substance alone. Cinnamaldehyde (Crocker and Perry 1990) and d-pulegone (Mason 1990) are both broadly effective vertebrate (bird and mammal) repellents that have both irritant and sickness-inducing effects.

Conclusion

Development and application of ecologically sound and effective repellents is dependent upon a knowledge of the sensory *Umwelt* of the species in question (von Uexkull 1934). Even when aversive strategies can be successfully applied, their continued utility will likely depend on application of other techniques in a mosaic of management strategies designed to meet requirements of a specific location, time, and context. Indeed, selective removal of wildlife (either in terms of local population suppression or removal of specific individuals) may often be prerequisite for effective implementation of non-lethal alternatives. For this reason, integrated strategies that incorporate both lethal and non-lethal methods will often be the most logical course for effective predation management. The high cost of development and application of alternative technologies for endangered species applications and the highly specific minor-use markets for such products, which limit private industry interest in the problem, present challenges to the emergence of new technologies needed to help assure effective recovery of species threatened by predation.

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Effectiveness of Livestock Guarding Animals for Reducing Predation on Livestock

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Abstract
*Predation is a major problem faced by domestic sheep (*Ovis aries*) and goat (*Capra hircus*) producers in the western United States. Producers have been incorporating livestock guarding dogs (*Canis familiaris*), llamas (*Lama glama*), and donkeys (*Equus asinus*), which appear to be effective in reducing these mortalities. The increased use of guarding animals to mitigate predation on livestock may reduce animosity toward predators and result in more positive attitudes toward the conservation of carnivores.*

Introduction
Predation by coyotes (*Canis latrans*), domestic dogs (*Canis familiaris*), mountain lions (*Felis concolor*), black bears (*Ursus americanus*), foxes (primarily *Vulpes vulpes*), eagles (primarily *Aquila chrysaetos*) and bobcats (*Felis rufus*) has been a major problem faced by domestic sheep producers, especially in the western United States (National Agricultural Statistics Service 2000). Pearson (1986) reported that coyotes, the primary predator of sheep, killed an average of one to 2.5% of adult ewes and four to 9% of lambs annually in the 17 western states. Several methods, including the use of livestock guarding dogs, llamas, and donkeys, have been used to reduce these mortalities (Andelt 1996). This paper reviews several studies of the effectiveness of livestock guarding animals and speculates on the implications for conservation of carnivores.

Livestock guarding dogs
Livestock guarding dogs have been used in the United States since the early 1970s to protect sheep (*Ovis aries*) and goats (*Capra hircus*) from predators. Most guarding dogs are members of breeds that have been developed selectively in Europe and Asia to protect livestock from bears (*Ursus* spp.) and wolves (*Canis lupus*). The most common breeds used for guarding livestock in the United States are Great Pyrenees, Akbash, and Komondor, whereas Anatolian, Kuvasz, Maremma, and Shar Planinetz are less common (Green and Woodruff 1988; Andelt and Hopper 2000). Most guarding dogs weigh 34 to 45 kg and are at least 64 cm at the shoulders. Successful guarding dogs are trustworthy (will not harm sheep), attentive to sheep, and aggressive toward predators (Coppinger et al. 1983). These traits are "instinctive"; they develop in most dogs with proper handling and minor training (Andelt 1995).

Andelt (1985) reported that guarding dog pups cost an average of $240 in Kansas and Green et al. (1984) reported pups cost an average of $331 and $458 (depending on breed) in the western United States. Annual maintenance fees (food, veterinary care, and miscellaneous costs) averaged $235 to $250 (Green et al. 1984; Andelt 1985).

The National Agricultural Statistics Service (2000) reported that 28% of sheep producers in the United States and 23% of producers in Colorado used guarding dogs to protect sheep during 1999. Andelt and Hopper (2000) reported that the percentage of sheep with guarding dogs in fenced pastures and on open range in Colorado increased from 7% in 1986 to 65% in 1993. They also indicated that producers primarily with large numbers of sheep have incorporated guarding dogs.

Sheep producers in Colorado who did not use livestock guarding dogs lost 5.9 and 2.1 times greater proportions of lambs to predators than producers who had dogs in 1986 and 1993, respectively (Andelt and Hopper 2000). Mortality of ewes to predators and lamb mortality on open range decreased more from 1986 to 1993 for producers who obtained dogs between these years compared to producers who did not have dogs. Thirty-six producers in North Dakota reported guarding dogs reduced predation on sheep by 93% (Pfeifer and Goos 1982). Producers in Colorado indicated that guarding dogs greater than nine months of age saved more time in sheep management than the amount of time spent feeding and working with each dog (Andelt 1992). Overall, guarding dogs are a cost-effective means of reducing predation (Green et al. 1984; Andelt and Hopper 2000).

Disadvantages of guarding dogs include: some dogs not staying with or harassing sheep; some dogs, especially Komondors, being overly aggressive toward people (Green and Woodruff 1988; Andelt 1992); and the dogs can be subject to injury and premature death. Many of the disadvantages are relatively uncommon. Most producers surveyed feel strongly that the advantages of their dogs far outweigh the disadvantages.

Green and Woodruff (1988) reported that the rate of success in protecting livestock from predators did not vary among Great Pyrenees, Komondor, Akbash, Anatolians, Maremma, and hybrids, nor was the rate of success different among males and females or intact and neutered dogs. Dogs that were reared with livestock from at most two months old, however, had a significantly higher rate of success than dogs that were older than two months when placed with livestock. Andelt (1999) reported that ratings of the effectiveness of guarding dogs by producers using one breed of dog in Colorado did not differ among breeds, but producers that used multiple breeds rated Akbash more effective than Great Pyrenees and Komondors.

Llamas

Llamas have been used to deter predation primarily by coyotes, red foxes (*Vulpes vulpes*), and dogs since the early 1980s. The National Agricultural Statistics Service (2000) reported that 13% of the sheep producers in the United States used llamas to protect sheep from predators during 1999. Llamas are naturally aggressive toward coyotes and dogs. Typical responses of llamas to coyotes and dogs are being alert, alarm calling, walking to or running toward the predator, chasing, kicking, or pawing the predator, herding the sheep, or positioning themselves between the sheep and predator.

Franklin and Powell (1993) surveyed 145 producers that used llamas, primarily in Montana, Wyoming, Colorado, California, and Oregon. Most producers used one gelded male with 250 to 300 sheep in 250 to 300 acre pastures. Nearly all llamas were not raised with sheep and were not trained to guard sheep. One llama was more effective than multiple llamas for deterring predation; the effectiveness of gelded males, intact males, and females was similar. Producers reported, however, more problems with intact (25% of 61 intact males) than gelded males (5% of 135 gelded males) attempting to breed with ewes. Sheep that were introduced to llamas in corrals initially sustained lower mortality than those introduced in pastures. The success of llamas was not related to age when the llama was introduced, age of llama (after one or two years old) when guarding, if lambs were present or absent when the llama was introduced, or between open and covered (forested, shrub lands, gullies, ravines, etc.) habitat.

Franklin and Powell (1993) reported that gelded male llamas cost \$700 to \$800, whereas intact males were about \$100 less. Most producers reported that daily care for llamas was the same as for sheep and that no special feeds were provided. Average annual expense was \$90 for feed (not including pasture) and veterinary costs were about \$15.

Franklin and Powell (1993) reported that 21% of ewes and lambs were killed annually before acquiring a llama and 7% afterwards. Meadows and Knowlton (2000) reported that producers with llamas lost significantly fewer sheep ($n = 42$) to predators than producers without llamas ($n = 128$) during the first year of use, but sheep mortalities did not differ between producers with ($n = 35$) and without ($n = 32$) llamas during the second year in Utah.

Donkeys

The National Agricultural Statistics Service (2000) reported that 9% of sheep producers in the United States used donkeys (*Equus asinus*) to protect sheep from predators during 1999. Donkeys apparently have an inherent dislike for dogs and other canids. They will bray, bear their teeth, run and chase, and attempt to bite and kick an intruder (Green 1989).

Donkeys apparently are most effective in small open pastures or where sheep graze together. Green (1989) and Walton and Feild (1989) recommended using only one jenny or gelded jack per pasture because two or more donkeys often stay together instead of being with the sheep. Intact jacks generally are too aggressive around sheep. Donkeys generally should be allowed four to six weeks for bonding with sheep before they are used to deter predators. Donkeys should be removed during lambing because they might trample lambs or disrupt the ewe-lamb bond. Green (1990) recommended challenging a donkey with a dog to test its response to canids; donkeys that are not aggressive should not be used.

Walton and Feild (1989) reported that the average purchase price per donkey was \$144. They also reported that average annual upkeep per donkey was \$66.

Bonding sheep and goats to cattle

Bonding young sheep to cattle (Anderson et al. 1987; Hulet et al. 1987) and goats to sheep that have been bonded to cattle (Hulet et al. 1989) has reduced coyote predation. This technique has not been readily adopted by sheep producers, possibly because of the additional labor and expense involved with bonding sheep and goats to cattle.

Relative effectiveness of guarding animals

Benefits of using guarding animals include a decrease or elimination of predation, reduced labor to confine sheep at night, more efficient use of pastures for grazing, reduced reliance on other predator control techniques, and a greater peace of mind. A comparison of surveys where producers reported the average annual value of sheep saved per guarding animal suggests guarding dogs, compared to llamas, saved more sheep from predators (Table 1). Guarding dogs and llamas have been rated as more effective than donkeys for deterring predation. The above comparisons should be interpreted conservatively because guarding dogs, llamas, and donkeys were not surveyed in the same studies nor under the same sheep management conditions.

Advantages of donkeys and llamas over guarding dogs include: less prone to accidental death; longer-lived; do not require special feeds; stay in the same pasture as sheep; apparently do not need to be raised with sheep; more compatible with other depredation control techniques such as traps, snares, M-44s, and livestock protection collars; and donkeys are cheaper than guarding dogs. Guarding dogs apparently have some advantages over donkeys and llamas. Guarding dogs deter predators in fenced pastures and on open range, whereas llamas and donkeys appear most effective in fenced pastures smaller than 300 acres. Guarding dogs are effective in deterring bear and mountain lion predation (Green and Woodruff 1989; Andelt and Hopper 2000), whereas some donkeys (Green 1989) and possibly llamas are afraid of bears and mountain lions. Guarding dogs also appeared successful in protecting cattle from wolf predation (Coppinger et al. 1988), and were fairly effective in keeping wolves and black bears from carrion feeding sites in Minnesota (Coppinger et al. 1987).

Several methods, including livestock confinement, disposal of livestock carcasses, herders, fencing, frightening devices, trapping, snaring, sodium cyanide ejectors, "denning" (locating the dens of depredating coyotes and killing the pups and/or adults), aerial hunting, ground shooting, hunting with decoy dogs, livestock protection collars, and poison baits have been used to reduce predation on livestock (Andelt 1996). Poison baits were withdrawn from use in 1972 (Andelt 1996), and use of some methods such as trapping, snaring, sodium cyanide ejectors, gas cartridges for denning coyotes, and livestock protection collars have been restricted or eliminated by ballot initiatives in some states such as Arizona, California, Colorado, and Massachusetts (Manfredo et al. 1999). The public also has rated guarding animals as more acceptable than most other techniques for reducing predation (Arthur 1981; Reiter et al. 1999). Thus, guarding animals are one of the few remaining successful techniques, in some states, that livestock producers can use to mitigate predation.

Green and Woodruff (1989) and Green et al. (1993) reported that guarding dogs repelled black bears and grizzly bears (*Ursus arctos*) during most encounters, and Andelt and Hopper (2000) reported that producers with guarding dogs, compared to producers without guarding dogs, sustained fewer ewe and lamb mortalities to black bears. Thus, guarding dogs may offer

Table 1. Average annual value of sheep saved from predation by each livestock guarding animal and ratings of effectiveness of guarding animals as reported in various studies.

Factor	Guarding dogs	Llamas	Donkeys
Value of sheep saved	\$3,836[a] \$2,506[c] \$3,733[c]	\$1,253[b]	
Ratings of effectiveness			
very effective	71%[d]		
excellent and good	95%[e] 84%[c]		20%[g]
very effective and effective		80%[b] 90%[f]	
good and fair			59%[g]

[a]Green et al. (1984)
[b]Franklin and Powell (1993)
[c]Andelt and Hopper (2000)
[d]Green and Woodruff (1988)
[e]Andelt (1992)
[f]Meadows and Knowlton (2000)
[g]Walton and Feild (1989)

some potential for reducing grizzly bear predation on livestock, which may result in conserving more bears.

Many of our carnivores are found on private lands and in areas where livestock are grazed on private and public lands. Producers need to have successful techniques for resolving predation on livestock. Without successful methods, predation on livestock will increase and animosity by livestock producers toward predators likely will also increase, which will result in fewer attempts to conserve carnivores. Without successful techniques, producers also may sell their land for other uses, such as subdivisions, which are less conducive to carnivore conservation. Thus, guarding dogs, llamas, and donkeys should be thought of as valuable additions to the toolbox of management practices that reduce predation on livestock. In addition, use of guarding animals may result in enhanced conservation of carnivores.

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Opinions

Historical Attitudes and Images and the Implications on Carnivore Survival

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Abstract

*This paper examines how mythological images and historical attitudes emerge and influence our interactions with different predator species, such as the grizzly bear (*Ursus arctos*), cougar (*Puma concolor*), lynx (*Lynx canadensis*), wolf (*Canis lupus*), coyote (*Canis latrans*), and raven (*Corvus corax*). The author will compare the relationship between humans and carnivores, and how attitudes and beliefs have impacted different predator species. Do people regard certain carnivores as more fierce, dangerous, or problematic? Is there more animosity and disparate levels of hostility or tolerance toward the different carnivores? Have these attitudes influenced concepts and ethics applied to wildlife management? How is the value of predators measured, considered or applied? Can understanding the different perceptions help resolve complicated issues, such as reintroduction, critical habitat, depredation conflicts, animal damage control, and management? The author believes scientific knowledge is not enough to achieve acceptance of carnivores. The purpose of this inquiry will be to discover if knowledge and education can develop understanding and tolerance of all predators, and thus enhance the commitment to co-exist with carnivore species*

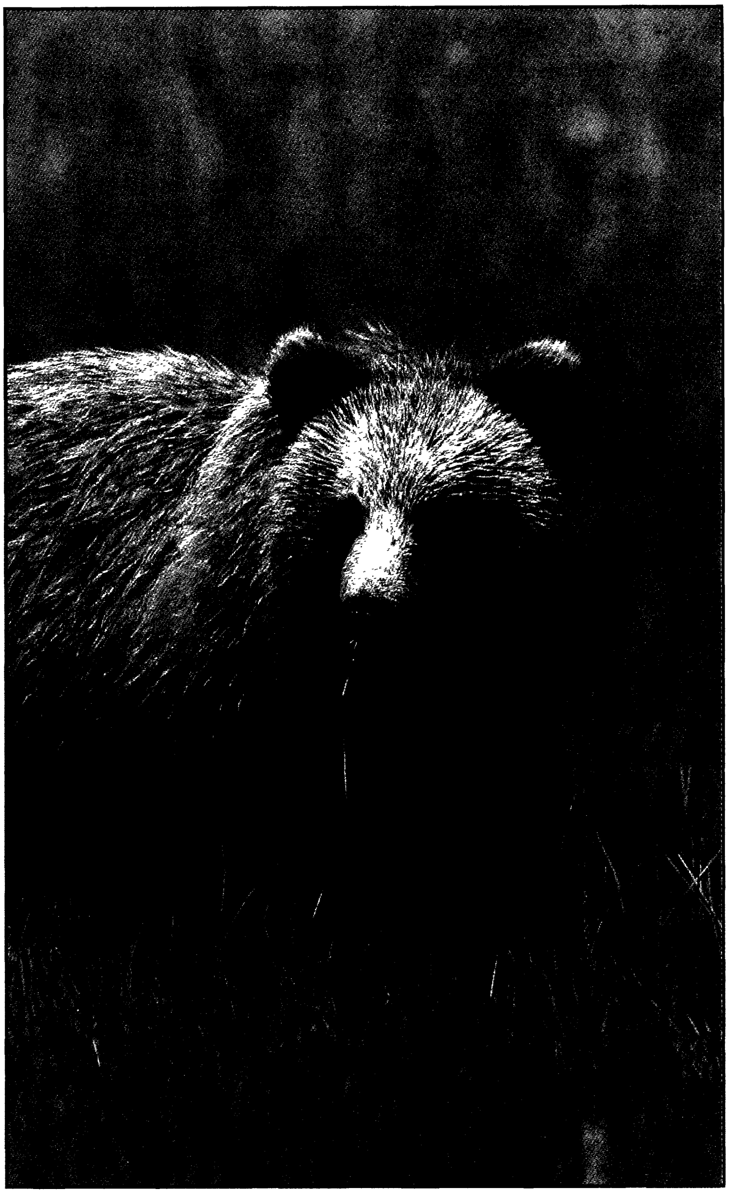
Mythical images and historical attitudes may still influence human interactions with carnivores such as the grizzly bear (*Ursus arctos*), cougar (*Puma concolor*), lynx (*Lynx canadensis*), wolf (*Canis lupus*), coyote (*Canis latrans*), and raven (*Corvus corax*). The earliest historic records, creation stories, and fables were examined. Notably, at the advent of agriculture, Akkadian literature delineates the split between humans and nature, and the god Ea predicts that from this time forward, nature will be hostile to man. He asserts that lion, wolf, famine and plague will not be removed from humankind's dilemmas, but provided "to rise up and cut the people low" (Gardner1984). This division between wilderness and the tamed domestic lands that humans seek does seem to have taken place, and remains a current conflict among predators, wilderness advocates, and ranchers.

Historical attitudes and cultural beliefs have impacted many different predator species. From Viking bearskin wearers to the arrival of humans in the New World, most carnivores continued to meet the same fate. Most of the legends revolve around the fear of wilderness, the idea of good and bad animals, and the need to remove all that stands in the way of progress. Species with fang and claw that hunted the good prey (which humans wanted for themselves) are traditionally described as the bad animals. The predator is reduced to the status of marauder and thief, and hence subjected to extermination.

Do people regard certain carnivores as more fierce, dangerous, or problematic? Some species have been described with more vivid hostile imagery that did not reflect actual biological evidence of their threat to humans, but had more to do with deeply rooted bias and mythological symbolism.

The cougar is commonly described as the coward. Theodore Roosevelt described a cougar he had treed as "the big horse-killing cat, the destroyer of the deer, the lord of stealthy murder, facing his doom with a heart both craven and cruel" (Danz 1999; Worster 1977; Roosevelt 1913). The cougar is repeatedly described as a cunning, merciless, and sneaky cat. The portrayal is made with little concern that a trait like stealth is a necessary ability to survive as a cougar, and has little to do with intentions of mere cruelty for cruelty's sake. Predation is often perceived as murder, not as a pursuit of food.

The bear, however, is characterized more in terms of admiration, combining descriptions of its sav-



Grizzly bear (*Ursus arctos*). Photograph by B. Moose Peterson/ WRP.

ageness with the animal's almost human dignity. Descriptions such as "unbelievable size of the brute" and "lordly intelligence" (Young 1980) serve as examples of the bear's more dignified status among the carnivores. The name Old Ephraim, taken from the Biblical patriarch of noble lineage, is honorably bestowed upon the bear.

Other cultures, such as the Native Americans, viewed bears as half human, for the bear walked the same trails, fished for the same salmon, dug for roots, harvested berries, seeds, and nuts, caught meat, and could stand upright like humans. While the grizzly bear was feared as dangerous, it was considered a powerful shaman of the animal world. The depiction of the bear is not infamous, though it was considered dangerous for humans to encounter. Keith Johnson, a master guide, actually admires the alleged viciousness of the bear, saying, "A grizz or a brown is a bear, but a black is a dog. They don't even belong in the same category. I don't have much respect for a black bear as a vicious animal" (Kaniut 1983).

Native Americans admired the perseverance and hunting prowess of the wolf. These traits were despised by wolf hunters who consistently describe the wolf as ruthless, treacherous, and a cruel demon that exists in spite of man's will. Myths also accuses the wolf of not only devouring human bodies, but also devouring the soul for the devil. The "yellow eyes as the lamps of the devil" is apparently still an image that persists (Berg unpublished manuscript). A Minnesota state legislator recently exclaimed, "wolves are serial killers" (McAuliffe 1999), and a sign held at hearings at declared, "Pedophiles and wolves kill children" (Sandstrom 1998)

The coyote is described as a tenacious varmint (the last of the outlaws), detested for its intelligence, and defined as crafty or the mythological Trickster. As prophesied for the wolf, fears are exaggerated about predators, and people proclaim that it is only a matter of time before a coyote is going to jump onto someone's deck, grab their child, and run off. (Interestingly, a proposed black bear hunt in New Jersey was canceled after wildlife organizations protested. But there was minimal outcry over the coyote hunt that was also proposed.) People see the bear as a cuddly animal worth protecting because of their teddy bear experience. Conversely people fear the wolf and coyote because of the negative childhood stories about these canines. Unfortunately the Teddy Bear image causes problems because people feel they can treat real black bears as the cute stuffed bear they have at home. This ultimately puts the people and bears in danger.

Mythical depiction has not spared any of these species. Even the lordly bears or monarchy of the big cats were considered killers in the way of progress. The value of a varmint has traditionally been calculated as being worthless. The historic, negative attitudes prevail; throughout history, humans have not progressed, learned from mistakes, improved methods, nor become enlightened on the value of predators. Unfortunately, predator programs and proposals of the year 2000 are not much improved from those of the past. Society seems to be in an era of rational compromise that may be headed in reverse.

Perhaps it has been forgotten that conservation evolved from the progressive political movement which valued nature, lands, trees, and wildlife as a commodity to be used for human economic success. The movement set forth the goals of more efficient management and maximum utilization of natural resources, not conservation or the preservation of biodiversity. Between 1901 and 1909, as the progressive ideology developed, it soon included an extensive program designed to exterminate predators and make America safe from their thieving presence. The rescue of nature from these killers would no longer be carried out just by pioneers, ranchers, and lone wolf hunters. The new Bureau of the Biological Survey (BBS), as part of the Department of Agriculture, would now reinforce the ferocity against predators. Thus, a major focus of conservation in the early years was the well-funded, staunch effort to deliberately eradicate predators. Today, this practical technique of killing any creature that does not suit humans has been renamed flexible management.

It is important to remember that the tremendous loss of our nation's predators was not an accidental byproduct of progress, but a well-organized extermination. The BBS was also a center for state bounty information, which published pamphlets on the habits of predators, and bulletins about the best trapping scents and poisons with which to kill the animals. In 1915, Congress appropriated $125,000 directly to fund this war on predators. This amount kept increasing; thus in 1950, $1,098,00 was allocated, and by 1970, $3,267,000 (Advisory Committee on Predator Control 1972). The program was defined by Jenks Cameron of the BBS as "suppressive warfare against undesirable, injurious wildlife," and "the protection and encouragement of wildlife in its desirable and beneficial forms" (Worster 1977).

In these modern times, most of us have thought that the idea of poison, especially 1080, would not loom again over wildlife, or that bounties—repeatedly proven to be ineffective, fraudulent, and wasteful —would not be promoted. However, Wyoming recently proposed a $1000 bounty, fortunately vetoed by the governor. But in Minnesota, Governor Jesse Ventura just signed a law that includes a $150 bounty, now called a predator payment. The new management law also allows discretionary killing by ranchers.

People are familiar with legislators' negative views on predators, but what about the modern scientists? Have their data redefined the value of the varmint by its ecological place in the ecosystem? It is quite disappointing to see recent scientific papers calculating monetary formulae to promote the view that it is cost efficient to only maintain a population of 1400 wolves in the wilderness and semi-wilderness of Minnesota (e.g., Mech 1998). The average annual cost, according to these calculations, would be $86 per wolf of 1438 wolves living primarily in the wilderness, and an additional $197 per wolf outside the designated area (surplus animals, dismissed as unnecessary extras).

Unfortunately, this study wrongly assumed the cost of the compensation program were payments made to verified wolf depredation cases only. In fact, the compensation granted by state conservation officers is not based on scientific verification, and often conflicts with the Federal examination findings. Considerable payments were made to cases that were not wolf-related or verified, and some were documented by the Federal program as outright fraud.

The thesis also wrongly assumes that most or all wolves outside of the designated area contribute to depredation. Current wolf plans appear to be determined by how many wolves humans can kill and get away with, rather then how few wolves create problems and need to be removed. The study wrongly excludes the savings that could occur with good, preventative husbandry methods, set in place with non-lethal advancements.

However, is the only measure of their existence based on a bargain price value system? E.O. Wilson says it is prudent "to judge every scrap of biodiversity as priceless" (Wilson 1992). What is the real cost if wolves are not allowed to exist?

It is most distressing that a biologist should calculate the economic value of wolves, rather than focusing on their ecological value, by saying that "wolves inhabiting wilderness cost little to society" (Wilson 1992). Wilson's words seem to fit this sad situation: "If a price can be put on something, that something can be devalued, sold and discarded" (Wilson 1992).

The study ponders an even more disturbing premise by noting that trapping and hunting would not be efficient methods of wolf control. The following question is raised: Without using poison or substantial financial incentives, would the Minnesota wolf population be controllable in 2005? It is proposed that the sooner controls begin, the easier and less costly it will be.

Sadly, society seems to have entered a monumental era derived from the philosophy of Descartes. Apparently, all nonhuman animals are machines, and nature (as a giant mathematical mechanism) is to be measured, counted, computerized, and placed on a GIS map, with a statistical and monetary formula to determine how many specimens are profitable and thus allowed to survive. It seems that humans have yet to find what Aldo Leopold called ecological conscience (Leopold 1949). As long as predator control is based on the bullet as the cheapest method, there will be no earnest attempt to initiate preventative, non-lethal, and humane management programs.

Fear is a force still working against predators. Although some carnivores are capable of attacking people, these are rare encounters. Education has to include not only how to avoid predator encounters, but also teaching a willingness to give them space. It has been clearly demonstrated that modern technologies that decimate nature are adequate. Society needs to strive and find ways for living successfully in association with our native fauna and flora. Do people want wilderness, or Disney areas with just enough species to entertain from a safe distance? Is a wilderness experience a mob howling adventure? Do carnivores need to fit into approved behaviors? Are people willing to not intrude some habitats, or at least enter with knowledge and respect? Do carnivores have an intrinsic right to exist without directly helping humanity?

The majority of Americans support protection of large predators and restoration of a diverse, natural ecosystem. They do not support the abuse and over-utilization of public lands, nor the destruction of wild animals, wild lands, and natural resources for private interest groups. Humans have a responsibility to protect other species, even if it is inconvenient. Species should not be removed from protection for short term political favors to special interest groups. The stakeholders of our lands have been narrowly defined as consumptive users for too long. More importantly, future generations and wild creatures are the most vulnerable to today's actions. What legacy will society leave for future generations?

Carnivore recovery should be based on ecological principles, not political pandering. Recovery must ensure the long-term survival of the species, and also restore functional ecosystems that will benefit the full range of wildlife and human communities that depend on them.

Coexistence with large predators requires tolerance and education. The future of carnivores is still threatened by hostile attitudes. Wild carnivores are a part of American culture and heritage; their preservation cannot be neglected. Responsible stewards need to be on a path of coexistence, not returning to the sanctioned destruction of America's predators.

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Bringing Down the Walls

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Educational venues to view, interact, and learn about wildlife occur in many settings, including zoos, preserves, refuges, rehabilitation centers, national parks, and other wild lands. Each human believes his or her venue or political camp to be the best model for teaching ecology. The world is rapidly changing, both in the effects created by increasing human populations, and the often-heroic attempts to halt and even reverse such trends. In the 1960s, "ecology" was a new, mysterious word to most people. Thirty years later, the term "conservation" has been expanded beyond a resource management definition and now includes concern with retaining the planet's biodiversity and the actions that affect such change.

These concerns for sustainability, cessation of extinction, and enhancement of the welfare of endangered species ultimately will succeed only if they do not fall on deaf ears, but on the ears of those who know the language and arena of concern. As the family structure has taken on a different face from that of the past, the connection between young people and wildlife must be nurtured and not left to chance. Parents and grandparents often were the inspiration for wildlife wonderment through shared outings into the wild. Each person can probably recall those mentors who were the first to introduce nature.

Today's living and working conditions do not foster such opportunities. Urban living situations, increased job demands, widening ethnic perspectives, an aging population, and changes in nuclear family structure are but a few of the major shifts drastically affecting the nature education of young people. Alternative avenues for inspiring an interest in wildlife and wild lands must be recognized and enhanced for their ability to establish a closer relationship between people and wildlife.

The need for effective conservation education is absolutely urgent today. The desire to understand or merely view animals has been joined by the need to conserve their populations and ecosystems. Over the last few decades, the explosive growth of human populations has been linked with changes in the abundance of other species. The understanding of animals' needs and of human impacts on animal populations became critical; the future of these animals depends on people's actions. This understanding is no longer optional. Everyone in this discipline knows the message, but in spite of its size, this group is still in the minority.

Today, conservation education needs to be recognized as a specialty that benefits from diverse technologies, varied educational efforts, and multi-disciplinary expertise. The conservation message needs to shift from merely endangerment, captive breeding, or even reintroduction to include and emphasize the importance of saving habitat. The expertise to facilitate the delivery of this message exists. It is present within conservation groups and in areas that were previously ignored.

Zoos and aquariums draw 121 million visitors each year. Captive wildlife facilities have historically been considered substandard or poor substitutes for wildlife experiences. While hearing wolves howl at the Grizzly Discovery Center is not the same experience as hearing a pack of wolves in Lamar Valley, the zoo still allows the visitor to have a close and personal moment with this species. The next headline about a wolf is more likely to be read by someone who knows the sound of a howl than one who does not. Even more importantly, the impact on wild habitats should be considered if human-animal connections in a wild setting were the only experiences allowed.

Openness to learning about new exhibition techniques, husbandry programs and educational outreach programs offered in today's zoological institutions needs to be embraced as a viable non-consumptive augmentation to wildlife education. For the general visitor, zoos and aquariums serve the conservation community by providing information in a relaxed, family-oriented setting. Zoos and aquariums are positioned to play an ever-changing role in conservation through education.

Conservation requires an expanding perspective, but educators are divided and confounded by disagreements about the message and the appropriate media of delivery. Conservationists must share united intentions in order to be effective. Advocacy groups, wildlife protection groups, and zoological educators share a common goal: to instill in others the passion to protect, conserve, restore, and ultimately live harmoniously with the unique wildlife and wild lands in the environment. Therefore, the crucial partnerships that make the conservation message reach the most ears, in the most effective manner humanly and humanely possible, should be fostered.

News from Zoos

Minnesota Zoo, China Work to Save World's Most Endangered Tiger

Minnesota Zoo Conservation Director Dr. Ron Tilson led a team of Chinese conservation officials on an expedition through Longyan, Fujian Province, in an effort to save the critically endangered South China Tiger. Dr. Tilson also held a tiger conservation workshop, which marks the beginning of a partnership between China and foreign officials, including AZA's Tiger Species Survival Plan, on the South China Tiger Project.

More than 20 conservation officers from seven provinces attended the workshop, which highlighted tiger tracking and search techniques. Following the expedition, the team visited the South China Tiger Captive Breeding Program in Meihuashan Mountain, where "first phase facilities" have been constructed to breed and prepare the tigers for release. The South China Tiger, know to locals as a "Mountain God," is the most endangered subspecies of tigers know to exist. Fewer than 30 live in the wild and only 60 are held in Chinese Zoos. [Source: Communique]

Namibian Cheetah Ambassadors Come to the States

Ten young Namibian cheetahs arrived in the United States in April for distribution to AZA accredited zoos who participate in the Cheetah Species Survival Plan (SSP). These cheetahs, comprising a group of cats from eight different litters, all originated from commercial farmland areas and eventually found sanctuary at the Cheetah Conservation Fund (CCF).

The cheetahs represent a presidential gift from Namibia to the United States in recognition of the support it has given to cheetah conservation efforts in Namibia. Four of the cheetahs are housed at the Cincinnati Zoological Gardens and the other six were given to the White Oak Conservation Center. They will be integrated into the Cheetah SSP, which was established in 1982 and manages of all captive cheetahs at different facilities in North America as a unit. Working within this program, all the facilities holding cheetahs cooperate on issues concerning reproduction, genetics, diets and general husbandry of the species.

CCF selected these ten cheetah based strictly on the criteria that they are non-releasable animals. The circumstances under which these cheetahs were orphaned are mostly tragic, with their mothers being shot. However, all of the cubs found their way to CCF through the concern for their welfare of the people that caught them and in two cases, through the speedy intervention of a concerned neighbor. CCF has been the recipient of three grants from the AZA's Conservation Endowment Fund. [Source: Jack Grisham, AZA Cheetah SSP Coordinator]

Mexican Gray Wolves Make Zoo New Home

The National Zoo announced that it has made a home for three Mexican gray wolves as part of an international recovery effort to reintroduce the highly endangered wolf subspecies into the wild in Arizona, New Mexico and Mexico. At a news conference this morning at the new wolf exhibit, located uphill from the seal pool, zoo and animal conservation officials said the wolves will stay at the zoo until they can be released into a wilderness area in either Arizona, New Mexico or Mexico.

The National Zoo participates in the Mexican Wolf Species Survival Plan (SSP), which manages the Mexican wolves held in captivity. The SSP and the US Fish and Wildlife Service work cooperatively to support the Mexican Wolf Recovery Program, the international effort to reintroduce Mexican Wolves.

The Mexican gray wolf is the most rare and genetically distinct subspecies of gray wolf in North America but have not been seen in the United States and Mexico since 1970 and 1980, respectively. There are about 200 Mexican gray wolves living today, and nearly all of them were born and raised in captivity.

Conservation officials said they work with the ranchers to try to keep the wolves away from humans and livestock, and they attach radio collars to the animals to track their whereabouts. [Adapted from an article by Karlyn Barker, The Washington Post]

Information for News from Zoos *is provided by the American Zoo and Aquarium Association*

Mexican wolf (*Canis lupus baileyi*). Photograph by B. Moose Peterson/WRP.

Endangered Species
UPDATE

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