

Natural History Characteristics of Focal Species in the New Mexico Highlands Wildlands Network

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River otter account by Paul Polechla and Kathy Daly

WILDLANDS PROJECT



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Gray Wolf (*Canis lupus*)

*Please see accompanying Excel chart of Gray Wolf Habitat Characteristics in the folder titled "Focal Species Habitat Characteristics Charts" on the compact disk.

INTRODUCTION

Historically, the gray wolf (*Canis lupus*) was widely distributed throughout the NM Highlands planning area. It was primarily associated with mountainous areas and occupied habitats ranging from foothill grasslands to subalpine meadows. The gray wolf is a top predator and preys primarily on large ungulates, especially deer (*Odocoileus* spp.) and elk (*Cervus elaphus*). It requires large core areas for population persistence and safe linkages between patches of suitable habitat. Gray wolves are fully protected as endangered, threatened, or reintroduced "experimental" populations by the Endangered Species Act throughout the conterminous United States. Ecologically, the gray wolf is a top carnivore that serves to maintain ecosystem stability and promote overall biological diversity on a landscape scale.

DISTRIBUTION

Historic. Originally the gray wolf was widespread throughout North America, occurring from near Mexico City to arctic regions and from the Atlantic to Pacific oceans (Young 1944). It was absent only from extreme desert regions, such as southwestern Arizona, southern Nevada, and southern California (Young 1944) and from the southeastern United States, which was home to the red wolf (*Canis rufus*) (Goldman 1944).

"Apparently, [gray] wolves were very common in New Mexico at the beginning of its settlement by the Spaniards and later by the Americans" (Young 1944:41). Bailey (1931) documented the historical presence of wolves throughout New Mexico, including the eastern plains. Conflicts between gray wolves and domestic livestock in the Southwest began with early Spanish settlements (which extended into southern Colorado) in the 1600s. Wolf-livestock conflicts peaked in the late 1800s and early 1900s when large numbers of livestock were stocked on Southwestern rangelands and forests following the completion of railroad lines to the region (Brown 1983).

Bednarz (1988) estimated an historical abundance of about one gray wolf per 62 km² (24 mi²) of suitable habitat or about 1500 wolves occupying New Mexico prior to federal predator control measures initiated in 1915. Bailey (1931) estimated wolf density in the Gila National Forest in 1906 at about one wolf per

township (36 mi² or 93 km²). U.S. Biological Survey trapper, J. Stokely Ligon, estimated the New Mexico state-wide population of gray wolves at 103 adults in 1917, 45 in 1918, and practically eliminated in 1927 (Bailey 1931). Bednarz (1988) proposed 1942 as the estimated year of functional extirpation of gray wolves from New Mexico. Dispersing wolves from Mexico continued to enter New Mexico and Arizona for another three decades. The last wild wolf in New Mexico was discovered (dead) in the Peloncillo Mountains (south of the NM Highlands planning area) in 1970 (Brown 1983).

Current. The U. S. Fish and Wildlife Service is currently restoring gray wolves to the Gila National Forest in the southern portion of the NM Highlands planning area. As of April 2002, two Mexican gray wolves (*C. l. baileyi*) were known to be free-ranging in the Gila National Forest. About 14 more Mexican gray wolves were known to be free-ranging in the adjacent Apache National Forest in Arizona; and an additional 16 released or wild-born wolves have unknown statuses (Colleen Buchanan, USFWS, personal communication).

Potential. Areas of approximately 1000 km² (386 mi²) or greater within foothill or mountainous terrain with the following characteristics generally represent potentially occupied range for gray wolves: 1) persistent populations of deer and/or elk; 2) low densities of humans and roads; and 3) either connected by linkages of suitable habitat to, or not more than 80 km (50 mi) from, areas of suitable gray wolf habitat of at least 10,000 km² (3860 mi²). Large core areas are necessary to provide a source of dispersing gray wolves to recolonize smaller areas which are individually unlikely to sustain viable populations over long periods of time.

HABITAT

General. The abundance and availability of hoofed mammalian prey species (e.g., deer and elk) dictate the potential for wolves to inhabit an area (Fuller 1997). Otherwise, the gray wolf is a habitat-generalist, historically occupying all major vegetative community types that supported large ungulates in the Northern Hemisphere, except tropical rain forests and arid deserts (Goldman 1944; Young 1944; Mech 1970; Fuller 1997). Wolves are adaptable and are not generally affected by non-intensive human activities within otherwise suitable habitat, except as those human activities (e.g., illegal shooting, predator control, vehicle collisions, or elimination of prey) might facilitate the killing or elimination of wolves (Fritts and Carbyn 1995; Noss et al. 1996; Fuller 1997). Because of high human-caused mortality, wolves generally occur where both

road and human densities are low (Fuller et al. 1992). However, Mech (1995a) notes that improved public attitudes toward wolves, which presumably translates to increased human tolerance, has led to wolves occupying areas of higher road density in Minnesota. Mech (1995a) postulates that gradual habituation results in wolves becoming more tolerant of human disturbances.

Preferred. Ninety-two percent of the variation in wolf density throughout North America is directly related to variation in ungulate biomass (Fuller 1997). That is, the greater the ungulate biomass, the higher the density of wolves until, theoretically, other factors (e.g., territoriality) set an upper limit to wolf numbers (Boyce 1995; Messier 1995).

Since humans cause significant mortality in many gray wolf populations, isolation or protection from humans or a high level of tolerance by humans increase the quality of gray wolf habitat. Except for the possibility of vehicle collisions, roads have little effect on potential habitat use by wolves. Wolves show a slight aversion up to about 1 km (0.6 mi) from well-traveled roads, but readily use lightly-traveled, closed, or unplowed roads as travel ways (Thurber et al. 1994; James and Stuart-Smith 2000). The primary negative effect of roads relates to the access they provide for humans to enter wolf-occupied areas and kill wolves (Mech et al. 1988). Following the initiation of moose hunting in Alaska, Thurber et al. (1994) observed that wolves tended to avoid a well-traveled highway (previously not avoided by wolves) and subsequently established pack territories did not cross the highway. Thus, roads may influence spatial distribution of packs (Thurber et al. 1994).

Generally, wolf populations do not survive in areas with road densities >0.6 km/km² (1 mi /mi²) (Thiel 1985; Jensen et al. 1986; Mech et al. 1988; Fuller 1989; Mladenoff et al. 1995; but see Mech 1995a). In Wisconsin, Mladenoff et al. (1995) found that road densities did not exceed 0.23 km/km² (1 mi/2.8 mi²) in core and sensitive (e.g., den and rendezvous sites) areas within pack territories. The relationship between road density and wolf survival has not been elucidated for more open landscapes in the West (Weaver et al. 1996), but greater visibility and openly expressed hostility toward wolves by some interest groups suggest that even lower road densities may be necessary there.

Negative interactions between wolves and livestock or pets, although not common, usually cause human animosity toward wolves, which often results in illegal killing or agency-initiated control of wolves (Mech 1995a). Wolves also tend to avoid human settlements and developments (Paquet et al. 1996;

Thurber et al. 1994) and are sensitive to disturbances near active den sites (Weaver et al. 1996). In Minnesota, wolves persist in areas where human densities are less than 4 persons/km² (0.4 mi²) and road densities were <0.70 km/km² (1 mi/0.9 mi²) and in areas where human densities are <8 persons/km² (0.4 mi²) and road densities were <0.50 km/km² (1 mi/1.2 mi²) (Fuller et al. 1992). W. Robinson placed the critical threshold for human population density at somewhere between 6 and 12 persons per square mile (2.3-4.6 persons/km²) (Klinghammer 1975:433).

Thus, the relative quality of wolf habitat is inversely related to density of humans, livestock, pets, and roads and directly related to the level of favorable human attitudes toward wolves.

Because gray wolves have large territorial requirements, relatively large areas are required to ensure population viability (Soulé 1980; Fritts and Carbyn 1995; Noss et al. 1996). Mech (1995b:544) defined candidate areas for wolf recovery as “extensive wilderness areas with, or near, minimal areas of livestock grazing or human residences.” However, in the West and Southwest, where most wilderness areas and public lands are grazed by domestic livestock, creative incentives to induce ranchers to modify traditional livestock husbandry practices in ways that lessen the potential for livestock depredation by wolves (or to switch to other more wolf-friendly income-producing endeavors) may be necessary to ensure long-term survival of wolf populations in otherwise high-quality wolf habitat (Rasker and Hackman 1996; D. Parsons, personal communication).

Den Sites. Types and sites of gray wolf dens vary greatly (Bailey 1931; Mech 1970). Dens may be in rock crevices, hollow logs, overturned stumps, but are usually in a dug burrow on an elevated site (Bailey 1931; Nowak 1991). Four of six dens observed by Peterson et al. (1984) were dug among the roots of large trees. The most commonly shared attribute of wolf dens is proximity to water, presumably because of the nursing female’s increased daily requirement for liquid (Mech 1970; Nowak 1991). Human disturbance may cause wolves to abandon their dens (Ballard et al. 1987).

FOOD HABITS / HUNTING BEHAVIOR

Preferred prey for gray wolves in the planning area include white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*) (Brown 1983), and elk (*Cervus elaphus*) (Brown and Parsons 2001). Alternative prey species may include pronghorn (*Antilocapra americana*), javelina (*Tayassu tajacu*), beaver (*Castor*

canadensis), rabbits (*Sylvilagus* spp.), hares (*Lepus* spp.), and small mammals. Some predation on domestic livestock can also be expected (Brown 1983; Fritts et al. 1992; Bangs et al. 1995; Brown and Parsons 2001), but contemporary research indicates that livestock constitutes a relatively small part of wolves' diets (Fritts et al. 1992; Gipson et al. 1998).

Prey of wolves tend to be very young, very old, malnourished, injured, or otherwise disadvantaged (e.g. by deep snow), making them more vulnerable to predation (Mech 1970; Peterson et al. 1984; Nowak 1991). Weaver et al. (1996:966) described wolves as "expanding specialists" that specialize in killing vulnerable individuals within populations of large ungulates (e.g., elk and moose, *Alces alces*), but readily generalize to more common prey, usually deer in northern areas where wolf food habits have been studied. On the Kenai Peninsula of Alaska, wolves seemed to defer to brown bears (*Ursus arctos*) at kills, but did not consistently do so for black bears (*Ursus americanus*). Bears emerging from winter dens frequently scavenged on wolf-killed moose (Peterson et al. 1984).

Predation methods of gray wolves differ from those of other large predators in Southwestern ecosystems. Wolves pursue their prey by chasing, sometimes over long distances, and often hunt in groups, while mountain lions (*Puma concolor*), for example, hunt singly and usually rely on ambush or a short rush to capture their prey (Young 1946; Mech 1970). Thus, gray wolves exert a unique evolutionary influence on their prey.

POPULATION DYNAMICS

Life History. The basic social unit in gray wolf populations is the pack, usually consisting of 2-8 closely related wolves with strong bonds of attachment that travel, hunt, feed, and rest together in a loose association (Mech 1970; Nowak 1991; Wayne et al. 1995). The "alpha" or breeding pair are usually unrelated (Smith et al. 1997). Breeding usually occurs only between the alpha male and female (\geq two years old) once each year in February or March (Mech 1970; Peterson et al. 1984; Nowak 1991). However, multiple litters of up to three per pack have been reported for the recovering wolf population in the Yellowstone National Park area, where food availability is high (Smith and Murphy 1998). Gestation is about 63 days; litters average about five pups; generally, fewer than half of pups born survive their first year; not all litters born within the same pack are likely to survive; and adults have an annual survival rate of about 80% and a few may live up to ten years in the wild (Mech 1970; Smith and Murphy 1998). Age of females at first

parturition averages about three years; annual productivity averages about four pups per adult female; and the average female wolf produces about six female pups over her lifetime (Weaver et al. 1996). Alpha wolves dominate the behavior of subordinate wolves, try to prevent intrusions into the pack's territory by other wolves, and generally lead the family group in hunting (Mech 1970).

Population Structure and Viability. A viable wolf population requires an area of some minimum size, adequate prey, and security from excessive human exploitation (Fritts and Carbyn 1995; Fuller 1997; Haight et al. 1998). These three factors could be considered the "critical elements" for wolf population viability and conservation. Vucetich et al. (1997) emphasize that wolf conservation efforts should be more concerned with the number of packs rather than the number of individual wolves, because, generally, only one adult pair per pack breeds each year.

Wolves need large areas to assure population viability. Noss et al. (1996:950) aptly state that "[a]reas apparently needed to maintain viable populations [of large carnivores] over centuries are so large as to strain credibility; they certainly strain political acceptance." The total area that is necessary to support a viable wolf population, at least for the short term, is influenced by many factors and cannot be determined with certainty (Fritts and Carbyn 1995). A review of the relevant literature by Fritts and Carbyn (1995) suggests that core areas of suitable habitat in the range of 8,000-26,000 km² are required to support viable wolf populations. Alternatively, a complex of smaller areas (500-3,000 km²) of suitable wolf habitat connected by effective dispersal linkages could accomplish the same conservation objective for wolves (Fritts and Carbyn 1995; Noss et al. 1996; Haight et al. 1998). Vucetich et al. (1997) predicted that wolf populations of fewer than 100 individuals may be inadequate for long-term wolf survival. Habitat fragmentation on a scale that precludes effective dispersal of wolves may result in genetic deterioration of isolated local populations, which tend to be comprised of closely related individuals (Wayne et al. 1995).

Population Density. The density of wolf populations is directly related to available prey biomass, and can vary from 5 to 100 wolves/1000 km² (386 mi²) (Van Ballenberghe et al. 1975; Fuller and Keith 1980; Peterson et al. 1984; Nowak 1991; Wydeven et al. 1995; Fuller 1997; Lariviere et al. 2000). Wolf densities in the lower end of this range are more common. Fuller (1997) determined that the number of wolves per 1000 mi² (2590 km²) is 4.19 times the biomass index (number of deer equivalents per square mile). Bednarz (1988) estimated

historic wolf density in New Mexico at about 16/1000 km² (386 mi²).

Home Range. Territories are maintained by the alpha pair through an intricate network of scent-marking and by howling as a means of securing the needs of the family group: food, water and shelter (Mech 1970; Peters and Mech 1975, Harrington and Mech 1979). Territory size is inversely related to available prey biomass; and Fuller (1997) derived a formula for estimating territory size if prey biomass is known. Other factors affecting territory size include pack size, seasonal movement of prey, and heterogeneity of habitat (percentage of territory actually suitable/useable by wolves). A wide range of territory sizes has been reported (Nowak 1991). Very large territory sizes are associated with northern regions with large (e.g., moose) and/or migratory (e.g., caribou, *Rangifer tarandus*) prey. Most wolf pack territories in more southern regions fall within the range of 50-500 km² (20-200 mi²). Bednarz (1988) predicted that wolves reintroduced to New Mexico would establish territories in the range of 200-400 km² (77-155 mi²).

Causes of Death. Generally, wolf populations tend to decrease when yearly mortality exceeds 35 to 40 percent (Keith 1983; Peterson et al. 1984; Fuller 1989; Haight et al. 1998; Lariviere et al. 2000). Age-specific mortality is greatest in yearlings (especially those that disperse from their packs), followed by pups, and lastly adults (Peterson et al. 1984; Fuller 1989; Pletscher et al. 1997). In stable populations, pups may consist of about 40 percent of the population and adults, 50 percent (Van Ballenberghe et al. 1975; Peterson et al. 1984). In expanding populations, pups may comprise half the population (Fritts and Mech 1981). In heavily exploited populations, however, less than 35 percent are pups, and yearlings are nearly absent (Gasaway et al. 1983).

In most areas of North America humans are the primary cause of mortality among wolves (Fritts and Mech 1981; Peterson et al. 1984; Ballard et al. 1987; Fuller 1989; Boyd et al. 1995; Fritts et al. 1995; Forbes and Theberge 1996; Bangs et al. 1998). In a colonizing wolf population in Wisconsin, humans caused 72% of mortality from 1979 to 1985, but only 22% from 1986 to 1992 (Wydeven et al. 1995), suggesting an increasing level of tolerance for wolves by humans following recolonization (see also Mech 1995a). Humans caused 95 percent of known mortalities in a recolonizing wolf population in the Rocky Mountains between 1984-91 (Boyd et al. 1995). Actual causes include legal and illegal shooting and trapping; vehicle collisions; control of livestock-depredating wolves; poisoning, and displacement of wolves through harassment or intentional translocations eventually leading to death of the

wolf from other causes. In contrast, Meier et al. (1995) found that 52% of documented wolf mortality in Denali National Park and Preserve in Alaska was caused by intraspecific strife (wolves killing other wolves). They theorized that this level of intraspecific strife was a normal consequence of wolf territoriality in the absence of extensive human interference.

Natural mortality predominates in the few large National Parks in North America where wolves are protected from human exploitation (Fritts and Carbyn 1995). Disease, parasitism, insufficient food, and intraspecific strife cause most natural mortality (Mech 1970; Mech and Karns 1977; Todd et al. 1981; Carbyn 1982; Brand et al. 1995).

Diseases and parasites affecting wolf populations include canine parvovirus (CPV), canine distemper virus (CDV), rabies, Lyme disease, heartworm and lice (Custer and Pence 1981; Brand et al. 1995; Johnson 1995; Mech and Fritts 1987; Mech and Goyal 1993; Mech et al. 1985; Thieking et al. 1992). Pup survival in wolf packs is related to the incidence of CPV in wolf populations and adult pack mates (Mech and Goyal 1993). Although they are susceptible to rabies, outbreaks of this disease among wolves are rare (Mech 1970; Johnson 1995). Diseases of wolves are also found in domestic dogs and may be transmitted to wolf populations by them. The wolf's contribution to the overall parasite or pathogen problems in any given area is expected to be slight because of the relatively small population size and low population density of wolves.

Population Status. Wolves were extirpated from the entire NM Highlands planning area by the mid 1900s (Brown 1983; Bednarz 1988). Ongoing reintroduction efforts led by the U.S. Fish and Wildlife Service have a goal of restoring a population of at least 100 wolves to the Apache and Gila national forests (USFWS 1996). More than half of this population is expected to occupy the Gila National Forest, which forms the southern portion of the NM Highlands planning area.

MOVEMENTS

Dispersal. At about two years of age, an average of 26% of gray wolves (slightly more males than females) disperse away from their natal packs in search of vacant habitat and/or a mate (Weaver et al. 1996). Dispersers usually travel alone and have a high risk of death (Peterson et al. 1984). Their travels take them through unfamiliar terrain where food and shelter may be difficult to locate and they may be unable to avoid territorial wolves and people. Wolves that locate a mate and an unoccupied patch of suitable habitat usually establish a territory (Rothman and Mech 1979, Fritts and Mech 1981).

Wolves are capable of dispersing hundreds of kilometers, with two documented dispersals exceeding 840 km (522 mi) (Fritts 1983; Boyd et al. 1995). Weaver et al. (1996) examined several studies of wolf dispersal and calculated an average dispersal distance of 85 km (53 mi), with males dispersing slightly farther than females. In recolonizing wolf populations, where suitable habitats are not fully occupied by wolves, mean dispersal distances tend to be somewhat longer (113-144 km [70-90 mi]) (Boyd et al. 1995; Wydeven et al. 1995).

The propensity for dispersal and long dispersal distances of gray wolves make them highly successful in colonizing suitable, unoccupied habitats within about 85 km (53 mi) of extant source populations (Weaver et al. 1996).

Migration. Forbes and Therberge (1995) documented wolf movements in response to seasonal deer movements in and near Algonquin Provincial Park in Ontario, Canada. Apparently, the exodus of prey needs to be nearly complete to induce wolves to follow. Migrational movements of wolves have been documented in tundra ecosystems where the primary prey are migrating caribou herds (Kelsall 1968; Parker 1973, both cited by Forbes and Therberge 1995) and in Wood Buffalo National Park where wolves follow migrating bison (Carbyn 1993).

Barriers to Movement. Little information has been published on how dispersing wolves navigate the landscape, especially where it is fragmented (Weaver et al. 1996). In Minnesota, large lakes and the Mesabi Iron Range (characterized by deep open pit mines, rock dumps, tailings ponds, taconite plants, highways, and several human settlements) presented barriers to movements of translocated wolves (Fritts et al. 1984). Along the Ontario-Michigan border, Jensen et al. (1986) observed that areas of human development/settlement where road density was greater than 0.6 km/km² (1 mi/mi²) apparently served as a barrier to wolf dispersal. Mladenoff et al. (1995) described development corridors along major highways as "semipermeable barriers" to the movement of wolves into eastern Wisconsin. However, Mech (1995a:272) states that wolves dispersing from core populations in Minnesota successfully negotiated "highways, traffic, residences, habitat fragmentation, and other human disturbances."

Three wolves dispersing from northern Minnesota to or toward Wisconsin and upper Michigan had to have crossed Interstate Highway 35 and two of these wolves probably also crossed 4-lane Minnesota Highway 53. All three wolves most likely traveled around the city of Duluth, Minnesota and most likely crossed extensive open areas, wolf-free habitat, and/or areas with high road densities (Mech et al. 1995). However, one of the

wolves was killed on Wisconsin Highway 51 after having traveled at least 555 km (345 mi) from her original capture point in Minnesota. Recolonization of wolves in Wisconsin and upper Michigan by wolves dispersing from Minnesota, demonstrates that, when protected from deliberate killing, dispersing wolves can successfully cross unfavorable areas and colonize new areas in semi-wild landscapes, even though some human-caused mortality occurs (Mech 1995a; Mech et al. 1995; Mladenoff et al. 1995; Haight et al. 1998).

Use of Linkages. Natural recolonization of wolves in Montana resulted from wolves using connected habitats along the Rocky Mountains to disperse from Canada (Boyd et al. 1995). Thurber et al. (1994) found that wolves were attracted to a lightly-used, closed pipeline road and to secondary gravel roads and used them as a travel ways. Thus, lightly-used or mostly closed unpaved roads within potential linkages may not be detrimental to, and may enhance, wolf movement through the linkage. Human interference (e.g., heavily-traveled roads, settlements, and killing of wolves), rather than distance, will most likely limit the movement of wolves between suitable habitat patches (Forbes and Boyd 1997).

The presence of discrete habitat linkages for dispersal seems to be less important to wolves than to most other species (Fritts and Carbyn 1995). Perhaps the most useful movement pathways for wolves are characterized by broad, heterogeneous linkages enhanced by the absence of potential barriers to their movement (e.g., highways and developed areas) and protection from human-caused mortality (Noss et al. 1996; Forbes and Boyd 1997). Under a metapopulation scenario, protection of dispersing wolves is critical to the long-term occupancy of patches of suitable wolf habitat (Haight et al. 1998).

ECOLOGY

Effects on Prey. Predator-prey relationships can be and most often are extremely complex. The effect of wolf predation on prey populations depends on the number of wolves, prey kill rates, and the capacity of prey populations to sustain losses to wolves. Such effects may be more significant than previously thought (D. Pletscher, personal communication). Differences in these three basic components can produce a full array of results from little effect to complete elimination of some prey populations (Seip 1995). Other limiting factors such as hunting, poor nutrition, and predation by other predators will reduce the capacity of a prey population to withstand losses to wolves (Seip 1995; Vales and Peek 1995). In contrast, effective predator avoidance strategies of prey reduce wolf hunting effi-

ciency and increase the capacity of prey populations to sustain losses to wolves (Seip 1995). In multiprey systems, relative vulnerability of different prey species and prey switching by wolves in response to changes in the abundance or vulnerability of various prey species can substantially effect the population response of given prey species (Seip 1992; Dale et al. 1995; Messier 1995). Seip (1992) concluded that wolf predation was the primary cause of decline for the Quesnel Lake caribou herd. Whereas, the nearby Wells Gray Park caribou herd, which had minimal contact with wolves, increased slightly during the same period. Wolf populations in the Quesnel Lake area were sustained by moose in the winter, when caribou were generally not available to wolves, and showed no decline with declining caribou populations. Seip (1992) predicted that wolves could potentially extirpate the Quesnel Lake caribou population.

Generally, as prey populations increase, the total number of prey killed by wolves will increase exponentially because of the multiplicative effect of an increase in the number of wolves (i.e., numerical response) and the increased kill rate (i.e., functional response) of individual wolves (Seip 1995). Theoretically, wolf and prey populations will stabilize around population densities where the number of prey killed by wolves equals the potential annual incremental increase of the prey population (Seip 1995). Other environmental factors and time lags in predator responses will cause cycles around the equilibrium point.

In southeastern Alaska, deer populations are higher on islands where wolves are absent compared to islands where wolves are present (Smith et al. 1987, as cited in Ballard et al. 2001). Models developed for wolf reintroduction projects in Wyoming and Idaho (Boyce 1995) and Arizona and New Mexico (Green Hammond 1994) predicted moderate declines in ungulate populations (5-22%) relative to the no-wolf scenarios. Under plausibly favorable environmental/habitat conditions, Green Hammond (1994) predicted that prey populations would increase above pre-existing levels following the establishment of a wolf population. Mech et al. (1995) documented an increase in caribou populations despite a doubling of the wolf population and noted no discernible decline in moose and wild sheep (*Ovis* spp.) populations. Ballard et al. (1987) concluded that wolf predation was not preventing moose population growth during their study in Alaska, but found evidence suggesting that wolf predation may have reduced recruitment of moose calves into the adult population.

Some studies suggest that predation by wolves and other top predators may retard or prevent the recovery of ungulate populations following a decline due to other causes (e.g., severe weather or human over-harvest) (Ballard et al. 1987; Dekker et

al. 1995; Seip 1995; Ballard et al. 2001). Gasaway et al. (1992) concluded that predation of moose by wolves and bears was the major factor limiting moose at low densities and observed that moose populations irrupted following predator reduction. The recent review of this topic by Ballard et al. (2001) suggests that predation does not cause ungulate population declines, except possibly in some elk, moose, and caribou populations (Gasaway et al. 1992; Seip 1992; Dekker et al. 1995).

It is well-established that wolves prey disproportionately on prey with the following characteristics: young age (fawns and calves), old age, low bone marrow fat (nutritionally impoverished), disease or parasites, arthritis, or other physically debilitating injuries or conditions (Mech 1970; Peterson et al. 1984; Mech et al. 1995). Adult male ungulates are more vulnerable to predation during the annual rut, presumably because of their poor nutritional condition during this time (Mech et al. 1995).

Effects on Mesopredators. Interference competition, the direct displacement of competitively subordinate individuals, commonly occurs among sympatric canid species (Peterson 1995). Generally, larger canids displace smaller ones (e.g., wolves kill coyotes [*Canis latrans*], and coyotes kill foxes [*Vulpes* spp., *Urocyon cinereoargenteus*]). Several studies have documented wolves killing coyotes and both behavioral and spatial avoidance of large canids by smaller canids (Peterson 1995; Smith et al. 1996; Arjo and Pletscher 1999). The relatively recent expansion of the range of coyotes eastward and northward following the extirpation of gray wolves probably resulted from the elimination of interference competition (Nowak 1991; Ballard et al. 1999). This theory is supported by the fact that coyotes are rare within the core range of Minnesota's wolf population, but common immediately outside (Peterson 1995). Wolves effectively eliminated coyotes as serious predators of deer in British Columbia and Alaska (Ballard et al. 1999). However, Arjo and Pletscher (1999) concluded that in northwestern Montana behavioral adaptations by coyotes and separation of the two species in space and time allowed sympatric coexistence of wolves and coyotes.

Ecological Effects. Following a field trip to an undisturbed region of the Sierra Madre in Chihuahua, Mexico, Leopold (1937) wrote: "Deer irruptions are unknown. Mountain lions and wolves are still common... There are no coyotes in the mountains... I submit for conservationists to ponder the question of whether the wolves have not kept the coyotes out? And whether the presence of a normal complement of predators is not, at least in part, accountable for the absence of irruption [of deer populations]? If so, would not our rougher mountains [in Arizona and New Mexico] be better off and might we not have more normalcy in our deer herds, if we let the wolves and lions

come back in reasonable numbers?"

Modern ecological theory and empirical evidence strongly suggest that top predators, such as gray wolves, exert a controlling influence over entire ecosystems through a cascade of regulatory effects across the various trophic levels of the ecosystem—the so called “top-down effect” (Palomares et al. 1995; Henke and Bryant 1999; Terborgh et al. 1999; Miller et al. 2001). Potential immediate effects of removing top predators include increases in the abundance of large herbivores (e.g., deer and elk) and mesopredators (e.g., bobcats, *Felis rufus*, and coyotes) (Henke and Bryant 1999; Terborgh et al. 1999; Dugelby et al. 2001; Miller et al. 2001). These proximate ecological changes may then generate a cascade of effects operating through ecosystem processes such as herbaceous consumption, predation, competition, and behavioral exclusion ultimately resulting in ecosystem simplification, loss of ecosystem stability, and loss of biological diversity (Henke and Bryant 1999; Terborgh et al. 1999). For example, Henke and Bryant (1999) observed that the number of rodent species in their western Texas study area declined from 12 to 2 following removal of coyotes. They concluded that the Ord’s kangaroo rat (*Dipodomys ordii*) gained a competitive advantage over smaller rodent species in the absence of coyote predation. The other remaining rodent was a woodrat (*Neotoma micropus*). Henke and Bryant (1999) also observed an increase in mesopredator abundance following coyote removal. Large predators that can, and do, kill several different species of prey and can readily switch among them, such as the gray wolf and coyote, can be expected to exert stronger influences over lower trophic levels and ecological processes than specialized predators or omnivores (Terborgh et al. 1999).

In Alaska and the Yukon, where wolves and bears (brown and black) were near carrying capacity, moose populations (148/1000 km² [386 mi²]) were well below carrying capacity based on food availability (Gasaway et al. 1992). That is, top predators were holding moose populations at a low-density dynamic equilibrium. Where wolf and bear populations were held below carrying capacity (by hunting or control measures), moose population densities were much higher (663/1000 km² [386 mi²]). Gasaway et al. (1992) noted that moose remained abundant in areas with only a single moose predator (i.e., wolves, black bears, or brown bears) or where the predators were brown and black bears but not wolves. This suggests that wolves are key to the top-down regulation of ungulates by the suite of top predators present in this ecosystem. Where moose are minor prey in wolf-bear-multiprey systems, moderate moose densities have been observed (Gasaway et al. 1992).

Further evidence of top-down ecosystem effects of wolves is provided by White et al. (1998). They attribute the decline of

aspen stands in national parks in the Rocky Mountains of North America, at least in part, to an increase in elk populations resulting from the elimination of predation by wolves. Elk are very effective browsers of young aspens, which can prevent their regeneration. Human-caused changes in fire regimes also contribute to the decline of aspens. White et al. (1998:457) conclude that “existing data provide strong evidence to support the role of top-down processes (e.g., predation and human-caused fires) in the long-term structuring of montane trembling aspen communities.”

JUSTIFICATION AND FOCAL VALUE

Umbrella. Because of its need for large territories and use of a variety of habitats, protection of habitat for a viable population of gray wolves will protect linked habitats for many other species (Noss et al. 1996). Because gray wolves tend to be habitat generalists, the extent of biological diversity conservation conferred by wolf conservation depends greatly on the juxtaposition of suitable gray wolf habitat and regional concentrations of biological diversity (Noss et al. 1996).

Keystone. By preying on deer, elk, and other ungulates and killing mesopredators such as coyotes, gray wolves appear capable of exerting a top-down regulation of ecosystems (Terborgh et al. 1999; Miller et al. 2001). Recent research from Yellowstone National Park suggests that the presence of wolves in a landscape may not only affect prey numbers, but also the behavior of prey species such as elk (*Cervus elaphus*), which in turn can affect the recruitment of certain tree species and cause shifts in vegetation communities (Ripple and Larsen 2000). Results of extensive research strongly suggest that, under certain circumstances, gray wolf predation can 1) suppress and dampen oscillations of prey populations; 2) cause a more even distribution of prey across the landscape thus reducing potential habitat damage; 3) effect the composition of vegetative communities; 4) enhance biological diversity by contributing to the suppression of mesopredator populations; and 5) promote ecosystem stability. Empirical evidence suggests that gray wolves likely play a keystone role in ecosystems and contemporary ecological theory supports their contribution to such a role (Miller et al. 2001). Certainly, the case for top-down regulation of ecosystems is strengthened when one considers the collective predation effects of the full suite of top predators present (see Gasaway et al. 1992).

Flagship. Perhaps more than any other animal, the gray wolf is a powerful indicator of shifting attitudes of North Americans toward the natural world (Kellert et al. 1996). Opinion polls

consistently demonstrate broad public support for wolf restoration and conservation; however, negative views are often prevalent among people living near existing or proposed wolf populations (Kellert et al. 1996; Schoenecker and Shaw 1997). Conservation efforts focusing on gray wolves should generate considerable popular support, as well as generate substantial controversy.

Wilderness Quality Indicator. Gray wolves tend to avoid or become extirpated from areas characterized by even moderate densities of roads and humans. Thus, large core areas with wilderness qualities (especially low road and human densities) may be required for the protection and preservation of viable populations of gray wolves. Wolves are not necessarily wilderness-dependent, although they do benefit from it. Typically, species that use wilderness are species that are directly persecuted or are associated with habitats that humans convert to other forms (Mattson 1997). Wolves do not require wilderness areas *ecologically* (especially in areas with high human tolerance like Poland and Minnesota), but in many areas (such as the American Southwest) they do need wilderness *socially* due to human intolerance. Wolves qualify as wilderness indicators only in regions where they are persecuted. Without significant changes in human values and culture in the Southwest, wolves will need habitat that is remote from negative human behavior if they are to survive in viable numbers (Mattson 1997).

MANAGEMENT RECOMMENDATIONS

Reintroduce Wolves. Studies need to be conducted to determine the amount and distribution of potentially suitable wolf habitat and the feasibility of wolf reintroduction within the NM Highlands planning area. If feasible, reintroduction plans should be developed and implemented as soon as practicable.

Establish Refugia. Refugia and habitat connectivity are essential for wolf recovery to occur (Boyd et al. 1995). Wolf population growth and survival will likely be limited by humans and their developments (i.e., increased road and human density within otherwise suitable wolf habitat). The integrity and security of suitable wolf habitat must be protected by maintaining wildland refugia with low road and human densities (Fuller et al. 1992). Areas in excess of 8000 km² (3000 mi²) characterized by suitable wolf habitat should be considered for refuge designation. Measures to increase native ungulate prey populations within refugia (e.g., restoration of natural fire regimes and curtailment of ungulate hunting) would enhance their value for wolf conservation. Protected refugia would serve as reservoirs, contributing to the restocking or maintenance of

population viability of adjacent populations depleted by various mortality or habitat suitability factors. Human hunting or trapping of wolves should be prohibited in refugia (Lariviere et al. 2000). We recommend full protection for wolves in core refugia, even following their delisting as endangered or threatened species. We assume here that refugia will not contain private lands; or, if they do, only lands whose owners are supportive of wolf protection will be included.

Areas of suitable wolf habitat in excess of 500 km² (200 mi²) should be protected and enhanced to support subpopulations of gray wolves. Eventual levels of protection in these areas should be consistent with long-term conservation goals for gray wolves and responsive to potential conflicts with human endeavors (Mech 1995a).

Protect and Restore Linkages. Few remaining patches of suitable gray wolf habitat are large enough to guarantee long-term population persistence. Movement of wolves among subpopulations separated by less suitable habitats is essential to the long-term genetic health and viability of a restored metapopulation of gray wolves in the NM Highlands planning area. Human-caused mortality presents the greatest threat to the successful movement of wolves among subpopulations. Intensive human developments and activities have also been shown to impede wolf movements. Wildlife movement linkages will best serve wolves if road densities are maintained below 0.4 km/km² (0.64 mi/mi²); human densities are maintained below 4 persons/km² (10/mi²); linkage lengths (i.e., distances between patches of suitable habit) do not exceed 80 km (50 mi); and wolves are accorded full protection within designated movement linkages.

Land management agencies should strive to restore nonfunctional or diminished linkages and protect existing linkages where possible. Development authorities and interests should be made aware of the critical importance of habitat linkages.

Road Closures. Indirectly, because they provide human access, roads are the most significant contributing factor to wolf mortality. In critical core areas and linkages between them where existing road densities exceed suitability thresholds or human-caused mortality is documented, some nonessential roads (e.g., off-road vehicle trails and dirt roads) should be closed. We do not advocate the closing of important thoroughfares or paved roads or highways.

Public Outreach. Public interest in wolves and their restoration and management runs very high (Kellert et al. 1996). Human communities in or near occupied or potentially suitable

wolf habitat tend to be rural and generally have the most negative attitudes toward wolves (Kellert et al. 1996; Schoenecker and Shaw 1997). Successful wolf recovery must be sensitive and responsive to the needs of human communities (Mech 1995a; Kellert et al. 1996; Rasker and Hackman 1996).

Difficulties surrounding wolf restoration and management are largely social and political (Clarkson 1995; Bangs et al. 1998; Parsons 1998). Bangs and Fritts (1996:412) predict that “wolf restoration will for some time continue to be unnecessarily controversial, expensive, complex and challenging.” Boitani 1995 believes that wolf conservation is accomplished by psychology and education. Kellert et al. (1996) assert that increased education about wolves often reinforces existing views or attitudes rather than changing them; but, contrary to this assertion, Fritts et al. (1995) state that continued emphasis on education is essential to successful restoration and management of wolves. Ultimately, the fate of wolves in most of North America is more a function of human tolerance than of nature reserve availability (Fritts and Carbyn 1995). We believe that effective public outreach and involvement programs are essential to successful gray wolf recovery efforts.

Habitat Enhancement. Habitat enhancement that improves the carrying capacity for and productivity of ungulate populations would benefit and potentially support higher wolf populations (Boertje et al. 1995). We favor restoration or mimicking of natural processes (e.g., fire) to achieve habitat improvements.

Wolf-Livestock Conflicts. Rates of depredation on domestic livestock by wolves in North America are low and range from 0.23 to 3.0 per 1000 cattle and 0.54 to 2.66 per 1000 sheep (Bangs et al. 1995) but, nevertheless, cause substantial controversy and anti-wolf sentiment. Bangs et al. (1995) believe that livestock damage prevention, control of depredating wolves, and compensation programs improved communications with agriculturists and their tolerance of predators and reduced animosity between the rural public and resource agencies. Further, they believe that this result enhances success of recovery and long-term conservation of wolf populations.

Some control of “problem” wolves will, undoubtedly, be necessary. But such control should 1) not substantially impede the progress of recovery efforts, 2) be limited to specific offending animals, and 3) should be tied to incentives for changes in livestock management practices that further reduce the potential for conflicts, especially on public lands.

The Defenders of Wildlife compensation program for owners of livestock killed or injured by wolves should be continued with modifications that provide incentives for livestock operators to improve management and reduce the potential for depredation by wolves. Substantial thought and effort should be applied to the development of economic incentives for wolf recovery—the Defenders of Wildlife “Wolf Country Beef” project being one example. Voluntary retirement of livestock grazing permits on public lands should be vigorously pursued.

Management Zones. As wolves move into agricultural zones and areas with higher densities of humans, conflicts with humans (e.g., livestock depredation and killing of pets by wolves) will increase proportionately to the availability and vulnerability of livestock and pets (Mech 1995a). Such conflicts, left unresolved, lead to public and political opposition or agency reluctance to pursue wolf recovery (Mech 1995a). Public demand for resolution of wolf-human conflicts may be best addressed through a zoned management approach (Clarkson 1995; Mech 1995a). Under such an approach, protections for wolves would be incrementally relaxed from cores and linkages to compatible use areas to high-intensity human use areas. This could and, according to Mech (1995a), should include very liberal (or even unregulated) killing of wolves by landowners and the general public in areas where wolf recovery is deemed incompatible with human activities. Zones should be as large as possible because wolf populations require large areas and wolves disperse over long distances (Clarkson 1995). Consideration should be given to protecting wolves in pockets of suitable habitat that may lie outside of broader protected zones. Such a scheme, while more difficult to manage, may support more wolves in more areas (Mech 1995a). The scale and configuration of a zoning scheme requires careful consideration of wolf recovery and conservation goals, management capabilities, and potential conflicts with human activities (Clarkson et al. 1995; Mech 1995a; Haight et al. 1998).

Monitoring. Once wolf reintroduction programs have been initiated, monitoring becomes a critically important management activity. Monitoring programs should focus on population status, causes of mortality, disease, wolf-human conflicts, and threats to habitat quality and quantity and to recovery success (Fuller et al. 1992; Mech 1995a; Wydeven et al. 1995). Linkages between subpopulations could be identified by radio-tracking movements of dispersing wolves. Making a “best guess” from a map often misses the mark.

Managers need to identify and map subpopulations that are sources, sinks, and vulnerable to extinction because of small size or poor connectivity. Long-term monitoring of trends in key

source and vulnerable subpopulations could help managers determine how various mortality factors affect metapopulation dynamics and how human developments may degrade habitat and linkages. In the absence of adequate monitoring data, management recommendations should be purposefully conservative.

Multi-jurisdictional Planning. Conservation planning for large carnivores must be conducted over vast spatial scales and must consider connectivity among local subpopulations (Noss et al. 1996). Land areas large enough to support a viable metapopulation of wolves will likely encompass a multitude of jurisdictions. Multi-agency coordination will be important for successful wolf restoration and management (Fritts and Carbyn 1995; Mladenoff et al. 1995). Establishment of regional planning authorities through appropriate means (e.g., legislative or administrative) should be encouraged and pursued.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of wolf ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about wolves to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

REVIEWERS

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Mountain Lion (*Puma concolor*)

*Please see accompanying Excel chart of Mountain Lion Habitat Characteristics in the folder titled “*Focal Species Habitat Characteristics Charts*” on the compact disk.

INTRODUCTION

The mountain lion (*Puma concolor*) is widely distributed throughout the NM Highlands planning area. It is primarily associated with mountainous areas and occupies habitats ranging from desert scrub to subalpine meadows. The mountain lion preys primarily on large ungulates, especially deer and elk. It requires large core areas for population persistence and safe linkages between patches of suitable habitat. The mountain lion is a protected big game animal in New Mexico. Hunting, regulated by the New Mexico Department of Game and Fish (NMDGF), occurs throughout most of the NM Highlands planning area. The statewide population is believed to be in excess of 1000 animals. Ecologically, the mountain lion belongs to a suite of top predators that serve to maintain ecosystem stability and promote overall biological diversity on a landscape scale.

DISTRIBUTION

Historic. The mountain lion was widely distributed throughout both North and South America ranging from southern Yukon to the southern tip of Chile (Lindzey 1987). Young (1946) described its range in North America as “practically transcontinental.” Mountain lions historically occupied suitable habitats that supported large ungulate prey species (i.e., deer and elk) throughout the NM Highlands planning area, except perhaps for the eastern plains (Bailey 1931; Findlay et al. 1975). However, they may have ranged some distance into the plains, especially along riparian corridors, which provided hiding and stalking cover (Bailey 1931; Young 1946).

Current. Mountain lions occupy “broken and mountainous” country in New Mexico from the Pecos River westward (Findlay 1975). They are expected to occupy most suitable habitats within the NM Highlands planning area unless human-caused mortality or habitat degradation has extirpated local populations. Evans (1983) identified the following regions of highest mountain lion densities within the NM Highlands planning area based upon kill data from the 1979-1983 sport hunting seasons: Sangre de Cristo Mountains including the Pecos Wilderness; and contiguous portions of the Apache, Cibola, and Gila national forests including the Gila

Wilderness in southwestern New Mexico. Regions of secondary concentrations identified by Evans (1983) included the Manzano Mountains south of Albuquerque and Mesa de las Viejas north of Albuquerque. Evans (1983) cautions that distribution maps based on hunter kill data may be distorted due to uneven hunter distribution and land ownership status, which may affect hunter access. Since 1985 the greatest harvest of mountain lions has been from the northwest and southwest mountains (NMDGF 1997).

Potential. Areas of approximately 200 km² (77 mi²) or greater within foothill or mountainous terrain with the following characteristics generally represent potentially occupied range for mountain lions: 1) viable populations of deer and/or elk; 2) suitable terrain and vegetative cover for hiding, stalking, and denning; 3) relatively low densities of humans and roads; and 4) either connected by linkages of suitable habitat to, or not more than 13 km (8 mi) from, areas of suitable mountain lion habitat of at least 2590 km² (1000 mi²). The large core areas are necessary to provide a source of dispersing mountain lions to recolonize smaller areas which are individually unlikely to sustain viable populations over long periods of time. The New Mexico Gap Analysis project (Thompson et al. 1996) identified about 44,372 km² (17,132 mi²) of potential mountain lion habitat within New Mexico. The NMDGF (1997) believes this to be a minimum estimate (also, Bill Dunn, NMDGF, personal communication).

HABITAT

General. Mountain lions are “habitat generalists” and occupy a wide range of habitats. Physical and structural characteristics of the terrain and vegetation are important in determining habitat suitability (Seidensticker et al. 1973; Lindzey 1987). The presence of large ungulate prey that is vulnerable to mountain lion predation is considered an essential habitat component (Bailey 1931; Seidensticker et al. 1973; Ackerman et al. 1982, cited in Lindzey 1987; Koehler and Hornocker 1991). Mountain lion habitat has been described as rough, broken foothills, canyons, and mountainous country in association with montane forests, shrublands, and piñon-juniper woodlands with available water (Findlay 1975; Fitzgerald et al. 1994). According to the NMDGF (1997), mountain lions occupy all habitats in New Mexico except large, open plains. No mountain lion home ranges extended more than 1-2 km (0.6-1.2 mi) from mountainous terrain in the San Andres Mountains, where adjacent desert basins did not support significant numbers of native ungulate prey (Sweaner et al. 2000). In summary, mountain lion habitat in New Mexico ranges from deserts to

subalpine meadows, with the most consistent feature being the presence of mule deer (NMDGF 1997).

Preferred. Mountain lions are incapable of catching swift prey by running them down; they hunt by hiding or stalking followed by pouncing or a short, swift pursuit (Young 1946; Hornocker 1970; Armstrong 1972; Seidensticker et al. 1973; Fitzgerald et al. 1994). Thus, hiding and stalking cover is an essential component of mountain lion habitat (Koehler and Hornocker 1991). Seidensticker et al. (1973) found that repeated kills were often made in the same areas, suggesting that some areas offer better opportunities for killing prey. Physical ruggedness of the terrain, which provides stalking cover and den sites, is a commonly-mentioned attribute of good mountain lion habitat (Bailey 1931; Young 1946; Seidensticker et al. 1973; Koehler and Hornocker 1991; Fitzgerald et al. 1994; Beier 1996). Mountain lion density appears to be directly related to the density of large ungulate prey species (Hornocker 1970; Seidensticker et al. 1973; Hemker et al. 1984; Logan et al. 1996; Pierce et al. 2000).

Van Dyke et al. (1986a) found that mountain lions established home ranges in areas that lacked recent (≤ 6 years) logging activity, had lower than average road densities, and few or no permanent, human disturbance sites. The road density in their Kaibab Plateau study area was 0.4 km/km² (0.65 mi/mi²). They suggested that areas experiencing permanent or repeated habitat alteration, even if the human presence is temporary, are less suitable for mountain lions; and that areas with continuing, concentrated human presence or residence are unsuitable, even if habitat impacts are minimal. Lions in areas near humans abandoned crepuscular activity periods and peak activity switched to night hours (Van Dyke et al. 1986a).

Den Sites. Beier et al. (1995) examined four den sites in the Santa Ana Mountains of southern California. Each site was in a small canyon with dense brush. None of the canyons were traveled regularly by other mountain lions. No animal trails, scat, remains of prey, or broken vegetation were evident at the den sites. Dens were in the bottoms of unscoured drainages within the upper one-fourth of the drainage. Beier et al (1995) note that none of the dens they observed were associated with a cave or rock outcrop, despite the availability of such features. Young (1946) described potential den sites as rocky caverns, uprooted trees, protected nooks, and dense thickets. Den entrances may be concealed by heavy brush or downed logs (Musgrave 1926, as cited in NMDGF 2000). K. Logan and L. Sweanor examined over 50 dens (K. Logan uses the term "nurseries") in the San Andres Mountains of New Mexico (K. Logan, personal communication). Dens occurred in boulder piles,

undercut ledges, and in dense shrub thickets. Shrubs provided effective lateral and overhead cover from weather and probably predators. Dens usually had entries formed by spaces between large rocks or boulders that kittens used for refuge. Normally there were one or more beds trampled into the soil surface, and where there were two or more, they were connected by pathways. They documented two instances in which mountain lion mothers used the same dens for consecutive litters (K. Logan and L. Sweanor, unpublished data).

FOOD HABITS / HUNTING BEHAVIOR

The diet of mountain lions consists mainly of ungulates (Young 1946; Hornocker 1970; Shaw 1983; Lindzey 1987; Logan et al. 1996; Wehausen 1996; Logan and Sweanor 2001, and many others). Ungulates within the NM Highlands planning area include elk (*Cervus elaphus*), mule (*Odocoileus hemionus*) and white-tailed (*O. virginianus*) deer, pronghorn (*Antilocapra americana*), bighorn sheep (*Ovis canadensis*), and javelina (*Tayassu tajacu*), but elk and mule deer are expected to be the primary prey of mountain lions (Hornocker 1970; Shaw 1983; Logan and Sweanor 2001). Important alternative prey may include jackrabbits (*Lepus* spp.), cottontails (*Sylvilagus* spp.), porcupines (*Erethizon dorsatum*), marmots (*Marmota flaviventris*), raccoons (*Procyon lotor*), and beavers (*Castor canadensis*) (Young 1946; Shaw 1983; Fitzgerald et al. 1994; Sweitzer et al. 1997; Logan and Sweanor 2001). Contrary to popular belief, mountain lions will scavenge carcasses of large mammals that die from other causes (Logan and Sweanor 2001); 9% of ungulates that died from other carcasses were scavenged by lions. Domestic livestock, especially calves and sheep, are also preyed upon by mountain lions (Shaw 1983; Hoffmeister 1986). Shaw (1983) believes that depredation by mountain lions is a function of the type of cattle management and relative abundance of other prey species, especially deer. Calves are particularly vulnerable if born in lion habitat. While cattle kills usually involve single calves or yearlings, surplus killing of domestic sheep (15-20 in one night) is more common.

Mountain lions lie in ambush or stalk their prey to within a few meters (yards) and pounce or rush in for the kill (Young 1946; Hornocker 1970; Armstrong 1972; Beier et al. 1995). They must search and find prey in situations that allow an undetected approach (Hornocker 1970; Seidensticker et al. 1973), which underscores the importance of stalking cover as a favorable habitat attribute. When large prey was killed, lions localized for 2-5 days within 4.2 km (2.6 mi) of the kill.

POPULATION DYNAMICS

Life History. Male mountain lions grow up to 2.44 m (8 ft) long and weigh up to 72.6 kg (160 lbs), and females reach 2.13 m (7 ft) in length and weigh up to 49.9 kg (110 lbs) (NMDGF 1997). Breeding age is about 2 years; gestation period is about 3 months; litter size averages about 3 kittens at the nursing stage and 2 at weaning; the interval between litters is about 1.5 years; age at independence from their mothers is about 1-1.5 years; age at dispersal from their natal home range is 1-2 years; mean annual productivity is about 1.5 kittens per adult female; few females reproduce past the age of 10-12 years; and the average female mountain lion has a lifetime production of 3-4 female young (Logan et al. 1996; Weaver et al. 1996). In southern New Mexico litters are born year round but peak from July to September, which corresponds with the birth of mule deer fawns and the rainy season (Logan et al. 1996).

Home Range. Mountain lions establish and defend home ranges or territories (Seidensticker et al. 1973). Home ranges of males are typically larger than those of females (Lindzey 1987; Sweanor et al. 2000). Male home ranges typically overlap a number of female home ranges. The degree of overlap among resident males is usually reported as minimal (Lindzey 1987; Anderson 1992; Logan et al. 1996), but greater overlap is reported from studies in which virtually all adult males are monitored (Logan and Sweanor 2001). Home ranges of females commonly overlap, but resident females generally avoid each other in the overlap areas (Hemker 1982). Logan et al. (1996) found greater overlap of male home ranges than of female home ranges, but explained that this may have been an artifact of the time frame of data collection during which the population of resident males may have been in a state of flux.

Size of mountain lion home ranges may depend upon various factors, such as topography, availability of stalking cover, ungulate movements and distribution, ungulate density and carrying capacity, and vulnerability of prey (Seidensticker et al. 1973). Home ranges averaged 187 and 74 km² (72 and 29 mi²), respectively, for adult male and female mountain lions in the San Andres Mountains of New Mexico, and territories of males overlapped territories of about 5 females and their cubs (Sweanor et al. 2000). Home range sizes in the San Andres Mountains varied from 59-336 km² (23-130 mi²) for males and 30-209 km² (12-81 mi²) for females; however, mean home range sizes were generally smaller than those reported for other areas (Logan et al. 1996). Lindzey (1987) presented a range of reported home range sizes of 125.5-826 km² (48-319 mi²) for males and 32.5-685 km² (13-264 mi²) for females, excluding very large home range sizes reported from Texas. Home range

size decreases and population density increases with increasing density of large ungulate prey (Quigley et al. 1989, cited in Weaver et al. (1996), provided adequate stalking cover is available (Weaver et al. 1996). Logan et al. (1996) noted that home ranges of mountain lions in the San Andres Mountains were larger where inclusions of large expanses of unsuitable habitat characterized the landscape.

Causes of Death. Logan et al. (1996) presented a detailed synopsis of causes of mountain lion mortality, which include legal and illegal shooting and trapping by humans, intraspecific aggression (lions killing lions), vehicle collisions, research-related capture injuries; disease; prey-inflicted injuries; and starvation. In the un hunted population studied by Logan et al. (1996), 44% of kitten deaths, 100% of subadult deaths, and 52% of adult deaths were caused by male mountain lions. Humans were the major cause of death in hunted mountain lion populations (Logan et al. 1996, citing many authors). Logan and Sweanor (2001) argue that hunting of mountain lions by humans may significantly alter the natural selection process and reduce the genetic health of hunted populations. Disease, especially septicemic plague, was an important cause of death of adult mountain lions (17%) on the San Andres Mountains (Logan et al. 2001). Depredation control was the leading cause of death for a mountain lion population in southeastern Arizona (Cunningham et al. 2001).

Self-Regulation. Seidensticker et al. (1973) and Hornocker (1970) concluded that the presence of adult resident mountain lions was the primary factor limiting the density of resident males and females; that resident adults with established home ranges comprised the breeding population; and that the resident lion breeding population appeared to be below the level set by prey density. In contrast, Logan and Sweanor (2001) argue convincingly that social behavioral attributes of mountain lion populations do not self-regulate populations, but, rather, that lion populations are ultimately regulated by prey abundance and availability.

Population Structure and Viability. Sweanor et al. (2000) found that mountain lions in southern New Mexico exhibit a metapopulation structure characterized by subpopulations separated by expanses of unsuitable habitat linked by dispersing lions. Males dispersed about 3 times farther than females and were more likely to cross large expanses of unsuitable habitat. Males were probably most responsible for genetic exchange between subpopulations. Wide-scale losses of breeding females can significantly reduce mountain lion populations because it substantially reduces the number of available female recruits to replace lost females (Lindzey et al. 1992, cited in Weaver et al. 1996).

Population viability modeling in southern California by Beier (1996) predicted that patches of suitable habitat in the range of 1000-2200 km² (386-849 mi²) are required to ensure the persistence of a population of mountain lions for at least 100 years. He stressed that these should be considered “minimum areas”, and that much larger areas or periodic immigration or translocation of lions would be required for longer term persistence of the population. Smaller patches of mountain lion habitat have higher probabilities of population extinctions and must rely on linkages to larger, viable populations for persistence. Emphasizing the importance of immigration to small populations, Beier (1993) predicted that a mountain lion population could persist in an area as small as 600-1600 km² (232-618 mi²) provided up to 3 males and 1 female immigrated to the area per decade. Beier (1993) cautions that his model results apply to mountain lion densities observed in the Santa Ana Mountains of California (1 adult per 100 km² [39 mi²]), and should be adjusted for local adult lion densities and other parameters elsewhere. Mountain lion densities observed on the San Andres Mountains in New Mexico were about 2 adults per 100 km² (Logan et al. 1996).

Smaller patches of suitable habitat may also be important within linkages between larger patches, especially for dispersal of females, whose genes may flow in a “stepping stone” pattern relayed by subsequently dispersing offspring (Beier 1996). Beier (1995) observed dispersing mountain lions (8 males and 1 female) using a series of small, transient home ranges during dispersal in southern California. Refugia of large patches (>1000 km² [386mi²]) of suitable mountain lion habitat, where mountain lions are not subjected to human-caused mortality, would support persistent source populations for rescuing (by producing emigrants) declining populations in smaller surrounding patches (Beier 1996; Weaver et al. 1996, Sweanor et al. 2000). The success of refugia as population sources could be increased by enhancing habitat productivity and eliminating human harvest within refugia (Weaver et al. 1996). The distance between refugia should be informed by the dispersal patterns and capabilities of mountain lions (Weaver et al., 1996; Sweanor et al. 2000), especially females because they disperse only about a third as far as males.

Population Status. Predator control programs began in the late 1800s, and mountain lion populations in the West reached all-time lows in the mid 1900s (Hornocker and Quigley 1987). Bailey (1931), using information obtained from J. Stokely Ligon, estimated the statewide population at 400 mountain lions in 1917. Berghofer (1967, cited in NMDGF 1997) estimated the population at 350 animals in the mid 1960s. Regulation of mountain lion harvests following its protection

as a “game animal” in 1971 and the persistence of relatively large tracts of suitable habitat and pathways for dispersal among them has allowed populations to increase (NMDGF 1997). The average annual harvest in New Mexico from 1985-1997 was 105 mountain lions. Logan et al. (1996) estimated the statewide mountain lion population at 4.3 adults / 259 km² (100 mi²) of suitable habitat for a population estimate of 737 adults and 531 subadults (total = 1268 mountain lions). Continued habitat fragmentation and disruption of dispersal linkages between habitat patches will cause reductions in mountain lion populations by increasing extinction risks within isolated subpopulations.

MOVEMENTS

Dispersal. Mean dispersal distances in the San Andres Range (NM) were 13 km (8 mi) for 21 females and 116 km (72 mi) for 13 males (Sweanor et al. 2000). Average dispersal distance for both sexes of mountain lions derived from several studies was 85 km (53 mi) (Weaver et al. 1996). Anderson et al. (1992, cited in Logan et al. 1996) summarized dispersal distances for 65 mountain lions as follows: 29-274 km (18-170 mi) for males and 9-140 km (6-87 mi) for females. Generally, males disperse about 3 times farther than females. Sweanor et al. (2000) observed that no breeding adults with established home ranges migrated between subpopulations. Dispersing mountain lions are typically not sexually mature. Female mountain lions often remain in their natal home range, resulting in the formation of matriline. Similar natal philopatry has not been documented for males, except in Florida (Maehr 1997, cited in Sweanor et al. 2000).

Barriers to Movement. Mountain lions dispersing from the San Andres Mountains crossed up to 100 km (62 mi) of unsuitable habitat, including two animals that crossed the Rio Grande; and movements through inhospitable habitat were generally direct and of short duration (Sweanor et al. 2000). Fitzgerald et al. (1994) make the general statement that few physical barriers seem to limit movements of mountain lions; observations by others are contradictory. Vehicle collisions were the single greatest cause of mortality for resident mountain lions in fragmented habitats in southern California (Beier 1995) and Florida (Maehr et al. 1991, cited in Ruth et al. 1998). However, Beier (1995) monitored one lion that successfully used an underpass to cross a freeway 16 times in 8 months. Sweanor et al. (2000) documented seven successful surface crossings of 4-lane U.S. Highway 70 prior to its expansion to 6 lanes. Following the widening, only two attempted crossings were documented, both were struck and killed by vehicles. No

radio-collared lions with home ranges abutting the newly widened highway were documented crossing it. Reported barriers to successful movement and dispersal of mountain lions include hard-surfaced roads, multi-lane highways, urban development, and artificial lighting at night (Beier 1993; Beier 1995; Van Dyke et al. 1986b; Ruth et al. 1998; Sweanor et al. 2000).

Use of Linkages. Beier (1995), working in an area fragmented by urbanization, concluded that dispersing mountain lions will use habitat linkages that are located along natural travel routes, have ample woody cover, have underpasses integrated with roadside fencing at high-speed road crossings, lack artificial outdoor lighting, and have less than 1 human dwelling per 16 ha (40 acres). Presumably, similar features would promote movement across desert bajadas between core areas, but field work on this issue is needed. Small patches of suitable habitat (e.g., 130 km² [50 mi²]) enhance linkages between larger patches, even though they may not be continuously occupied by mountain lions (Beier 1996). Use of highway underpasses by mountain lions has been documented (Beier 1993; Foster and Humphrey 1995). For large carnivores, dispersal is mainly an issue of circumventing barriers to long-range movements (e.g., highways or developed areas) and minimizing human-caused mortality (e.g., shooting, trapping, or being struck by vehicles) (Beier 1993; Noss et al. 1996).

ECOLOGY

Effects on Prey. Mountain lion predation on large ungulates serves to dampen and protract population oscillations, maintain ungulate populations at or below the carrying capacity of their habitat, and, thus, prevent serious damage to ecosystems, particularly at the primary producer (plants) trophic level (Hornocker 1970; Logan et al. 1996; Logan and Sweanor 2001). A consequence of this relationship may be exacerbation of population declines and slower recovery of ungulate populations from extreme low levels (Logan et al. 1996; Ballard et al. 2001; Logan and Sweanor 2001). In specific, local situations predation by mountain lions may significantly reduce or eliminate small populations of bighorn sheep and preclude successful restoration efforts (Wehausen 1996). Following a decline in the mule deer population in the Great Basin of Nevada, mountain lion predation nearly extirpated a local population of porcupines (Sweitzer et al. 1997). In most (perhaps all) cases in which mountain lions endanger small populations, the impact of predation has been exacerbated by disease (e.g., scabies and pneumonia in bighorn – Logan and Sweanor 2001), woody plant invasion due to overgrazing or fire suppression (Sweitzer

et al. 1997), or artificially high lion populations subsidized by year-round livestock operations (E. Rominger, NMDGF, personal communication). Short-term control of mountain lions may be necessary as an emergency measure to prevent extinction of some populations of bighorns, pronghorn, or other ungulates, but long-term conservation of these ungulates should follow a holistic approach.

Mountain lions tend to prey disproportionately on very young and old members of ungulate populations (Hornocker 1970; Logan et al. 1996; Kunkel et al. 1999). Prey with injuries or in weakened condition may also be more vulnerable to predation by mountain lions (Hornocker 1970; Kunkel et al. 1999). Mountain lions may have a preference for deer over elk (Kunkel et al. 1999).

Hornocker (1970) and Seidensticker et al. (1973) stated that mountain lions influence the movement and distribution of their ungulate prey across the landscape, and speculated that this could reduce localized overuse of food resources and promote ecosystem stability. However, these studies did not present data that documented any redistribution of ungulates, and Logan and Sweanor (2001) did not observe this phenomenon on their San Andres Mountains study area.

Mountain lions killed more fawns and old female deer and fewer 1.5-2.5-year-old deer than did hunters (Kunkel et al. 1999). The same pattern was observed for elk. Thus, the evolutionary pressure of mountain lion predation on its ungulate prey species is not replaced by sport hunting humans.

Effects on Mesopredators. Mountain lions kill bobcats and coyotes when they associate near kills made by lions (Boyd and O’Gara 1985, cited by Koehler and Hornocker 1991; Koehler and Hornocker 1991; Beier and Barrett 1993; Logan and Sweanor 2001). This “interference competition” may influence the use of ecosystem resources by bobcats and coyotes as well as the niche relationships and overall structure of the carnivore community (Koehler and Hornocker 1991).

Ecological Effects. Modern ecological theory and empirical evidence strongly suggest that top predators, such as mountain lions, exert a controlling (“top-down”) influence over entire ecosystems through a cascade of regulatory effects across the various trophic levels of the ecosystem (Terborgh et al. 1999:41; Miller et al. 2001). Potential immediate effects of removing top predators include increases in the abundance of large herbivores (e.g., deer and elk) and mesopredators (e.g., bobcats and coyotes) (Terborgh et al. 1999; Dugelby et al. 2001; Miller et al. 2001). These proximate ecological changes may generate a

subsequent cascade of effects operating through ecosystem processes such as herbaceous consumption, predation, competition, and behavioral exclusion ultimately resulting in ecosystem simplification, loss of ecosystem stability, and loss of biological diversity (Terborgh et al. 1999; Miller et al. 2001). Following their comprehensive study of mountain lions in a Chihuahuan Desert mountain range, Logan and Sweanor (2001:361-362) concluded that “top-down and bottom-up forces interacted simultaneously and were modified by weather to shape large-mammal community dynamics.”

JUSTIFICATION AND FOCAL VALUE

Umbrella. Because of its need for large territories and use of a variety of habitats, protection of habitat for a viable population of mountain lions will protect interconnected habitats for many other species (Noss et al. 1996). Low population density due to large territorial requirements, especially for males, renders mountain lion populations particularly sensitive to habitat fragmentation (Beier 1996).

Keystone. By preying on deer, elk, and other ungulates and killing mesopredators such as coyotes and bobcats, mountain lions may contribute to the top-down regulation of ecosystems by the suite of top predators present. Mountain lion predation may 1) dampen oscillations of prey populations and cause a more even distribution of prey across the landscape thus reducing potential habitat damage; 2) enhance biological diversity by contributing to the suppression of mesopredator populations; and 3) promote ecosystem stability. Empirical evidence that mountain lions play a keystone role in ecosystems is largely lacking (Noss et al. 1996), thus the suggestion of their contribution to such a role is based largely upon ecological theory (Miller et al. 2001).

Flagship. Large predators including the mountain lion fascinate a wide range of people. Human perceptions and attitudes toward mountain lions are generally more positive overall than for some other large predators, such as the wolf (Kellert et al. 1996). Conservation efforts focusing on mountain lions should generate popular support.

Wilderness Quality Indicator. Mountain lions tend to avoid or become extirpated from areas characterized by the presence of roads, human activities (e.g., logging), and human occupation. Thus, large core areas with wilderness qualities are required for the protection and preservation of viable populations of mountain lions.

MANAGEMENT RECOMMENDATIONS

Establish Refugia. Perhaps the most effective management action for the mountain lion would be the establishment of large strategically located refugia, where mortality of mountain lions would be limited to natural causes. Areas in the range of 1000-3000 km² (386-1158 mi²) or larger characterized by suitable mountain lion habitat should be considered for refuge designation. Measures to increase native ungulate prey populations (such as restoration of natural fire regimes or full protection of ungulates) within these refugia would enhance their value for mountain lion conservation. Protected refugia would serve as reservoirs, contributing to the restocking of adjacent populations depleted by hunting or other causes (Lindzey 1987; Logan et al. 1996; Logan and Sweanor 2001).

Protect and Restore Linkages. Few remaining patches of suitable mountain lion habitat are large enough to guarantee long-term persistence of their mountain lion populations (Logan and Sweanor 2001). Movement of lions among subpopulations separated by unsuitable habitats is essential to the long-term genetic health and viability of the larger metapopulation of mountain lions in the NM Highlands area. Human-caused habitat fragmentation and barriers to animal movements are the greatest threats to mountain lion populations and the most difficult threats to prevent. Habitat linkages will best serve mountain lions if they are located along natural travel routes, have ample woody cover, include underpasses integrated with roadside fencing where high-speed roads cross corridors, lack artificial outdoor lighting, and have less than 1 human dwelling unit per 16 ha (40 acres).

Land management agencies should strive to restore nonfunctional or diminished linkages and protect existing linkages where possible. Development authorities and interests should be made aware of the critical importance of habitat linkages.

Zone Management. Logan and Sweanor (2001:385-388) propose a “zone management” scheme for managing, protecting, and conserving mountain lion populations in New Mexico. We support this approach and recommend that the New Mexico Department of Game and Fish implement such a management scheme.

Limit Harvest of Adult Females. Persistence of mountain lion populations is particularly sensitive to the presence of breeding females. Since females tend to disperse over much shorter distances than males, the potential for immigration of females into isolated habitat patches may be the key to re-establishment of extirpated subpopulations. Adult females may

have dependent kittens at any time of the year (Logan et al. 1996). Orphaned cubs less than 9 months old usually die, and older dependent cubs are more likely to die when their mother is killed (Logan and Sweanor 2001). Thus, the prevention of over-hunting of the adult female segment of established lion populations will enhance population persistence.

Quota systems that terminate hunting seasons when a predetermined number of females have been killed should be implemented (Ross et al. 1996; Lindzey 1987; Logan and Sweanor 2001). Establishing quotas on the killing of females will reduce the potential for over-hunting adult females and promote the persistence of lion populations, especially smaller, more isolated populations.

Limit Take of Mountain Lions in Small Habitat Patches.

Game managers should be particularly sensitive to the potential for over-hunting mountain lions, especially females, in small habitat patches. Hunting regulations should contain measures to prevent the extirpation of small, isolated populations. Measures may include quota systems as described earlier or eliminating hunting, especially on areas that are critical stepping stones within important linkages between larger populations. There should be no hunting in any habitat patch that is truly isolated, and no killing of females in any patch smaller than 400 km² (150 mi²).

Limit Mountain Lion Control. Wildlife and land management agencies should encourage livestock husbandry practices that reduce depredation by mountain lions. Non-lethal solutions to depredation problems should be actively sought and seriously considered. Lethal control of problem mountain lions should be limited to identified problem animals. Government assistance in controlling problem predators should be conditioned upon subsequent and feasible corrective action by livestock operators. Controlling mountain lion populations to increase big game populations has not been demonstrated to be effective, except in very specific, localized situations (e.g., bighorn sheep restoration).

Monitoring. Admittedly, mountain lion populations are difficult and expensive to monitor at a level necessary to inform management decisions. We believe that with large core areas (including some refugia), aggressive protection of habitat linkages, and the conservative hunting recommendations mentioned above, no expensive monitoring of mountain lions would be necessary. In areas where fragmentation or the potential for over-hunting is an issue, Sweanor et al. (2000:807) recommend that “[a]gencies that manage cougars in fragmented habitats need to identify and map subpopulations that are

sources, sinks, and vulnerable to extinction because of small size or poor connectivity. Long-term monitoring of trends in key source and vulnerable subpopulations could help managers determine how human off-take affects metapopulation dynamics and how developments may degrade habitat and corridors. Corridors linking patches of interest could be identified by radiotracking movements of dispersing subadults.” Beier (1996) also recommends the use of radio telemetry to identify important linkages. Making a “best guess” from a map often misses the mark. In the absence of adequate monitoring data, management recommendations should be purposefully conservative.

Multi-jurisdictional Planning. Beier (1996:318) concluded that the main obstacle preventing the conservation of a mountain lion metapopulations in his study area is “the lack of regional planning authority.” He described a 2 km (1.2 mi) long travel route that fell within the jurisdictions of three counties and two incorporated cities. Land areas large enough to support a viable metapopulation of mountain lions will likely encompass a multitude of jurisdictions. Establishment of regional, multi-jurisdictional planning authorities through appropriate means (e.g., legislative or administrative) should be encouraged and pursued.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of mountain lion ecology in order to establish appropriate policies and make sound management decisions (Logan and Sweanor 2001). In addition, the public needs accurate information and knowledge about mountain lions to inform their opinions and values and their understanding of appropriate management measures (Logan and Sweanor 2001). Knowledge is the key to informed conservation advocacy and actions by both agencies and the public.

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Black Bear (*Ursus americanus*)

INTRODUCTION

The black bear is the largest extant member of the order Carnivora in the NM Highlands planning area. Contrary to its taxonomic classification, however, it is mostly herbivorous in its dietary habits. Black bears are common and widely distributed in large tracts of forests and woodlands and immediately adjacent shrublands. They require large areas of suitable habitat and safe, densely forested linkages among habitat patches for population viability. If necessary, black bears migrate seasonally in search of high-quality foods. The black bear is managed as a game species by the New Mexico Department of Game and Fish, and most populations within the NM Highlands planning area are hunted. Populations in the NM Highlands planning area are relatively secure. The black bear is adversely affected by human encroachment and habitat fragmentation, and its popularity with both hunters and nonconsumptive users fosters considerable public interest and support for nature conservation.

A comprehensive, 8-year field investigation of black bear ecology in New Mexico was recently completed (Costello et al. 2001). Transmitter collars were placed on 239 bears. Both field study sites were within the NM Highlands planning area—the Sangre de Cristo Mountains in the north and the Mogollon Mountains in the south. We are most fortunate to have current site-specific information for the black bear, and we rely heavily upon the Costello et al. (2001) report for the following information unless otherwise attributed.

DISTRIBUTION

Historic. Historically, black bears were widely distributed throughout all major forested regions (deciduous and coniferous) in North America (Hall 1981). Fossil evidence suggests that they were not common in open habitats, such as grasslands, shrublands, and desert areas. Nowak (1991) speculates that black bears avoided open habitats because of a lack of trees, which provided a means of escaping predation by grizzly bears (*U. arctos*). In support of this theory, Nowak (1991) notes documented range extensions of black bears following the extirpation of grizzly bears. In the early 1900s, black bear distribution in New Mexico was greatly reduced by unregulated hunting, trapping, and use of poisons by both private individuals and government predator control agents. In 1917, J. Stokely Ligon estimated the statewide population of black and brown bears at

157 (Bailey 1931). Subsequent estimates for black bears were 660 in 1925, 3000 in 1967, and nearly 6000 today (Costello et al. 2002). Black bears were first protected as a game animal by the NMDGF in 1926, and the taking of black bears has been regulated continuously since 1927.

Current. Black bears are currently distributed throughout the wooded foothills and coniferous forests of the major mountain ranges in NM Highlands planning area—Sangre de Cristo, San Juan, Jemez, Zuni, Sandia, Manzano, Magdalena, San Mateo, and Mogollon Mountains and associated ranges.

Potential. Little potential exists for significant range expansion for black bears in the NM Highlands planning area.

HABITAT

General. In New Mexico black bears showed a strong tendency to select closed forest/woodland habitat types, and more than 80% of all bear locations were in these types. Shrublands received limited use and open grasslands and woodlands (e.g. savannas) were generally avoided by black bears, except within 500 m (0.3 mi) of closed-canopy habitat edges. These observations were consistent with other studies of black bear habitat use (Lindzey and Meslow 1977; LeCount and Yarchin 1990, both as cited in Costello et al. 2001). Habitat use may vary seasonally depending upon the availability and location of preferred foods. Harding (2000) observed substantial use of sage (*Artemisia* spp.) steppe habitats in Utah during spring and early summer.

Black bears exhibit a behavioral avoidance of roads, especially in areas open to hunting (Brody and Pelton 1989; Powell et al. 1996). Bears may react to increases in road density by shifting home ranges to areas of lower road densities (Brody and Pelton 1989; VanderHeyden and Meslow 1999). Roads increase mortality through the facilitation of legal and illegal killing and vehicle collisions (Brody and Pelton 1989; Powell et al. 1996; Powell et al. 1997; Pelton 2000). In North Carolina, 50% and 75% of legally hunted black bears are killed within 0.8 km (0.5 mi) and 1.6 km (1 mi) of roads passable by four-wheel-drive vehicles (Powell et al. 1997). Kill rates in the Southwest may be higher because of increased visibility within habitats here. The New Mexico Department of Game and Fish (2000) recommends that road densities not exceed 0.8 km/km² (1.3 mi/mi²) in black bear habitat. Roadless areas smaller than average bear home ranges likely have insufficient escape value for bears (Powell et al. 1997). Powell et al. (1997) state that large roadless areas are an essential component of suitable bear habi-

tat in the Southern Appalachian Region. We suspect that roadless areas are equally important to bear survival and habitat suitability in the NM Highlands planning area. On the other hand, lightly used or seasonally closed roads may serve as travel corridors and seasonal feeding sites for bears (Pelton 2000).

Preferred. Preferred habitats of black bears in the NM Highlands planning area include subalpine coniferous (spruce-fir) forests (*Picea engelmannii*, *Abies lasiocarpa*, *A. concolor*, *Pseudotsuga menziesii*); subalpine broadleaf (aspen) forests (*Populus tremuloides*); upper (*Pseudotsuga menziesii*, *A. concolor*, *Picea pungens*) and lower (*Pinus ponderosa*, *Pinus edulis*, *Juniperus* spp., *Quercus gambelii*) Rocky Mountain montane coniferous forests; and Rocky Mountain/Great Basin closed coniferous (pinyon-juniper) forest (*Pinus edulis*, *Juniperus* spp.). Shrublands and grasslands adjacent to these closed-canopy forest types provide additional habitat, and may be especially important in spring and early summer when bears forage primarily on grass and ants (Harding 2000). Forest openings produce important foods for bears (Boileau et al. 1994; Verts and Carraway 1998; Pelton 2000). However, these openings should be small and occur within a matrix of large contiguous forested tracts with minimum human disturbances (Pelton 2000).

Water is an important component of black bear habitat (Verts and Carraway 1998; Vanderheyden and Meslow 1999). Five additional components of preferred habitat are: escape cover; sources of hard or soft mast foods in fall; spring and summer feeding areas; movement corridors; and winter denning habitat (Pelton 2000). Chaparral scrub provides important escape cover in Southwestern habitats (Pelton 2000). The absence or failure of mast crops significantly reduces subsequent reproductive performance (often causing population-wide reproductive failure) of female black bears (Pelton 2000; Costello et al. 2001). Spring and summer foods are important to the recovery of black bears emerging from winter dens, especially females with cubs. Grasses, berries, and insects, especially ants, bees, and wasps, provide important nutrients at this time of year (Verts and Carraway 1998; Harding 2000; Pelton 2000). Seasonally available foods are often widely distributed throughout bear home ranges. Movement corridors with thick cover along ridge tops, saddles, side drainages, streams, and rivers provide safe passage to important food resources (Pelton 2000).

Den Sites. Over half of 390 dens visited during the New Mexico study were either rock cavities or excavations under rocks. About one-third of the dens were natural tree cavities or excavations under trees. The most common habitat types at den sites were mixed conifer forests (45%), pinyon-juniper woodlands (21%), and spruce-fir forests (13%). Most tree cav-

ity dens (83%) were in mixed conifer habitats. All tree cavity dens examined in the New Mexico study were used by females. This finding is consistent with Pelton's (2000:399) statement that "female bears prefer cavities in large, standing, dead or live trees." Of dens in pinyon-juniper habitats, 88% were associated with rocks. The mean slope at den sites was 28 degrees and most dens were located at the upper (42%) or mid (37%) areas of slopes. Only 8% of dens were on flat sites—ridge tops or valley bottoms. Ground dens are associated with thick understory cover (Pelton 2000). Dens are sometimes re-used, but most black bears selected new den sites each year of the New Mexico study.

Bears are highly likely to abandon and change dens following human disturbance (Goodrich and Berger 1994). Periods of activity during winter (when food is generally unavailable) may increase body attrition resulting in reproductive failure, starvation, or poor nutritional/physical condition upon emergence from dens. Poor condition may predispose black bears to other mortality factors. Secure den sites can reduce the effect of human disturbance upon denning black bears (Pelton 2000).

FOOD HABITS / FORAGING BEHAVIOR

Black bears in the New Mexico study mostly consumed plant matter. From den emergence to mid summer the diet was dominated by grasses (*Poa* spp., *Festuca* spp., *Muhlenbergia* spp., *Piptochaetium* spp.), sedges (*Carex* spp.), and forbs (vetch, *Vicia* spp., peavine, *Lathyrus* spp., golden pea, *Thermopsis rhombifolia*) Ants (Formicidae) and soft mast (e.g., fruits and berries of junipers, squawroot, *Conopholis alpina*, gooseberry, *Ribes* spp., hawthorn, *Crataegus* spp., squawbush, *Rhus trilobata*, wild plum, *Prunus americana*, kinnikinnick, *Arctostaphylos uva-ursi*, Wright silktassel, *Garrya wrightii*, and prickly pear cactus (*Opuntia* spp.) increased in importance from mid summer to the availability of hard mast (acorns and pinyon nuts), and both hard and soft mast were consumed from mid September to den entrance. Except for ants, animal foods were not important in the diets of black bears in New Mexico.

Carrion is eaten by bears and may be an important food item in the spring (Fitzgerald et al. 1994; Verts and Carraway 1998; Pelton 2000). Newborn ungulates (e.g. deer, *Odocoileus* spp., and elk, *Cervus elaphus*) are sometimes preyed upon by black bears but are able to escape such predation attempts by the age of about 2 weeks (Fitzgerald et al. 1994; Linnell et al. 1995; Verts and Carraway 1998; Pelton 2000). Domestic livestock are sometimes killed by black bears (Fitzgerald et al. 1994). While black bears are classified as carnivores, they function eco-

logically as omnivores; but mostly consume vegetable matter, especially herbaceous material, fruits, seeds, and nuts (Pelton 2000).

Black bears can become a nuisance in apiaries, fruit orchards, grain crops, garbage cans or dumps, and at bird feeders; and conflicts are common where bear habitats meet human settlements (Pelton 2000). Bears often peel the bark from trees (e.g. Douglas fir, alpine fir, corkbark fir, and limber pine) to feed on the underlying cambium layer, causing damage and often death of the tree (LeCount 1986; Nowak 1991; Verts and Carraway 1998). This feeding behavior may be a response to the lack of more preferred foods.

POPULATION DYNAMICS

Life History. Black bears are the largest extant carnivore in New Mexico. Adult males and females weigh an average of 104 kg (229 lbs) and 64 kg (141 lbs), respectively. Black bears exhibit black and brown color phases, and brown-phase bears can appear dark brown, light brown, cinnamon, and blonde. In the New Mexico study areas, 75% of bears were brown phase.

Except for females with cubs or yearlings, black bears tend to be solitary in their social organization (Fitzgerald et al. 1994) and polygynous in their reproductive habitats (Pelton 2000). Adult males and females associate solely for the purpose of mating (Pelton 2000). Home ranges of males overlap considerably and include the home ranges of one or more females, which may also overlap (Pelton 2000; F. Lindzey, personal communication). Black bears tend to congregate and exhibit intraspecific tolerance in areas where food is abundant (e.g., berry patches and garbage dumps) (Verts and Carraway 1998).

Black bears hibernate, even in the southern reaches of the NM Highlands planning area. In New Mexico, most black bears enter hibernation dens between mid October and mid November and emerge in April and May.

Most female black bears reach reproductive maturity at 3 years of age (range 1-5). The observed mean age at first reproduction for female bears in New Mexico was 5.7 years. Most breeding occurs in June and July; implantation of the embryo is delayed; and cubs are born in hibernation dens in late January or early February. Litter sizes range from 1 to 3 cubs in New Mexico (mean = 1.8), and the mean natality rate for females ≥ 4 years old was 0.77 cubs/female/year. Cubs remain with their mothers for about 16 to 18 months and den with their mothers the first winter following their birth. Prolonged parental care of

cubs sets the birth interval for females at a minimum of 2 years. Reproductive success of black bears is strongly influenced by the previous season's production of acorns and juniper berries. Failed mast production for two consecutive years usually results in complete reproductive failure.

Black bears may live more than 20 years, but few live longer than 10-12 years in the wild (Fitzgerald et al. 1994; Pelton 2000). In the New Mexico study, the oldest female documented was 27 years old and the oldest male was 23. Two females produced litters at the age of 22 years. Ages of black bears in the New Mexico study were estimated from cementum annuli and, thus, were not known ages.

Population Density. Costello et al. (2001) estimated black bear densities (excluding cubs) to be 17/100 km² (39 mi²) in the Sangre de Cristo Mountains (northern) study area and 9.4/100 km² (39 mi²) in the Mogollon Mountains (southern) study area. It should be noted that the northern study area was not hunted during the study, and that the southern study area was hunted. Data collected during the study were not sufficiently rigorous to determine if hunting affected bear densities. Lower bear densities were expected in the southern study area because home ranges were generally larger there, implying less productive habitat. Comparable black bear densities were reported by Fitzgerald et al. (1994) for Colorado. However, black bear population densities are expected to vary with habitat quality (L. Harding, personal communication).

Home Range. Mean primary home ranges (excludes long-range movements) of male black bears were 3-5 times larger than female home ranges. In the New Mexico study, female primary home ranges averaged 24 km² (9.3 mi²) in the Sangre de Cristo Mountains study area and 43 km² (16.6 mi²) in the Mogollon Mountains study area. Primary home ranges of males averaged about 131 km² (50.6 mi²) in both study areas. Increased movements occurred in the fall when bears were in search of highly-preferred hard mast foods (e.g., acorns) prior to hibernation. When long-range movements were included in home range calculations, some home ranges exceeded 3000 km² (1,160 mi²). More long-range movements were suspected (bears not located for >45 days) in the southern study area than in the northern study area. Costello et al. (2001:75) state that the larger home range estimates resulting from inclusion of long-range movements "may actually reflect the potential areas used by SSA [southern study area] bears." The authors further speculate that "the more arid conditions of the SSA, coupled with livestock grazing, may limit the availability of grasses to bears, and compel individuals to search more widely for other foods" (Costello et al. 2001:83). In the New Mexico

study, adult bears of both sexes and subadult females exhibited a “high degree of home range fidelity” (Costello et al. 2001:86).

Causes of Death. Most mortality of adult black bears is human-caused by legal hunter kills, illegal kills, depredation kills, and automobile collisions (Fitzgerald et al. 1994). Legal kills of bears by hunters tend to be higher in years with shortages of natural foods, presumably because bears travel farther and more of the time and often leave secure habitats in search of food. Natural causes of mortality include predation (by other black bears, grizzly bears, wolves, *Canis lupus*, and mountain lions, *Puma concolor*), diseases and parasites (not considered a major factor), and starvation; but natural mortality rates for adult bears are low compared to human causes (Paquet and Carbyn 1986; Schwartz and Franzmann 1991; Mattson et al. 1992; Smith and Follman 1993; Pelton 2000; Costello et al. 2001). Mortality among yearling bears was mostly from natural causes, but human causes were also documented during the New Mexico study. Mortality factors for cubs include predation, automobile collisions (Costello et al. 2001), and infanticide by adult males of unrelated cubs (Pelton 2000).

Cub mortality is high and was 44% in an area where roads were closed in Colorado (Fitzgerald et al. 1994). The next highest mortality (>35%) occurs among yearling and subadult bears from the time they leave their mothers until they have successfully established home ranges of their own (Pelton 2000). As bears age, mortality rates decline; however, the mortality rate of males (26%) is about double that of females (17%) (Pelton 2000).

Population Structure and Viability. In the New Mexico study areas, male:female ratios for all age classes were approximately 1:2; adult females comprised 35% of the population and adult males about 20%. The remaining 45% of the population was comprised of subadult (2–4 years old) and yearling bears. Cubs were not considered as recruited into the population unless they survived their first year. The fragmented arrangement of suitable habitat in the NM Highlands area favors a metapopulation structure for black bear populations. Metapopulations are comprised of several smaller interconnected subpopulations whose individual viability is critical to the long-term persistence of the metapopulation.

For identifying small tracts of suitable black bear habitat in New Mexico, Costello et al. (2001:95) designated a “minimum sustainable population” as 50 bears, but their use of this terminology was not intended denote population viability (C. Costello, personal communication). A more appropriate interpretation of this term would be a subpopulation inhabiting a

relatively isolated tract of suitable habitat with potential connectivity with larger habitat blocks. According to the authors, such a population could be supported within about 300 km² (116 mi²) of contiguous suitable habitat in northern portions of the NM Highlands planning area and about 500 km² (193 mi²) in southern portions. They also consider patches of suitable habitat sufficient in size to support 1–2 bears (>20 km² or 7.7 mi²) as potentially important habitat, provided these patches lie within 15 km (9.3 mi) of patches greater than 300 km² (116 mi²) in size.

Powell et al. (1996) studied black bears residing both inside and outside the approximately 260-km² (100-mi²) Pisgah Bear Sanctuary in North Carolina. They determined that the sanctuary provided increased protection for bears residing within it and provided dispersing bears for hunters outside it, but that its population may have been declining. Poaching occurred along roads within the sanctuary. No transmitter-collared male bears had home ranges that were totally confined within the boundaries of the sanctuary. Powell et al. (1996) concluded that sanctuaries are appropriate for managing black bears, but that sanctuary sizes need to be larger than the one they studied and that roads should be eliminated within sanctuaries. The beneficial effect of road elimination and increased size of sanctuaries for bears derives from decreased human access to bears and their habitat (Powell et al. 1996). Similarly, Rudis and Tansley (1995) recommended the retention or restoration of an extensive complex of interconnected blocks of remote (i.e., inaccessible) forests for the protection and conservation of black bears in Florida.

Because ideal situations rarely exist in nature, the “genetically effective population size” (N_e) is probably always smaller than the actual census size (N_c) of the population (Meffe and Carroll 1997:172). A genetically effective population is generally defined as an ideal, stable population with randomly mating individuals, even sex ratio, equal birth rates among females, and non-overlapping generations (Meffe and Carroll 1997). Using population age-sex data presented by Costello et al. (2001) and a formula for calculating N_e presented by Meffe and Carroll (1997:173), the genetically effective population size of black bear populations in the NM Highlands planning area is approximately one-half the actual number of bears present, excluding cubs.

A major goal of the NM Highlands conservation plan is to preserve or restore evolutionary processes in natural systems, which requires long-term population viability for resident species. Genetic variation within and among individuals comprising populations of animals is the currency of evolution. In short, genes that confer advantages to individuals are selected over

time (because their carriers are more fit and survive longer) and non-advantageous genes are not selected as often. As a general rule, the preservation of “evolutionarily important amounts of quantitative genetic variation” requires effective population sizes of “at least several hundreds of individuals” (Lande and Barrowclough 1987:119; but see Lande 1995). Franklin (1980) recommended an effective population size of 500 for long-term conservation. Rudis and Tansley (1995) recommend a metapopulation of five interconnected subpopulations of black bears totaling 1,000-1,500 animals as the minimum conservation goal for Florida. Following the “precautionary principle” (Meffe and Carroll 1997:546), we recommend core refugia that are large enough to support an effective population of at least 500 black bears. This translates to an actual population of about 1000 bears, excluding cubs. Such populations would have a high probability of long-term viability and persistence.

Using density estimates of Costello et al. (2001), core areas of suitable black bear habitat should equal or exceed 6,000 km² (2,300 mi²) in northern mountainous regions within the NM Highlands planning area and 11,000 km² (4,250 mi²) in southern regions. Alternatively, subpopulations comprising a metapopulation of 1,000-1,500 bears (Rudis and Tansley 1995) should each contain at least 50 bears ($N_e \geq 25$) and be interconnected by functional linkages. Thus, northern subpopulations would require blocks of interconnected suitable habitat of at least 300 km² (116 mi²) each and southern subpopulations would need blocks of at least 500 km² (193 mi²). Per Costello et al. (2001), smaller subpopulations should not be discounted as lacking conservation value, especially if linked to larger subpopulations. The sum of these interconnected blocks should total 6,000-9,000 km² (2,300-3,500 mi²) in northern regions and 11,000-16,000 km² (4,250-6,200 mi²) in southern regions.

Population Status. According to Costello et al. (2001), the NM Highlands planning area contains about 50,000 km² (19,300 mi²) of potentially suitable black bear habitat and supports an estimated population of about 5,300 bears (excluding cubs). This constitutes about 90% of the estimated statewide black bear population.

MOVEMENTS

Black bears move for the purposes of finding food, dispersing as subadults, finding mates, and finding suitable dens (Pelton 2000). They often use long-established trails consisting of individual foot impressions in the substrate (LeCount 1986;

Reimchen 1998). These trails may represent “corridors of least resistance” (Reimchen 1998:698) through natural habitats. Reimchen (1998) observed that during daylight, bears consistently moved off trails into surrounding forested habitats when encountering humans or other bears, but exhibited much higher fidelity to trails and much greater tolerance of human presence at night. Trails may serve an ecological role of facilitating bi-directional movements by black bears through habitats during darkness (Reimchen 1998).

Dispersal. In the New Mexico study, none of 21 transmitter-collared females whose natal ranges were known dispersed. The tendency of females to establish home ranges in or near their natal home range (Pelton 2000) significantly affects the rate of re-establishment of extirpated subpopulations. If distances between core populations and isolated habitats are large, translocations (especially of females) may be necessary to achieve population re-establishment. In contrast, all male bears that were monitored until at least age 4 dispersed. Five male dispersal movements were documented. The age of dispersing males ranged from 1.5-3 years, and dispersal distances ranged from 25-60 km (15.5-37.3 mi). In their study of black bears in Alaska, Schwartz and Franzmann (1991) also observed that all surviving subadult males dispersed.

Migration. Bears exhibit little migratory behavior except in search of seasonally abundant foods (Schwartz and Franzmann 1991; Fitzgerald et al. 1994). In Colorado, bears moved from 13 to 36 km (8.1-22.4 mi) between summer and fall feeding areas (Beck 1991, as cited in Fitzgerald et al. 1994). Comparable migration distances (17-31 km or 10.6-19.3 mi) were reported by Schwartz and Franzmann (1991) in Alaska.

Barriers to Movement. Brody and Pelton (1989) found that bears avoided roads in areas open to hunting, but that in many protected areas bears were attracted to roads by the presence of human food. Thus, they postulate that the response of black bears to roads is primarily a learned behavior. In North Carolina, the frequency of road crossings by black bears was inversely related to traffic volume, with the greatest road avoidance occurring at an interstate (multi-lane) highway. Use of highway underpasses by black bears has been documented in Florida (Foster and Humphrey 1995).

Rudis and Tansley (1995) state that black bears avoid human contact. This suggests that black bears are not likely to move through areas occupied by humans unless the density of humans is low.

Use of Linkages. Rudis and Tansley (1995) describe short distance movement linkages as being characterized by dense vegetation and widths of 10-60 m (33-197 ft). They suggest that long-distance travel or dispersal may require large blocks of contiguous forested habitat. However, we found no documentation of dispersing bears using linkages to move between blocks of suitable habitat.

ECOLOGY

Interspecies Interactions. Interactions between black bears and other species are not well documented in the literature. There is little evidence to suggest that black bears limit populations of other species through competition or predation. In Alaska, where high levels of predation on moose calves by black bears were documented, habitat quality was determined to be the ultimate factor controlling moose densities and population trends (Schwartz and Franzmann 1991).

Ecological Effects. Black bears function ecologically as omnivores, but mostly consume herbaceous material, fruits, seeds, and nuts (Pelton 2000). Energy obtained by bears from consumed plant material and some animals (alive and dead) is passed on to predators and scavengers completing the ecological cycling of energy and nutrients (Odum 1993). Seeds consumed by bears are dispersed when bears defecate, often considerable distances from where they were eaten. In their occasional role as carnivores, black bears may belong to a suite of top predators (potentially including mountain lions, gray wolves, and grizzly bears) that collectively regulate other animal and plant populations and ecological processes (Gassaway et al. 1992; Terborgh et al. 1999).

JUSTIFICATION AND FOCAL VALUE

Habitat Quality Indicator. Black bears occur in low densities, require large contiguous or connected areas for population persistence, avoid roads and open areas, have a relatively low reproductive potential, and subadult females rarely disperse from their natal areas. These qualities make black bears particularly sensitive to habitat fragmentation, which is a threat to habitat quality throughout the NM Highlands planning area. Black bears rely upon a variety of seasonally-available foods, which contribute to the quality of habitats for a variety of other species in the ecosystem.

Umbrella. Black bears have large area requirements for population persistence. They require a variety of forest, woodland, and shrubland habitats for feeding, hiding, and denning. Protection of habitat for viable populations of black bears will, by inclusion, protect the habitats of many other species.

MANAGEMENT RECOMMENDATIONS

Establish Refugia. Black bears suffer from high levels of human-caused mortality in areas with roads or access by off-road vehicles. Legal protection does not eliminate poaching. Persistent, viable populations are essential to the long-term survival and conservation of this species and would serve as important sources of dispersing bears (primarily males) for rescuing declining populations in surrounding areas. Protected areas in the range of 6,000 km² (2,300 mi²) in the northern areas and 11,000 km² (4,250 mi²) in the southern areas, or larger, characterized by preferred black bear habitat should be considered for refuge designation. Designated refuges should be roadless, or have low densities of roads with an access management plan that protects bears during critical periods. Hunting should be prohibited or at least severely restricted in refuge areas. Natural disturbance regimes should be allowed to operate within refugia, and prescribed fires may be a useful tool for improving black bear habitat.

Protect Subpopulations. Management practices should consider the vulnerability of local, isolated black bear populations to extinction. Management considerations should include habitat protection and enhancement, prohibition of take, and population augmentation through translocation of bears (especially females) from source populations. Areas of suitable black bear habitat exceeding 300 km² (116 mi²) in northern areas and 500 km² (193 mi²) in southern areas should be protected.

Protect and Restore Linkages. Thick, continuous cover should be retained and encouraged through management practices along ridge tops, gaps, ravines, and riparian corridors and around water sources. This will provide safe access for bears to seasonal food and water resources and for dispersing bears to patches of suitable unoccupied habitat. Linkages should include underpasses integrated with roadside fencing where high-speed roads cross linkages.

Road Closures. Roads are the most significant contributing factor to black bear mortality, because they provide human access to occupied habitats and humans cause most bear mortality. Some nonessential roads (e.g., off-road vehicle trails and dirt roads) should be closed in critical core areas and linkages between them where existing road densities exceed suitability thresholds (0.8 km/km² or 1.3 mi/mi²) or illegal killing is documented. We do not advocate the closing of important thoroughfares, paved roads, or highways.

Retain Large Trees, Snags, and Fallen Logs. Large trees, snags, and logs provide important denning sites, especially for female black bears. Thus, retention and perpetuation of old-growth forests through forest planning will likely enhance survival and reproductive success. In addition, some slash from logged areas should be piled in or near the edge of standing timber and left to provide additional den sites for bears.

Conduct Annual Mast Surveys. Production of acorns and, secondarily, juniper berries is strongly correlated to black bear reproductive success in New Mexico. Thus, reproductive declines or failures can be reasonably predicted. Hunt quotas should be adjusted to avoid over-hunting following reproductive failures. The lag effect of missing age cohorts on future reproductive potential of black bear populations should be considered in hunt quota adjustments.

Monitoring. Black bear populations are particularly sensitive to over-hunting, especially of females. Monitoring is important for the establishment of sustainable kill limits. The accuracy of various population estimating methods and indices varies considerably, potentially causing erroneous interpretation and ill-advised management decisions (Kane and Litvaitis 1992; Noyce and Garshelis 1997). Managers should include multiple estimators and indices in ongoing black bear monitoring programs (Kane and Litvaitis 1992; Pelton 2000).

Multi-jurisdictional Planning. Conservation planning for large carnivores must be conducted over vast spatial scales and must consider connectivity among local subpopulations (Noss et al. 1996). Land areas large enough to support a viable metapopulation of black bears will likely encompass a multitude of jurisdictions. Establishment of regional planning authorities through appropriate means (e.g., legislative or administrative) should be encouraged and pursued.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of black bear ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about black bears to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the pub

American Marten (*Martes americana*)

*Please see accompanying Excel chart of American Marten Habitat Characteristics in the folder titled “*Focal Species Habitat Characteristics Charts*” on the compact disk.

INTRODUCTION

The American marten (*Martes americana*) is a member of the weasel family (Mustelidae) and is slightly smaller than an average house cat. These small mammalian predators are sparsely distributed within the NM Highlands planning area. The American marten is a habitat-specialist that lives in old-growth spruce-fir forests and preys primarily on small mammals. It requires habitat protection and forested linkages among patches of suitable habitat for its survival and persistence. Ecologically, the marten is an important predator of small mammals and may be an important distributor of seeds of fruit-bearing shrubs. It is an indicator of habitat and wilderness qualities within the NM Highlands planning area.

DISTRIBUTION

Historic. American martens occupied temperate and boreal forests of the Rocky Mountains from northern New Mexico to Alaska; the Sierra Nevada, Cascade and Coastal ranges of California, Oregon, and Washington; most of Canada; and the Great Lakes and Northeast regions of the United States (Hall 1981). Their distribution is discontinuous in the southern Rocky Mountains because climate change isolated suitable habitats on the tops of mountains following the Pleistocene epoch (Gibilisco 1994). The southern limit of marten distribution in the southern Rocky Mountains roughly coincides with the southern limit of Engelmann spruce (*Picea engelmannii*) (Buskirk and Ruggiero 1994). The resulting patchy distribution is common at the edge of a species' distribution (Gibilisco 1994). Findley et al. (1975) depict historic range of martens in New Mexico as far south as the Mogollon Mountains, Black Range, Sacramento Mountains, and Guadalupe Mountains but provide no records of specimens south of Santa Fe. Bailey (1931) described marten distribution as uncommon in the Sangre de Cristo and San Juan Mountains, which he believed marked the southern limit of their range. He attributed their scarceness to trapping pressure for their valuable pelts.

Population declines resulted from exploitation of martens for their valuable fur and habitat alterations caused by logging (Strickland 1994). Martens were extirpated from much of the southern parts of their historic range (Strickland 1994), but

have become reestablished (through recolonization or reintroduction) in many states and provinces (Gibilisco 1994).

Current. Marten populations at the edge of their distributional range are naturally unstable and are especially vulnerable to local extirpation as a result of overharvest, habitat alterations, and fragmentation of forested environments. This appears to be the situation in the southern Rocky Mountains. The presence of American martens in New Mexico has been documented generally north of Santa Fe in the San Juan and Sangre de Cristo Mountains. Reports of martens in the Jemez Mountains have not been verified (B. Long, pers. comm.).

Potential. Areas of approximately 40 km² (15.4 mi²) or greater within high-elevation mesic coniferous forest types with the following characteristics generally represent potentially occupied range for American martens: 1) >20% of forest stand in old-growth age classes; 2) >30% canopy closure; 3) small openings with high understory plant diversity; 4) complex vertical and horizontal structure with an abundance of snags and large woody debris that breeches the snow surface in winter; 5) forested connectivity within and between patches of suitable habitat; and 6) an abundance of small mammalian prey.

HABITAT

General. The following description of habitat requirements and preferences of American martens is specific to the intermountain West. Habitat relationships are somewhat different for midwestern and eastern populations. American martens are “habitat-specialists” that occupy structurally complex, late successional forests dominated by conifers (Koehler and Hornocker 1977; Buskirk and Powell 1994). Martens will use patches of preferred habitat that are interconnected by other forest types, but will avoid similar patches that are separated by large open areas (Buskirk and Powell 1994). This explains why marten populations no longer exist in many small mountain ranges in post-glacial time and why recolonization of potentially suitable, but isolated, habitats has not occurred (Buskirk and Powell 1994; Gibilisco 1994). Open areas with abundant cover (e.g. shrubs or fallen trees) are used by martens for foraging (especially in summer), but open grasslands and alpine tundra are avoided (Koehler and Hornocker 1977; Buskirk and Powell 1994).

Preferred. American martens prefer old-growth mesic coniferous forest types with complex physical structure, especially near the ground (Buskirk and Ruggiero 1994; Thompson and Curran 1995). Mesic sites support greater understory plant

species diversity and higher vole populations (Koehler and Hornocker 1977). Small openings where large trees have fallen enhance understory plant and animal diversity (Thompson and Harestad 1994). Abundance of small mammals, especially voles (*Clethrionomys* spp., *Microtus* spp., and *Peromyscus* spp.) and pine squirrels (*Tamiasciurus* spp.), is directly associated with abundance of American martens (Strickland et al. 1982; Buskirk and Ruggiero 1994; Martin 1994). Optimum American marten habitat should include 20-25% of the forest at the landscape level in mature age classes at any given time; at least 50% conifers; large standing snags; and numerous fallen trees for subnivean (under the snow) foraging and rest sites (Soutiere 1979; Steventon and Major 1982; Buskirk et al. 1989; Berg and Kuehn 1994; Thompson and Harestad 1994). An abundance of large, near-ground woody structures provides martens with protection from predators, access to subnivean space where most prey are captured in winter, and protective thermal microenvironments (Buskirk and Ruggiero 1994).

In the Rocky Mountains, American martens prefer mesic high-elevation stands dominated by spruce (*Picea* spp.) and fir (*Abies* spp.) over stands dominated by dry-site species such as ponderosa pine (*Pinus ponderosa*) and lodgepole pine (*Pinus contorta*) (Buskirk and Powell 1994). In more southern reaches of their range, martens may select riparian forests for resting during winter (Buskirk et al. 1989). Mature spruce-fir forests are preferred habitat for important marten prey species such as southern red-backed voles (*Clethrionomys gapperi*), which depend on large downed logs and stumps (coarse woody debris) and dense forest canopy, and red squirrels (*Tamiasciurus hudsonicus*) (Martin 1994).

Martens avoid areas with less than 30% canopy closure and open areas with no overstory or shrub cover (Buskirk and Powell 1994). Habitat that has become "highly fragmented" (e.g., habitat patches <200 km² [77 mi²] separated by >5 km [3 mi] zones with <50% overhead cover) is not likely to be used by martens (Buskirk and Powell 1994). Robitaille and Aubry (2000) observed significantly fewer marten tracks within 400 m (0.25 mi) of roads compared to transects 800-1000 m (0.5-0.62 mi) from roads. This suggests a zone of reduced habitat suitability for martens near roads.

In areas where deep snow accumulates, American martens prefer cover types with sufficient vertical and horizontal structure (e.g., closed, multi-layered coniferous forests) to prevent snow from packing hard and with near-ground structures that breach the snow surface and provide access to subnivean spaces (Buskirk and Powell 1994).

Seasonal Habitat Preferences. Generally, martens use a wider range of cover types within their established home ranges (including small open areas) in summer but strongly prefer old conifer-dominated stands in winter (Buskirk and Powell 1994; Buskirk and Ruggiero 1994). Habitat suitability for martens appears to be limited by the availability of preferred winter habitat within their home ranges (Buskirk and Ruggiero 1994).

Resting Sites. American martens use resting sites for thermoregulation (i.e., to conserve energy by reducing heat loss in winter), resting in all seasons, and protection from predators (Buskirk and Ruggiero 1994). Generally, martens rest above ground in the warmer months and below the snow when temperatures fall below freezing (Buskirk et al. 1989). Martens remain in resting sites for several hours and often all day (Buskirk 1984). Energy conservation achieved through rest site selection reduces food consumption needs by 1-3 voles per day and corresponding foraging effort for martens (Taylor and Buskirk 1994).

Favored resting sites include logs, stumps, snags, pine squirrel middens and tree nests, tree cavities, tree limbs, dense clumps of tree limbs ("witches' brooms") caused by tree parasites (e.g. broom rust and dwarf mistletoe), ground burrows, rock fields, root wads, lumber piles, and buildings (Buskirk 1984; Spencer 1987; Buskirk et al. 1989; Buskirk and Powell 1994; Bull and Heater 2000). Resting sites were located under or adjacent to, as well as within cavities inside, logs, stumps, and the base of snags (Buskirk et al. 1989; Bull and Heater 2000). Most subnivean resting sites contained evidence of use by pine squirrels (Spencer 1987; Bull and Heater 2000). Large trees, logs, and snags are preferred over smaller ones (Spencer 1987).

Preferred resting sites are generally found within old-growth spruce-fir forests with relatively closed canopies and an abundance of standing snags in early stages of decay and coarse woody debris on the forest floor (Buskirk 1984; Buskirk et al. 1989; Buskirk and Powell 1994; Bull and Heater 2000). Coarse woody debris in addition to large trees with low limbs and snags provide important access points to subnivean resting and foraging sites (Buskirk and Ruggiero 1994). The distribution and abundance of preferred rest sites contribute to habitat quality and may limit the distribution and abundance of martens (Buskirk 1984; Spencer 1987; Taylor and Buskirk 1994).

Den Sites. American martens use two types of dens: natal (where young are born) and maternal (where young are reared). In order of observed importance, martens den in hollow logs, live tree cavities, rocks, snag cavities, stumps, slash piles,

underground burrows, pine squirrel middens, man-made structures, and root wads (Buskirk and Ruggiero 1994; Bull and Heater 2000). Den trees, snags, and logs are characterized by large diameters, emphasizing the importance of old-growth forests for martens (Buskirk and Ruggiero 1994; Bull and Heater 2000). The availability of structurally complex old-growth coniferous forests, which provide suitable natal dens, could have important implications for the conservation of American martens (Buskirk and Ruggiero 1994).

FOOD HABITS / HUNTING BEHAVIOR

American martens are opportunistic feeders and adjust their dietary preferences seasonally to take advantage of prey availability (Martin 1994; Verts and Carraway 1998; Bull 2000). They eat small mammals, birds, insects, and fruits (Strickland et al. 1982; Martin 1994). Voles are prominent in their diet (Koehler and Hornocker 1977; Soutiere 1979; Strickland et al. 1982; Gordon 1986; Fitzgerald et al. 1994; Martin 1994; Thompson and Curran 1995; Simon et al. 1999; Bull 2000). In some areas, pine and ground (*Spermophilus* spp.) squirrels may be important prey in late winter and early spring (Buskirk and Ruggiero 1994). Birds, eggs, insects, and fruits are important foods in summer and fall when they are available and vulnerable (Soutiere 1979; Strickland et al. 1982). Vegetative, bird, and insect components of martens' overall diets are likely secondary in importance to mammalian prey (Martin 1994). Martens readily consume carrion, especially in winter (Strickland et al. 1982; Fitzgerald et al. 1994).

Martens pursue their prey on the ground and in trees (Fitzgerald et al. 1994). Activity periods vary seasonally and martens are mostly crepuscular and nocturnal in winter with increased activity during daylight hours in the summer (Fitzgerald et al. 1994; Verts and Carraway 1998). In winter, much hunting occurs beneath the snow (Martin 1994). Fallen logs, trees with low limbs, and tree stumps that break the snow surface provide access to this subnivean zone (Strickland et al. 1982) and are routinely investigated by hunting martens, presumably for olfactory cues of prey presence (Buskirk and Powell 1994). Sherburne and Bissonette (1994) found that American martens used subnivean areas with higher prey biomass and higher amounts of coarse woody debris. They speculated that martens are able to discriminate between subnivean access points with high prey levels and those with low prey levels. Food may be cached at resting sites (Fitzgerald et al. 1994). Typically, older forests with accumulated coarse woody debris provide the forest floor structure necessary to enable martens to

forage effectively in the winter (Buskirk and Ruggiero 1994). Exceptions may include mid-aged stands where burns or wind-throw provide coarse woody debris on the forest floor (S. Buskirk, personal communication).

POPULATION DYNAMICS

Life History. American martens are members of the weasel family (Mustelidae). They are slightly smaller than an average house cat with the following body dimensions: total length 460-750 mm (1.5-2.5 ft); length of tail 170-250 mm (0.6-0.8 ft); and weight 0.5-1.2 kg (1.1-2.6 lbs) (Fitzgerald et al. 1994). Males are 20-40% larger than females (Buskirk and Ruggiero 1994). The following life history information is summarized from Verts and Carraway (1998) and Strickland et al. (1982). Martens are solitary, but males and females may pair briefly for mating; home ranges of males overlap little and home ranges of one or more females may occur within the home range of a male. Most female martens become sexually mature at about 15 months and produce their first litters at the age of 2 years. Males and females may mate with more than one partner and breeding occurs in the summer months. Total gestation ranges from 220-276 days; only one litter is produced per year; young are born in March and April; and average litters have about 3 young. Wild reproduction by 12 to 14-year-old females has been documented, although few martens (<15%) live beyond 4 years (Clark et al. 1989; Nowak 1991; Fitzgerald et al. 1994).

Population Density. American martens occur at very low densities compared to other mammalian carnivores of similar size (Buskirk and Ruggiero 1994). While marten populations are constantly in flux due to changes in prey availability, average population density in suitable habitat is about 1.2 adults/km² (3.1/mi²) (Strickland et al. 1982). In fall and winter, transient adults and juveniles may increase density to over 2 martens/km² (3.1/mi²).

Home Range. Mean home range size for males is 8.1 km² (3.1 mi²) and for females is 2.3 km² (0.9 mi²) (Powell 1994). Home range size is inversely related to prey availability and habitat quality and directly related to body size of the marten (Thompson and Colgan 1987; Buskirk and Ruggiero 1994).

Causes of Death. About half of martens die in their first year of life; 15-20% die in their second year; 12-13% in their third year; and few martens (<15%) live beyond 4 years (Strickland et al. 1982; Bull and Heater 2001).

The average annual harvest of American martens in North America in the 1980s was 192,000 animals (Strickland 1994). Reduced popularity of furs in the 1990s has led to reduced trapping pressure. In harvested populations, trapping is the predominant cause of death for martens (Buskirk and Ruggiero 1994). Trapping mortality may be additive to natural mortality (Strickland 1994). Bull and Heater (2001) found predation to be the major cause of mortality of an untrapped marten population in Oregon.

Predation on martens by coyotes (*Canis latrans*), bobcats (*Felis rufus*), red foxes (*Vulpes fulva*), other martens, great-horned owls (*Bubo virginianus*), fishers (*Martes pennanti*), lynx (*Lynx canadensis*), mountain lions (*Puma concolor*), eagles (*Aquila chrysaetus*, *Haliaeetus leucocephalus*), and northern goshawks (*Accipiter gentilis*) has been documented (Strickland et al. 1982; Buskirk and Ruggiero 1994; Slough 1994; Squires 2000; Bull and Heater 1995, 2001). Bull and Heater (2001) observed that most predation occurred between May and August and that no predation occurred between December and February, when martens spend more time in dense cover and under the snow.

Diseases and parasites are not known to limit American marten populations (Strickland et al. 1982; Slough 1994), with the exception of canine distemper which was determined to be the cause of a dramatic die-off of martens on Newfoundland Island (Fredrickson 1990, cited in Buskirk and Ruggiero 1994).

In roaded areas, vehicle collisions cause some marten mortality (Buskirk and Ruggiero 1994). Bull and Heater (2001) documented hypothermia as a cause of mortality in martens.

Population Structure and Viability. A metapopulation structure has not been described for marten populations; but, intuitively, this type of dispersed population structure seems plausible for the naturally and anthropogenically fragmented habitats that exist in the southern portions of its range (Buskirk and Ruggiero 1994), including the NM Highlands planning area. Generally, marten population structure is characterized by non-overlapping territories of same-sex adults, larger male territories overlapping one or more female territories, predispersal young occupying adult female territories, and dispersing juveniles often occupying marginal habitats (Buskirk and Ruggiero 1994). Martens are believed to be territorial; territory size and local population density are linked to food density; and intraspecific aggression has the potential to increase marten mortality rates (Fryxell et al. 1999; Bull and Heater 2001).

American marten populations fluctuate dramatically (up to an order of magnitude in less than a decade) in direct response to

fluctuations in prey populations (Buskirk and Ruggiero 1994; Martin 1994; Powell 1994). American martens are easily trapped, have high pelt values, have relatively low recruitment rates, and are vulnerable to over-trapping (Strickland 1994). Because marten reproductive rates are low, they are slow to recover from population declines (Buskirk and Ruggiero 1994).

Slough (1994) proposed that populations in excess of 50 animals were necessary for short-term viability and that over 500 animals were required to assure long-term fitness and genetic adaptability of a marten population. Schneider and Yodzis (1994) predicted that populations with 75-125 females were likely to persist for 500 years. Assuming an even sex ratio for untrapped populations (Powell 1994), this would equate to an adult population of 150-250 martens, which accords with the estimate of a minimum of 237 martens for population viability by Thompson and Harestad (1994). Lacy and Clark (1993) found that all but one simulated populations of 50 and 100 martens had a 100% probability of extinction within 100 years when no immigration into the population was assumed. Simulated immigration of one pair of martens per year was sufficient to prevent extinction, even with specified levels of trapping and logging. Smaller populations benefit from periodic infusions of immigrants from larger, more genetically diverse populations.

Assuming a population density of 1.2 individuals/km² (3.1/mi²), core populations in reserves of 200-400 km² (77-154 mi²) would be necessary to support nearby subpopulations in patches of suitable habitat ranging from 40-200 km² (15-77 mi²). Multiple populations of minimum viable size or larger linked by suitable migration corridors further ensure long-term survival of American martens in a region (Thompson and Harestad 1994). Small populations are at increased risk of extinction from natural catastrophes (e.g., severe storms and fires) and other random events that effect the survival and reproduction of individuals (e.g., an unusually skewed sex ratio); environmental conditions or ecological relationships (e.g., reduced food supply or increased population levels of competitors or predators); and genetic health of the population (e.g., inbreeding and genetic drift) (Meffe and Carroll 1997). Migrants from surviving populations can “rescue” declining populations or recolonize suitable habitats following local extirpations.

American marten populations decline following clear-cut logging at all but very small (0.5-3 ha [1.2-7.4 acres]) scales (Soutiere 1979; Buskirk and Ruggiero 1994; Thompson and Colgan 1994; Thompson and Harestad 1994; Forsey and Baggs 2001). Thompson and Harestad (1994) compared 10 studies of

habitat selection by martens. Use/availability ratios were consistently less than 1 for shrub, sapling, and pole stages of forest regeneration. Only old-growth stands consistently had use/availability ratios that exceeded 1, indicating a preference or selection for these habitats by martens. Clear-cutting removes overhead cover, removes large-diameter coarse woody debris, and tends to convert mesic sites to xeric sites with associated changes in prey communities to less preferred species (Buskirk and Ruggiero 1994).

Population Status. While the presence of martens has been documented in the NM Highlands planning area, little is known about the status, trends, or potential long-term viability of extant populations.

MOVEMENTS

Dispersal. Data on the dispersal and migrational movements of martens are scarce (Buskirk and Ruggiero 1994). American martens will travel through forested areas that are otherwise not preferred habitat (Buskirk and Powell 1994). Factors important to successful marten dispersal and population persistence of isolated populations include size of habitat islands, distance between habitat islands, distance to large core populations, and the nature of the zones that separate populations (Gibilisco 1994). Martens are known to traverse alpine tundra (1.1 km [0.7 mi]), forests with <25% canopy cover (>10 km [6.2 mi]), extensive burned forest (>20 km [12.4 mi]), and large rivers (>100 m [328 ft]) (Slough 1989). Reported dispersal distances for juvenile martens range from 27-60+ km (17-37+ mi) (Weckwerth and Hawley 1962; Strickland and Douglas 1987). No significant between-sex differences in dispersal distances have been reported.

Barriers to Movement. American martens tend to avoid open areas, but may travel up to 3 km (1.9 mi) from forested cover provided some physical cover (e.g., talus fields) is available (Streeter and Braun 1968; Buskirk and Powell 1994; Potvin et al. 1999). Distances of more than 5 km (3.1 mi) of unforested land below the conifer zone are believed to present a complete barrier to dispersal (Gibilisco 1994). Koehler and Hornocker (1977) observed that martens passed through, but did not hunt in, openings <100 m (<109 yards) wide, and Soutiere (1979) documented martens crossing 200-m-wide (219 yards) openings. Martens avoided bait placed >23 m (25 yards) from the forest edge within openings in winter (Robinson 1953). Bull and Blumton (1999) observed that transmitter-collared martens in northeastern Oregon avoided all harvested stands and stands with less than 50% canopy cover.

Use of Linkages. Use of habitat linkages by dispersing martens has not been documented. Gibilisco (1994) suggested that forested riparian zones may serve as dispersal corridors linking disjunct populations of martens. Martens' observed reluctance to cross areas lacking overhead cover suggests the importance of linkages with overhead cover to the colonization of patches of suitable habitat devoid of martens, the genetic and/or demographic augmentation of small subpopulations, and the persistence and viability of marten populations (Buskirk and Ruggiero 1994).

ECOLOGY

Interspecies Interactions. American marten populations tend to respond directly to changes in prey populations (Strickland et al. 1982; Thompson and Colgan 1987; Buskirk and Ruggiero 1994; Fryxell et al. 1999; Simon et al. 1999). Martens kill many small mammals but are only one of many predators of small mammals in the forest communities they occupy (Schneider and Yodzis 1994). There is no evidence to date that suggests that martens exert a regulatory effect on prey populations, except perhaps as a member of the larger suite of predators of small mammals.

Ecological Effects. Because American martens forage heavily on fleshy fruits in the summer and fall, they may be important dispersers of the seeds of fruit-bearing shrubs, such as blueberries (*Vaccinium* spp.) and salmonberries, raspberries, blackberries, and thimbleberries (*Rubus* spp.) (Buskirk and Ruggiero 1994; Hickey et al. 1999). Other ecological effects of martens have not been elucidated.

JUSTIFICATION AND FOCAL VALUE

Habitat Quality Indicator. Decline of American marten populations because of habitat loss is indicative of more insidious forest management problems—specifically, the inability to regenerate forest ecosystems to prior levels of complexity and the general lack of long-term, broad-scale vision in forest planning (Thompson and Harestad 1994).

Wilderness Quality Indicator. The American marten is dependent on the presence of old-growth, unlogged forests for habitat. The best protection for such forests is formal designation as Wilderness Areas.

MANAGEMENT RECOMMENDATIONS

Establish Refugia. Many relict populations of American martens owe their existence to refuges where furbearers were protected or inaccessible. Refuges provide population reservoirs for dispersal to surrounding areas (Strickland 1994). However, much of the knowledge necessary to inform refuge design, such as the required sizes and distances separating refugia, is lacking (Buskirk and Ruggiero 1994). Based upon published population viability assessments, population density estimates, and dispersal distances, we recommend the establishment of refugia in all patches of suitable habitat of 40 km² (15 mi²) or larger. Refugia in the size range of 200-400 km² (77-154 mi²) or larger are preferred. Refugia smaller than 200 km² (77 mi²) must be connected by forested cover with >50% canopy closure and no farther than 50 km (31 mi) from two or more other refugia and, preferably, one in excess of 200 km² (77 mi²). In the absence of suitable linkage between refugia smaller than 200 km² (77 mi²), supplementation of small populations with at least one pair of adults per year is recommended.

Protect Subpopulations. Schneider and Yodzis (1994) suggest that a system of multiple linked marten subpopulations (supported through landscape-level habitat management) will be necessary to ensure the long-term persistence of martens in a given region. Management practices should consider the vulnerability of local, isolated marten subpopulations to extinction (Buskirk and Ruggiero 1994; Gibilisco 1994; Schneider and Yodzis 1994). Management considerations should include habitat protection and enhancement, dispersal linkages among subpopulations, prohibition of take, and subpopulation augmentation through translocation of martens from larger populations. Populations that have persisted since prehistoric times likely represent locally adapted forms and warrant greater protection than populations created through translocations (Buskirk and Ruggiero 1994).

Protect and Restore Linkages. Because martens will not travel far from overhead forest cover, direct links of forest cover with greater than 50% canopy cover among suitable habitat blocks are essential and recommended (Witmer et al. 1998).

Reintroduce Martens to Suitable Habitats. Reintroductions have successfully re-established American marten populations in many areas (Slough 1994). All translocation attempts where 30 or more martens were released were successful. Quality of habitat is important to the success of reintroductions (Slough 1994). Reintroduction should be considered for areas of suitable habitat of sufficient size (>40 km² [15 mi²]) to support a subpopulation of martens (Buskirk and Ruggiero 1994).

Monitoring. Because fitness is difficult to assess in marten populations, population density is probably the most useful and attainable measure of population fitness and habitat quality (Buskirk and Ruggiero 1994). Martens are sensitive to habitat loss or degradation. Resource managers must implement effective monitoring strategies to detect effects of land management practices on habitat quality and numerical abundance and density of martens (Buskirk and Ruggiero; Raphael 1994). Promising techniques for monitoring marten populations are snow tracking, sooted track plates, and baited remote camera stations (Bull et al. 1992; Buskirk and Ruggiero 1994; Raphael 1994; Zielinski and Kucera 1995).

Forest Planning. The Wildlands Project prefers that suitable habitat conditions for American martens and other focal species be achieved through the restoration of natural process, especially in core refugia. In compatible use areas where logging is an accepted land use and marten conservation is a conservation objective, forest plans should provide sufficient old-growth coniferous forest to ensure the long-term survival and viability of American martens. Thompson and Harestad (1994) recommend the following timber harvest strategy for mature conifer-dominated forests, which they predict will increase the forest's carrying capacity for American martens: dispersed timber removal in 1-3 ha cuts of less than 25% of total stems with no more than 20-30% total forest removal. These criteria should be applied over the maturation cycle for old-growth spruce-fir forests at the landscape scale. The proportion of old-growth spruce-fir forest should exceed 20% at the landscape scale at any given time. Forest management should promote coarse woody debris and snags larger than 80 cm in diameter in forest communities identified as winter resting habitat for martens (Buskirk et al. 1989; Witmer et al. 1998). The most important consideration for forest planners is the dynamic extent and configuration of the remaining forest following the application of a timber harvest prescription (Potvin et al. 1999). Of course, forest planning should rarely be single-species oriented and should follow an approach that will benefit entire ecosystems and their full suite of occupants. The adverse effects of roads and logging practices should be fully considered and minimized. The Wildlands Project opposes the construction of new roads in existing roadless areas and advocates the closure of many existing but unnecessary roads in publicly-owned forest lands.

Fire. Fire is an important agent in creating forest diversity. "A mosaic of forest communities supporting discontinuous fuel types can...be expected to result in smaller and generally cooler fires, which would result in less marten habitat being replaced though time and space." Koehler and Hornocker (1977:504). High-intensity fires that consume coarse woody

debris and large snags are not beneficial to martens (Strickland et al. 1982). Martens tend to avoid large openings created by fire.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of marten ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about martens to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

REVIEWERS

Steven Buskirk, University of Wyoming
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Bighorn Sheep (*Ovis canadensis* ssp.)

*Please see accompanying Excel chart of Bighorn Sheep Habitat Characteristics in the folder titled “*Focal Species Habitat Characteristics Charts*” on the compact disk.

INTRODUCTION

The bighorn sheep (*Ovis canadensis*) is one of only two native members of the mammalian family Bovidae in New Mexico (the other being the bison). Two subspecies of bighorn sheep—Rocky Mountain (*O. c. canadensis*) and desert (*O. c. mexicana*)—are sparsely distributed throughout the NM Highlands planning area in steep, rugged mountainous terrain. Ecologically, bighorns are herbivores that occupy a unique grazing niche on steep slopes and are occasional prey for large predators, especially mountain lions. Bighorns require interconnected clusters of suitable habitat for population viability and migrate between seasonal and special-use ranges. The Rocky Mountain bighorn is protected by the New Mexico Department of Game and Fish (NMDGF) as a game species, and the desert bighorn is protected as a state-level endangered species. The status of bighorn populations in the NM Highlands planning area is tenuous. The bighorn’s popularity with both trophy hunters and non-consumptive nature enthusiasts fosters considerable public interest and support for nature conservation.

DISTRIBUTION

Historic. Bighorn sheep probably evolved from Asian ancestors that migrated to North America across the Bering Land Bridge during the Pleistocene epoch (Krausman and Shackleton 2000). Historically, bighorn sheep were distributed throughout the mountainous regions of western North America and adjacent river valleys and prairies from southwestern Canada to northwestern Mexico, including most mountain ranges within the NM Highlands planning area (Findley et al. 1975; Hall 1981; Shackleton et al. 1999). Once relatively abundant, bighorn sheep are now one of the rarest ungulates in North America (Valdez and Krausman 1999).

Two recognized subspecies occurred in New Mexico: Rocky Mountain bighorns (*O. c. canadensis*) in the San Juan, Jemez, and Sangre de Cristo mountains in the north, and desert bighorns (*O. c. mexicana*) elsewhere in the state (Findley et al. 1975; Hall 1981). Unregulated harvesting caused the extirpation of Rocky Mountain bighorn sheep by 1903 (Frey and Yates 1996). Native populations of desert bighorns survived until recently only in the San Andres Mountains of south-central New Mexico

(NMDGF 2001a). Several populations of both subspecies have been reestablished through reintroduction efforts. Some restoration efforts have established populations of Rocky Mountain bighorns within historic desert bighorn range.

Current. As of 2002, populations of Rocky Mountain bighorn sheep occurred in the following locations within the NM Highlands planning area: Latir Wilderness, Wheeler Peak, Pecos Wilderness, Manzano Mountains, Turkey Creek (Gila National Forest), and San Francisco River (Gila National Forest) (NMDGF 2001b; E. Goldstein, personal communication). Populations of desert bighorn sheep occur in the Sierra Ladrones, Fra Cristobal Mountains, and San Andres Mountains (NMDGF 2001a; E. Goldstein, personal communication). Three additional populations of desert bighorns exist south of the NM Highlands planning area in the New Mexico “bootheel” (NMDGF 2001a).

Potential. If reasons for past extirpations can be fully understood and addressed, potential exists within the NM Highlands planning area to restore and expand populations of native bighorn sheep.

HABITAT

General. Typical habitat for bighorn sheep occurs in river canyons and benches, foothills, and mountains on or near rugged terrain with steep slopes. Bighorn sheep seldom venture far from steep, rugged escape terrain, which provides protection from terrestrial predators. In addition to rugged physical terrain, bighorns require high-visibility habitats dominated by grasses and low shrubs (Fitzgerald et al. 1994; Krausman et al. 1999; Shackleton et al. 1999). Fire suppression has resulted in degradation of some bighorn sheep habitat by the encroachment of dense, tall shrubs and conifers (Singer et al. 2000b).

Preferred. Essential habitat components for bighorn sheep are food, water, open space, and escape terrain (Krausman et al. 1999). Food preferences are discussed in the following section. Smith et al. (1991) developed habitat suitability criteria for Rocky Mountain bighorn sheep which were further tested and refined by Johnson and Swift (2000) and Zeigenfuss et al. (2000). Unless otherwise attributed, the following habitat preferences are from these sources.

Bighorn sheep require escape terrain to avoid predation. Optimal escape terrain is defined as having slopes between 27-85° with occasional rock outcrops. Bighorns use zones within

300 m (330 yards) of escape terrain for foraging. Areas 1000 m (0.6 mi) wide are considered available for foraging when escape terrain is present on two or more sides. Similarly, for desert bighorn sheep Dunn (1994) recommended slopes of 60% or greater. Ewes prefer steeper, more rugged slopes of 2 ha (5 acres) or more in total area for lambing and rearing of young lambs during their first week of life (Smith et al. 1991). Zeigenfuss et al. (2000) found that the proportion of total suitable habitat that is suitable for lambing was one of the best predictors of population size and success. Suitable lambing habitat for Rocky Mountain bighorns may require water within 1000 m (0.6 mi), and southerly aspects are preferred (Smith et al. 1991; Zeigenfuss et al. 2000).

Bighorn sheep can meet a considerable amount of their water requirement from metabolic processes (oxidation) or from direct consumption of succulent vegetation, snow, or ice (Krausman et al. 1999; Krausman and Shackleton 2000). However, during hot, dry seasons when water loss due to thermoregulation is high and vegetative moisture content is low, nearby (< 3.2 km [2 mi]) water is considered essential for bighorns (Smith et al. 1991; Dunn 1994; Krausman et al. 1999; Krausman and Shackleton 2000).

Bighorn sheep tend to be intolerant of humans and their activities and have abandoned home ranges following increases in human activity (Krausman et al. 1999). Areas within 150 m (164 yards) of high-use areas are considered unsuitable habitat (Smith et al. 1991). King and Workman (1986) found that the flight response and time spent in behaviors associated with wariness of desert bighorn sheep in southeastern Utah increased with increasing levels of human disturbance. They speculated that increased energy expenditures and reduced foraging efficiency caused by human disturbance may predispose bighorns to reduced fitness and reproductive success and increased mortality, especially during critical time periods, such as lambing. However, bighorn sheep apparently can and do habituate to many predictable, nonthreatening human activities, such as rock quarries, busy highways, and residential areas (Fitzgerald et al. 1994; Johnson and Swift 2000).

Areas with high densities of domestic and exotic sheep and goats, cattle, feral burros, or elk are considered unsuitable for bighorn sheep because of food competition, disease transmission, and possibly behavioral avoidance of cattle by bighorns (Smith et al. 1991; Krausman et al. 1999; Krausman and Shackleton 2000). Domestic sheep and goats, and exotic relatives of bighorn sheep, such as mouflon sheep (*Ovis ammon musimon*), barbary sheep (*Ammotragus lervia*), and ibex (*Capra ibex*) carry diseases that are often lethal to bighorns, compete for lim-

ited food resources, and may cause genetic pollution through interbreeding with bighorns (Smith et al. 1991). Areas less than 20 km (12.5 mi) from populations of domestic or exotic sheep or goats are unsuitable because of transmissible diseases for which native bighorns lack immunity (Smith et al. 1991; Singer et al. 2000a).

Open space or habitat openness is important because bighorn sheep primarily rely upon sight to detect predators (Krausman et al. 1999). Horizontal visibility should exceed 62% of a square meter target when viewed at 28 m (30 yards) with the observer's eyes 90 cm (3 ft) above the ground. Invasion of tall shrubs or trees diminishes the value of bighorn sheep habitat, and fire is often prescribed to improve bighorn habitat (Krausman and Shackleton 2000).

Seasonal and Special Ranges. Seasonal and special "ranges" to which adult bighorns show a high degree of fidelity may include summer, winter, spring (lambing), fall (rutting), and salt lick ranges (Shackleton et al. 1999). Bighorns are unable to forage efficiently where snow depths exceed 25 cm (10 in). Under these conditions, they seek southerly slopes or wind-swept ridges near suitable escape terrain where forage remains available or migrate to lower elevations (Krausman and Shackleton 2000). Habitat suitability parameters for seasonal and special ranges are similar to those described above, and, often, a lack of a suitable seasonal range (especially winter range) or special range (especially lambing range) limits overall suitability of an area for bighorn sheep (Smith et al. 1991). A variety of environmental or behavioral stimuli may dictate seasonal habitat shifts. For example, desert bighorn sheep may respond to seasonal water availability (e.g. ephemeral water catchments) and use areas that are unsuitable during dry seasons (Krausman and Shackleton 2000). Ewes may make seasonal movements to more suitable lambing areas as defined above. Seasonal and special habitat requirements must be considered when assessing overall habitat suitability for bighorn sheep.

FOOD HABITS / FORAGING BEHAVIOR

Bighorn sheep occupy a unique grazing niche on steep slopes (Singer et al. 2000b). They tend to be more opportunistic than preferential in their choice of foods, eating whatever palatable foods are available (Krausman and Shackleton 2000). In terms of plant types, Rocky Mountain bighorns tend to show a preference for forbs, grasses, and browse in that order. The order of preference is reversed for desert bighorn sheep, with no clear preference between grasses and forbs (Krausman and

Shackleton 2000). Variations from these general preferences are common, especially when examined seasonally.

POPULATION DYNAMICS

Life History. Bighorn sheep are a wild, native member of the bovine family (Bovidae). Adult male Rocky mountain bighorns weigh about 79 kg (174 lbs) on average and females weigh about 59 kg (130 lbs); while desert bighorns are slightly smaller with males and females averaging 68 and 52 kg (150 and 114 lbs), respectively (Valdez and Krausman 1999). Large Rocky Mountain bighorn rams may weigh up to 145 kg (319 lbs) and large desert bighorn rams may exceed 100 kg (220 lbs) (Valdez and Krausman 1999). Both male and female bighorns reach sexual maturity at about 18 months; but in the wild, females first mate at 2.5 years or later, and males do not fully participate in the rut until 7-8 years of age (Krausman and Shackleton 2000). Bighorn sheep in expanding populations may reproduce at younger ages. Breeding in Rocky Mountain bighorn sheep usually peaks between mid-November and mid-December, but the breeding season for desert bighorns is much more variable and apparently influenced by environmental conditions at the time of birth (Krausman and Shackleton 2000). Based on observed birth dates ranging from late December to late May on the Fra Cristobal Mountains in south-central New Mexico (Z. Parsons, personal communication), breeding by desert bighorns there occurred from early July through early December. Males fight by clashing horns during the rut to compete for the opportunity to breed females. Larger-horned males generally do most of the breeding and are preferred by females (Krausman et al. 1999). Gestation lasts about 175 days and lambs weigh 3-5.5 kg (6.6-12.1 lbs) at birth (Krausman and Shackleton 2000). Ewes usually bear single lambs; twinning is extremely rare. Most bighorn sheep die before age 10, a few may survive to age 15 (Krausman et al. 1999).

Like most ungulates, bighorn sheep occur in groups of 2 to 100 individuals. Except for juvenile males remaining with their mothers, groups are sexually and spatially segregated for all seasons except the rut (Krausman and Shackleton 2000). Bighorn sheep are active during daylight hours (diurnal). Feeding behavior is most active in early morning and late afternoon and bouts of resting and feeding alternate during the day (Krausman et al. 1999; Shackleton et al. 1999).

Population Density. Reported densities of bighorn sheep populations range from 0.33 to 7.7 sheep per km² (0.8-20/mi²) (Zeigenfuss et al. 2000). Zeigenfuss et al. (2000) found that population densities of successful or marginally successful

translocated populations of bighorn sheep were between 0.57 and 1.53 sheep per km² (1.5-4/mi²), and that populations exceeding 3.0 sheep per km² (7.8/mi²) were released into patches of less than 10 km² (4 mi²) of suitable habitat. None of these higher-density populations exceeded 40 individuals. This suggests that higher reported bighorn sheep population densities may be artifacts of selected restoration sites being too small. Zeigenfuss et al. (2000) recommend the use of density estimates of 1.47 Rocky Mountain bighorns/km² (3.8/mi²) and 0.33 desert bighorns/km² (0.9/mi²) when planning restoration projects.

Home Range. Information on home range size of Rocky Mountain bighorn sheep is limited (Shackleton et al. 1999), and the measurement of home range is complicated by seasonal movements. However, it is well known that mature Rocky Mountain bighorns exhibit a high degree of fidelity to established seasonal ranges and return to them yearly (Shackleton et al. 1999). Numerous studies of desert bighorn sheep indicate that home ranges of desert bighorns include summer range close to permanent water, fall-winter range that may overlap with summer range, and spring (lambing) range with precipitous terrain close to water (Krausman et al. 1999). One study of desert bighorns in Utah delineated home ranges of 61 km² (23.5 mi²) for males and 24 km² (9.3 mi²) for females (Jense et al. 1979, as cited in Krausman et al. 1999). Being herd animals, many sheep would share home ranges.

Causes of Death. As with all ungulates, mortality rates are often highest (40-90%) during the first year of life. Predation is a common cause of lamb mortality. Other causes of lamb mortality may include pneumonia, severe weather, inbreeding depression, poor maternal nutrition, inattentive mothers, and human disturbance (Krausman and Shackleton 2000). Potential predators of bighorn sheep of various ages include coyote (*Canis latrans*), mountain lion (*Puma concolor*), wolf (*Canis lupus*), bobcat (*Lynx rufus*), gray fox (*Urocyon cinereoargenteus*), golden eagle (*Aquila chrysaetos*), and humans (Kelly 1980; Fitzgerald et al. 1994). Bighorn sheep are well adapted for avoidance of predation, provided escape terrain is nearby; and predation usually has little effect on population survival (Kelly 1980; Krausman et al. 1999).

Disease-related "die-offs" can kill 35-75% of a sheep population in a single year and can suppress recruitment for an addition 3-7 years (Gross et al. 2000). While bighorn sheep suffer from a variety of diseases, they are particularly susceptible to pneumonic pasteurellosis (*Pasteurella* spp.) which they readily contract from domestic sheep and goats and their exotic relatives (Smith et al. 1991; Gross et al. 2000). Most, usually all,

bighorn sheep die when placed in pens with domestic sheep, and evidence strongly suggests the transmission of fatal diseases from domestic to wild sheep in the wild (Singer et al. 2000a). About 85% of the San Andres Mountain (NM) herd of desert bighorns died from a virulent outbreak of contagious ecthyma caused by scabies mites (*Psoroptes ovis*). Various sources of stress and/or the presence of lungworm (*Protostrongylus* spp.) likely predispose bighorn sheep to disease die-offs (Krausman and Shackleton 2000).

Other causes of bighorn sheep mortality include vehicle collisions, natural accidents, drowning, and fence entanglement (Krausman et al. 1999).

Population Structure and Viability. Suitable habitat for bighorn sheep tends to be comprised of climax vegetation types that change slowly and occur in “islands” within and among mountain ranges (Douglas and Leslie 1999; Singer et al. 2000c). Natural habitat patchiness tends to cause individual herds to be small and disjunct in their distribution. Low reproductive potential, the tendency of ewes to have strong fidelity to their natal home ranges, the distance between patches of suitable habitat, and barriers to dispersal between patches results in infrequent colonizations of vacant habitats by bighorns (Bleich et al. 1990; Douglas and Leslie 1999; Singer et al. 2000c). However, Bleich et al. (1990) suggest that natural local extinctions and recolonizations may have been more common than previously thought.

Thus, bighorn sheep appear to be predisposed to a metapopulation structure, where limited interchange (mostly by rams) occurs among smaller, geographically separated subpopulations (Lande and Barrowclough 1987; Bleich et al. 1990; Singer et al. 2000c). Singer et al. (2000c) speculate that bighorns historically existed mostly in metapopulations and that human disturbances have accelerated extinction rates causing a current state of population disequilibrium and the existence of much unoccupied habitat. Ramey et al. (2000) advise that the establishment of single isolated populations of bighorn sheep with no potential for genetic exchange with other populations is of little long-term conservation value. They advocate the establishment of metapopulations.

A present-day example of a potentially viable bighorn sheep metapopulation exists in southeastern California in the vicinity of the community of Twentynine Palms. There, about 1,000 bighorn sheep occupy 15 of 31 mountain ranges, 10 currently vacant ranges previously supported bighorns, only 8 of the 15 subpopulations currently exceed 50 sheep, and bighorn movements of 6-20 km (3.7-12.4 mi) have been documented between 11 pairs of mountain ranges (Bleich et al. 1990).

Reestablishment of bighorn populations in key mountain ranges is critical to the restoration of connectivity among currently isolated subpopulations in this region. This metapopulation is bounded by significant barriers to dispersal and migration (i.e., interstate highways and the Colorado River).

Because ideal situations rarely exist in nature, the “genetically effective population size” (N_e) is probably always smaller than the actual census size (N_c) of the population (Meffe and Carroll 1997:172). A genetically effective population is generally defined as an ideal, stable population with randomly mating individuals, even sex ratio, equal birth rates among females, and nonoverlapping generations (Meffe and Carroll 1997). Douglas and Leslie (1999) believe that habitat fragmentation and local extinctions of subpopulations has led to effective population sizes that may be orders of magnitude smaller than the census count, and that small populations cannot persist without reproductive interactions with nearby populations. The need to preserve and restore functioning metapopulations is underscored by the fact that about two-thirds of all populations of Rocky Mountain and California (*O. c. californiana*) bighorn sheep populations have less than 100 individuals (Gross et al. 2000). Krausman and Leopold (1986) reported that only 7 of 59 populations of desert bighorn sheep in Arizona exceeded 100 animals in size. Today, populations in New Mexico range from 25-300 animals for six populations of Rocky Mountain bighorns and 9-71 animals for 6 populations of desert bighorns (E. Goldstein and Z. Parsons, personal communications).

While admitting that the concept of minimum viable population size in bighorn sheep is controversial, Singer et al. (2000b) consider metapopulations of 300-500 animals to be viable for at least 200 years, populations of 100-299 animals to be secure for shorter time periods, populations of 75-99 animals to be moderately secure, and populations of less than 50 animals to be vulnerable to extinction. They recommend restoration of single populations of 125 or more animals in clustered restoration sites 12-25 km (7.5-15.5 mi) apart, sufficient to support a combined metapopulation of more than 400 animals. Dunn (1994) recommended that subpopulations of desert bighorns in New Mexico be 15 km (9.3 mi) or less apart to be considered part of a functioning metapopulation.

Singer et al. (2000c) concluded that ultimate population size of translocated bighorns was correlated with N_e of the founding population, number of source populations represented in the founding group, and early contact with a second population. In the populations they studied, successful colonizations of suitable habitat patches occurred every 13.5 years on average for increasing populations and every 22 years for all populations

(increasing and decreasing). This indicates that time frames for natural recolonization of vacant habitat patches are quite long.

Bleich et al. (1990) defined the general requirements for a bighorn sheep preserve size as (1) sufficient suitable unoccupied habitat for future establishment of subpopulations and (2) large enough to support populations with long-term genetic health and the potential for long-term evolutionary processes to occur. In a study of reintroduced bighorn sheep populations in the Intermountain West, Singer et al. (2000c) found that the mean patch size for populations that increased to more than 100 individuals was 490 km² (190 mi²); whereas, the mean patch size for populations that declined to less than 30 bighorns or went extinct was 60 km² (23 mi²). Dunn (1994) evaluated the southern San Andres Mountains, which until recently supported the last remnant native herd of desert bighorns in New Mexico, and identified 421 km² (163 mi²) of suitable desert bighorn sheep habitat. This area historically supported 200 or more sheep. Thus, bighorn density in the southern San Andres Mountains was approximately 0.5 bighorns/km² (1.3/mi²), which is comparable to densities reported by Zeigenfuss et al. (2000).

Zeigenfuss et al. (2000) recommend a minimum patch size of 200 km² (77 mi²) and Singer et al. (2000c) recommend more than 400 km² (154 mi²) for bighorn restoration projects. These recommendations stand in sharp contrast to findings by Johnson and Swift (2000) that populations of 125 or more Rocky Mountain bighorns have persisted for 20-52 years in areas with 6-9 km² (2.3-3.5 mi²) of suitable habitat; and to the recommendation of Smith et al (1991) of 17 km² (6.6 mi²) of suitable habitat for successful (≥ 125 bighorns) restoration projects. The findings and recommendations of Johnson and Swift (2000) and Smith et al. (1991) suggest population densities of 7-21 bighorns/km² (18-54/mi²), which are much higher than population densities reported by other authors (Zeigenfuss et al. 2000).

Bleich et al. (1990) recommended that small, isolated patches of suitable habitat should be recognized as potentially important habitat and included within preserve designs as potential seasonal habitats and “stepping stones” for migration or dispersal movements.

Krausman and Leopold (1986) cautioned that the importance of small populations should not be discounted in bighorn sheep conservation programs. And Smith et al. (1991) advised that subpopulations of less than 125 bighorns can contribute to the formation of viable metapopulations provided movement among populations can occur.

We base our design recommendations on mean population densities of 1.0 bighorn/km² (2.6/mi²) of suitable habitat, preferred minimum subpopulation size of 125 bighorns, minimum metapopulation size of 400 bighorns, and minimum inter-patch distances within a metapopulation of 20 km (12.4 mi). Thus, optimum subpopulation habitat patch sizes should be at least 125 km² (48 mi²) with at least 360 ha (890 acres) of suitable lambing habitat (Smith et al. 1991). However, habitat patches of 10 km² (4 mi²) or more near similar sized or larger patches or within potential migration or dispersal corridors should be protected for their potential to provide seasonal, special, or temporary habitats; habitat for small interconnected subpopulations; and “stepping stones” for migration or dispersal movements.

Population Status. Current estimates of bighorn sheep populations in the NM Highlands planning area are as follows (E. Goldstein and Z. Parsons, personal communications): Rocky Mountain bighorn populations—Pecos Wilderness (300), Wheeler Peak (148), San Francisco River (65), Latir Wilderness (55), Manzano Mountains (30), and Turkey Creek (25); desert bighorn populations—Fra Cristobal Mountains (71), Sierra Ladrones (26), and San Andres Mountains (9).

MOVEMENTS

Dispersal. Dispersal is important to bighorn sheep for recolonization of unoccupied suitable habitat patches, gene flow among subpopulations, maintenance of the evolutionary potential of metapopulations, and for discovery of newly-created suitable habitat caused by fires or removal of livestock (Bleich et al. 1990; Singer et al. 2000c). Generally, bighorn sheep are considered to be poor dispersers and colonizers of unoccupied habitats (Gross et al. 2000); but nevertheless, moderate rates of dispersal are critical to the survival and long-term persistence of self-perpetuating metapopulations of bighorn sheep. Singer et al. (2000c) found that most colonizations of new habitat (n=24) by dispersing bighorns from 31 translocated populations were of patches 10-15 km (6.2-9.3 mi) distant; and Gross et al. (2000) computed that the maximum probability of dispersal was to a patch 12.3 km (7.6 mi) away. Both ewes and rams disperse but rams disperse more frequently than ewes and generally disperse longer distances (Bleich et al. 1990; Douglas and Leslie 1999; Wehausen 1999).

Migration. Under natural conditions most bighorn sheep populations (rams and ewes) migrate seasonally among 2-7 different seasonal ranges separated by distances of 8-18 km (5-11 mi) with elevation changes that may exceed 1,000 m (3,300 ft)

(Shackleton et al. 1999; Krausman and Shackleton 2000; Singer et al. 2000b). Factors influencing migration or movements in bighorn sheep may include home range knowledge, water and forage availability, location of mineral licks, snow accumulation, lambing and mating activities, season, presence of biting insects, topography, and age and sex class status (Krausman et al. 1999; Krausman and Shackleton 2000). In a review of 100 bighorn sheep translocation projects, Singer et al. (2000a) found that 100% of fully migratory populations were successful while only 65% of nonmigratory populations were successful. Rocky Mountain bighorns may migrate as far as 64 km (40 mi) annually and desert bighorns as far as 30 km (19 mi) (Singer et al. 2000a). The absence of migration in some present-day herds may be the result of human-caused habitat alterations and fragmentation and barriers within traditional migration routes (Singer et al. 2000a).

Barriers to Movement. Barriers that impede movement of bighorn sheep include sheer cliffs, wide valley floors, swift or wide rivers, lakes or reservoirs, dense vegetation with low horizontal visibility (< 30%), fences (if not designed for wildlife passage), motorized recreational activities, domestic livestock (especially sheep), concrete lined canals, highways (state, federal, and interstate) with 600 or more vehicles per day, and high-density centers of human activity (Bleich et al. 1990; Smith et al. 1991; Fitzgerald et al. 1994; Johnson and Swift 2000; Zeigenfuss et al. 2000).

Use of Linkages. Bighorn sheep habitat usually occurs as small, isolated patches within a larger matrix of unsuitable habitat. By necessity, dispersing or migrating bighorns usually must move through areas of unsuitable habitat (Bleich et al. 1990; Krausman and Shackleton 2000). Bleich et al. (1990) summarize the documentation of such movements by numerous researchers. Singer et al. (2000c) found that, compared to potential linkages not used by bighorns, linkages used by bighorns to recolonize vacant habitat patches were characterized by fewer water barriers, more open vegetation, and more rugged, broken terrain. They observed much higher rates of colonizations by bighorn sheep than observed by previous researchers, which supports increased attention by managers to the protection and restoration of linkages among potentially suitable habitat patches (Bleich et al. 1990; Krausman and Shackleton 2000; Singer et al. 2000c).

ECOLOGY

Interspecies Interactions. Like other large ungulates, bighorns consume grasses, forbs, and shrubs (Krausman and

Shackleton 2000). Considerable diet overlap exists between bighorns and elk, mule deer, cattle, domestic sheep, goats, horses, and burros (Krausman et al. 1999; Shackleton et al. 1999). Competition for food resources may affect various population parameters of bighorn sheep, such as lamb survival and recruitment, but such interactions have not been confirmed (Shackleton et al. 1999). However, Krausman et al. (1999) attribute the precipitous historic decline of desert bighorn sheep populations in the western U.S. to overgrazing by domestic livestock, but admit that such a cause-and-effect relationship is difficult to document. One study in Nevada documented that bighorn sheep density was three times higher in ungrazed versus grazed (by domestic cattle) habitats (Krausman et al. 1999). Krausman et al. (1999) predicted that competition for food resources by aoudad and Persian wild goats (exotic species) will eventually extirpate bighorn sheep populations using the same range. Population declines of bighorns also have been attributed to disease transmission from domestic sheep and goats and exotic wildlife (Krausman et al. 1999; Shackleton et al. 1999).

Some large predators prey on bighorn sheep, but bighorns' adaptations for predator avoidance—keen eyesight and sure-footedness in steep escape terrain—appear quite effective. Predation is unlikely to limit bighorn populations in areas with adequate escape terrain, but may explain their absence in areas deficient in escape terrain (Krausman et al. 1999; Shackleton et al. 1999). However, predation by mountain lions has been documented or suspected to be a major limiting factor in some bighorn populations, especially recently translocated populations (Krausman et al. 1999). In most (perhaps all) cases in which mountain lions endanger small populations of bighorn sheep, the impact of predation has been exacerbated by disease (e.g., scabies and pneumonia – Logan and Sweanor 2001), woody plant invasion due to overgrazing or fire suppression (Sweitzer et al. 1997), or artificially high lion populations subsidized by year-round livestock operations (E. Rominger, NMDGF, personal communication). Short-term control of mountain lions may be necessary as an emergency measure to prevent extinction of some populations of bighorns, but long-term conservation programs should follow a holistic approach.

Ecological Effects. Bighorns occupy a unique grazing niche on steep slopes (Singer et al. 2000b). Grazing and browsing by bighorns affects the availability of vegetation and successional processes within plant communities (Singer et al. 2000b). In addition to carrying out the ecological function of herbivory, bighorns are prey for large carnivores such as wolves, mountain lions, and bears (Singer et al. 2000b).

JUSTIFICATION AND FOCAL VALUE

Flagship. The bighorn sheep is a majestic, charismatic animal with considerable public appeal among both hunters and nature enthusiasts. The bighorn sheep is a focus of a major foundation—the Foundation for North American Wild Sheep—that is allowed to auction special hunting permits to benefit conservation programs for the species. The most recent permit for the opportunity to hunt one bighorn ram in New Mexico sold for \$75,000. To date \$759,500 has been raised for bighorn sheep conservation in New Mexico through this program. The restoration of bighorns from their near extirpation in the early 1900s is an ongoing conservation priority. Conservation programs that benefit bighorn sheep should readily capture the public's attention and interest.

Habitat Quality Indicator. Bighorn sheep are habitat specialists that graze in steep, rugged terrain. While naturally fragmented, habitats required by bighorns have become increasingly fragmented by human activities. In addition, important dispersal and migration routes have been rendered ineffective by human-caused barriers to bighorn movements. Bighorns are intolerant of human disturbances and activities, except in situations where the predictability of nonthreatening human activity results in habituation by bighorns. The presence and persistence of bighorn populations is a clear indicator of the quality of the interconnected steep-sloped habitats they require.

Wilderness Quality Indicator. “Mountain sheep epitomize wilderness. They occupy some of the most inaccessible, rugged, and spectacular habitats in North America. Their ability to negotiate precipitous terrain is legendary....Only the most adventurous and hardy outdoor enthusiasts dare to tread in such hostile habitats of temperature extremes and rugged terrain.” (Valdez and Krausman 1999:3).

MANAGEMENT RECOMMENDATIONS

Krausman and Shackleton (2000:533) summarize management needs for bighorn sheep as follows: “Habitat for bighorn sheep still exists in the west, but managers (and the public) have to ensure that sufficient habitat is protected, that movement corridors remain open, that human disturbance is reduced or kept to a minimum, and that transmission of diseases from livestock is eliminated. Only if these are accomplished will efforts to enhance viable populations of bighorn have a chance to be successful.” Underlining was added for emphasis.

Reestablish Populations. Transplantation has been proven to be an effective means of establishing bighorn sheep in vacant suitable habitats (Douglas and Leslie 1999). Over half of extant populations of bighorns in the western U.S. result from translocations (Singer et al. 2000a). Singer et al. (2000b) developed the following seven point restoration plan: (1) survey existing populations, (2) conduct GIS-based habitat assessments, (3) review of habitat assessments by a scientific advisory panel, (4) convene interagency panel to discuss metapopulation management and plan restoration projects, (5) draft interagency restoration and management plans, (6) conduct restoration activities, such as translocations, and (7) monitor populations and evaluate success of restoration efforts. Singer et al. (2000a) noted that translocation success was about twice as high when founders came from indigenous populations as compared to previously translocated herds. They further recommended that the founder herd size be 41 or more animals, that translocation areas be at least 20 km (12.4 mi) from domestic sheep, and that migration routes among seasonal ranges be free of barriers to bighorn movements. Singer et al. (2000c) recommended that founders be selected from more than one source population; and Singer et al. (2000d) cautioned that removals of reintroduction stock should be only from increasing source populations and should not exceed 5% of the source population annually.

Protect Habitats. Bighorn sheep require specialized habitats that are in short supply in the NM Highlands area; and bighorn populations are few and generally small. No further degradation of suitable bighorn sheep habitat should occur.

Improve Habitats. Many habitats may be suitable for bighorns with certain improvements. Where forests and dense shrubs have invaded otherwise suitable bighorn habitat, prescribed fires that mimic natural historical fire regimes may reestablish necessary forage quantity and quality and visibility for bighorns. Development of reliable water sources may improve habitats for bighorns (Krausman and Shackleton 2000); but Krausman and Leopold (1986) caution that provision of water sources may increase competition for food resources by attracting other ungulates, such as mule deer. Additionally, opportunities for predation may increase at or near water sources (Z. Parsons, personal communication).

Minimize Human Disturbances. Managers should monitor levels and effects of human activities in crucial bighorn habitats and implement measures to reduce or eliminate human disturbances that may jeopardize the long-term health and persistence of bighorn populations (Krausman and Shackleton 2000).

Protect and Restore Linkages. Singer et al. (2000a) found that sheep populations that migrate among seasonal ranges are more successful than those that don't or can't migrate. Most bighorn sheep populations exist in the form of metapopulations, which require periodic movement of individuals among subpopulations for long-term viability and persistence (Bleich et al. 1990; Singer et al. 2000b). Managers should recognize key habitat linkages and focus on preventing or eliminating barriers to critical bighorn movements between patches of suitable habitats.

Remove Domestic Sheep. The presence of domestic sheep within 20 km (12.4 mi) of existing bighorn herds or habitats suitable for the restoration of bighorns poses serious problems for bighorn sheep protection and restoration efforts (Singer et al. 2000a). With domestic sheep nearby, the potential for transmission of lethal diseases to bighorn herds is very high, because bighorn rams wander widely during rut and have been known to breed domestic sheep and carry diseases back to their herd (Gross et al. 2000; Ramey et al. 2000). Ramey et al (2000) suggest that the most cost-effective solution may be to buy out and retire domestic sheep allotments on public lands. Similar incentives may induce private livestock operators to stop grazing domestic sheep near bighorn herds or habitats suitable for bighorn restoration.

Eliminate Cattle Grazing in Bighorn Habitat. A high degree of dietary overlap exists between bighorn sheep and cattle (Krausman et al. 1999; Shackleton et al. 1999). Singer et al. (2000a) found that translocated populations of bighorns increased faster on ranges where cattle were absent and that translocation success rate declined by 27% when cattle grazed restoration areas. A study in Nevada documented that bighorn sheep density was three times higher in ungrazed versus grazed (by domestic cattle) habitats (Krausman et al. 1999). Given the fragile nature of bighorn populations, every advantage should be sought to enhance populations and the success of reestablishment efforts.

Multi-Jurisdictional Planning. Douglas and Leslie (1999) advised that, where bighorn sheep populations cross jurisdictional boundaries (which is nearly always the case), agencies and private landowners develop joint management plans and strategies to ensure long-term persistence of bighorn sheep populations.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of bighorn sheep ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs

accurate information and knowledge about bighorn sheep to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

Elk (*Cervus elaphus nelsoni*)

INTRODUCTION

The Rocky Mountain elk or wapiti (*Cervus elaphus nelsoni*) is the largest native member of the mammalian family Cervidae in New Mexico and is widely distributed throughout the NM Highlands planning area. Ecologically, elk are grazing and browsing herbivores that are primary prey for large predators. They occupy habitats ranging from shrublands to subalpine coniferous forests and prefer a mosaic of interspersed habitat types that offer both cover and forage. Elk require relatively large areas of suitable habitat for population viability and migrate seasonally between low and high elevation winter and summer ranges. The elk is protected by the New Mexico Department of Game and Fish as a game mammal, and most populations within the NM Highlands planning area are hunted. Elk are abundant in the NM Highlands planning area and populations are secure. The elk's popularity with both hunters and nonconsumptive users fosters considerable public interest and support for nature conservation.

DISTRIBUTION

Historic. Originally, elk (*Cervus elaphus* spp.) ranged throughout much of the United States except in New England, the Southeast, most of Texas, and the desert regions of the Southwest (Hall 1981). They were both a plains and mountain dwelling animal (Peek 1982). Elk also occurred in western Canada and in the northern reaches of the Sierra Madre Oriental and Occidental in Mexico (Hall 1981; Bryant and Maser 1982). Two subspecies of elk originally inhabited New Mexico: *C. e. nelsoni* (Rocky Mountain elk) in the Sangre de Cristo, San Juan, and Jemez Mountains in the north; and *C. e. merriami* (Merriam's elk) in the Mogollon, Datil, Gallina, Sacramento, White, and Guadalupe Mountains in the south. Merriam's elk became extinct around 1900 (Bailey 1931). The Rocky Mountain elk was also driven to near or possibly complete extirpation in northern New Mexico in the early 1900s (Bailey 1931). Elk have been restored to virtually all suitable range through restocking programs (Peek 1982) and may exceed their historical abundance (Truett 1996).

Current. Present elk populations are generally restricted to forested areas in western North America (Fitzgerald et al. 1994). Elk reintroduction efforts began in the early 1900s (Bailey 1931) and viable populations of elk have been restored throughout and perhaps beyond their historic range in New

Mexico (Truett 1996), except in the eastern plains. All reintroductions have been of the subspecies *C. e. nelsoni* (Findley et al. 1975).

Potential. Elk have recolonized historically occupied habitats within the NM Highlands planning area. Additionally, elk populations have expanded, and appear to be continuing to expand, into nontraditional habitats, such as the Chaco Canyon area (Darrel Weybright, NMDGF, personal communication).

HABITAT

General. Elk tend to associate with forest edges (the interface of forests and openings, also called "ecotones"), and require a mosaic of early, forage-producing stages and later, cover-forming stages of the forest sere in close proximity (Skovlin 1982; Thomas et al. 1988; Verts and Carraway 1998). Use of available forage by elk decreases with distance from cover, with about 90% of foraging occurring within 120 m of adequate cover (Verts and Carraway 1998). The quality or "effectiveness" of habitat for elk is a function of (1) size and spacing of cover and forage stands, (2) cover quality, (3) roads open to vehicular traffic, and (4) quantity and quality of forages (Thomas et al. 1988).

Elk tend to underutilize habitats near roads, especially during hunting season and when nearby hiding cover is scarce (Canfield et al. 1986; Millspaugh et al. 2000). Unsworth et al. (1998) found that elk used areas with greater canopy cover in roaded areas. Compared to habitat use in roadless areas (the standard for comparison), Lyon (1983) found that habitat use by elk declined to 75% at a road density of 1 mi/mi² (0.6 km/km²) and to only 10% at a road density of 7 mi/mi² (4.3 km/km²). Rowland et al. (2000) documented elk avoidance of open roads to a distance of 1.8 km (1.1 mi), and found that distance to open roads was a better predictor of elk avoidance than was open road density. Bull survival rates during hunting seasons in areas with road densities of less than 1 mi/mi² (0.6 km/km²) were nearly double the rate of survival in areas with more than 4 mi/mi² (2.5 km/km²) (Christensen et al. 1993), presumably because hunting pressure is greatest near roads. Road use restrictions can improve habitat suitability for elk. In southeastern Oregon, closure of roads to all but necessary administrative use (4 or less trips/week) significantly reduced home range and core activity area sizes and daily movements for Roosevelt elk and significantly increased survival of cow elk (Cole et al 1997). An effective method for improving elk habitat is road closures or severe road use restrictions, assuming the habitat is otherwise suitable for elk (Lyon 1983; Cole et al 1997).

Elk tend to avoid areas where cattle are present if other options exist, but will tolerate some cattle use when no cattle-free habitat is available (Christensen et al. 1993). In central Arizona, Wallace and Krausman (1987) found that the distribution and habitat use by elk changed significantly when cattle were introduced to the range. Elk use was significantly less on pastures grazed by cattle than on pastures not grazed by cattle. Use of habitats by elk shifted from open mesic and logged sites to more closed forest following introduction of cattle. Coe et al. (2000) also found that elk avoided areas used by cattle. Direct food competition with cattle and shifts in habitat use reduce the amount of forage available to elk (Lyon and Ward 1982; Wallace and Krausman 1987). Wisdom and Thomas (1998) summarized the evidence for competition between cattle and elk; they concluded that the potential for exploitative competition between the two ungulates was high, owing to strong similarities in diet and areas of use. However, Wisdom and Thomas (1996) also summarized the many ways in which competition between cattle and elk might be reduced, and provided methods to evaluate trade-offs in stocking rates between the two ungulates. Competition also exists between elk and domestic sheep and elk's social intolerance of sheep apparently exceeds that of cattle (Nelson 1982; Peek 1982; Fitzgerald et al. 1994).

Elk tend to avoid areas of human activity (Morrison et al. 1995; Phillips and Aldredge 2000). In the Lamar and Stephen's Creek drainage of Yellowstone National Park, elk fled when cross-country skiers approached within 400 m (438 yards; median flight distance). Generally, elk moved uphill to steeper slopes, away from roads, and closer to trees. They tended to use ridges as primary cover (by crossing over them) before moving into trees for additional cover. The median distance moved was 1,675 m (1 mi). Elk moved completely out of the drainage following 78% of disturbance events and usually returned within 2 days in the absence of further human activity (Cassirer et al. 1992). Elk are displaced from areas near logging and oil well operations to areas that provide security by vegetative cover or topographic barriers (Skovlin et al. 1989; van Dyke and Klein 1996). Kuck et al. (1985) found that elk responded to human and simulated mining disturbances (recorded mining noises) by moving across topographic barriers. Cow/calf pairs readily abandoned traditional calf-rearing areas following human and simulated mining disturbances. Daily distances moved, elevational changes, and size of use areas all increased significantly in response to disturbances (Kuck et al. 1985). Because elk have strong home range fidelity, temporary displacement has the effect of reducing home range size for the duration of the disturbance (Edge et al. 1985). In winter, the additional energy demands placed on elk after disturbance

by winter recreationists may adversely affect elk survival, especially on poor winter ranges (Thomas et al. 1988).

Preferred. Because elk are an ecotone species, optimum elk habitat contains patches of forest stands that vary between being dense enough to provide good thermal and hiding cover and open enough to promote good growth of forage (Skovlin 1982). Small openings are also important for forage production. The optimum ratio of thermal and hiding cover to feeding areas appears to be about 40:60 (Skovlin 1982). The greatest use by elk is within 274 m (300 yards) of the edge for cover and 91 m (100 yards) for forage areas (Leckenby 1984 as cited by Thomas et al. 1988). Thus, optimum widths of cover and forage patches (assuming elk is the sole species of management interest) would be 550 m (600 yards) and 183 m (200 yards), respectively (U.S. Forest Service 1986). Elk tend to respond positively to both natural (e.g., fire) and man-induced (e.g., clearcuts) disturbance regimes, provided the effect is a patchy environment where openings are small. In the Blue Mountains of northeastern Oregon, Skovlin et al. (1989) found that elk use increased significantly on small (0.8-8.5 ha or 2-21 acres) clearcuts (slash was piled and burned), but returned to pre-logging levels in five years following logging. They concluded that elk were attracted to increased forage available in newly logged clearcuts, but relied on adjacent uncut forests (where elk use was also high) for concealment and thermal cover. Natural disturbance mechanisms (e.g., fire, avalanche, wind storm) could produce a similar effect.

Another important aspect of preferred elk habitat is the degree to which various plant community successional stages and cover types are intermixed within an area (Skovlin 1982). Such a mosaic pattern of interspersed stages maximizes the amount of ecotones, which are favored by elk (Skovlin 1982). A diversity of crown closures in cover stands and foraging areas on winter range will provide foraging options for elk in response to varying weather and snow conditions (Thomas et al. 1988). Elk prefer moderate slopes (15-40%) over flats and steeper slopes (Skovlin 1982; Fitzgerald et al. 1994).

Skovlin et al. (1982) believed that elk seek "thermal cover" for protection against a wide range of weather and temperature (both low and high) conditions, which would otherwise increase the energy cost of homeostasis. Thomas et al. (1988) and Verts and Carraway (1998) also assumed that elk require thermal cover near foraging areas, and defined such cover as trees at least 12 m (39 ft) tall with a canopy closure of at least 70%. Skovlin (1982) also speculated that thermal cover could be a topographic feature such as a small protected basin. A mosaic of diverse forest structure offers a range of cover choices

for elk in response to different climatic and foraging conditions. Preferred bedding sites are in timber clumps on upper slopes. During cold seasons, preferred day beds are on south-facing slopes and night beds are on north-facing slopes (Skovlin 1982). Slope preference reverses during warm seasons. Thermal forest cover on upper north-facing slopes provides the coolest summer habitats and high quality forage, provided the forest canopy cover is not continuous (Skovlin 1982).

In contrast to the discussions of thermal cover by Skovlin et al. (1982), Thomas et al. (1988), and Verts and Carraway (1998), the work of Cook et al. (1998) provided experimental evidence of the lack of thermal regulatory benefits of thermal cover. However, Wisdom and Cook (1998) noted that areas of dense cover, often defined as thermal cover, are consistently and strongly selected by elk, particularly during late summer. Consequently, thermal cover likely provides other benefits to elk beyond thermal regulation. Wisdom and Cook (1998) hypothesized that areas of dense cover dramatically alter the phenology and quality of elk forage compared to areas of more open canopy, and that these differences contribute to a more diverse mosaic of foraging conditions for the species. The forage benefits of thermal cover have not been studied well, and deserve further investigation.

Elk seek hiding cover for security from the threat of predation and harassment. Hiding cover is an important component of high-quality elk habitat and is defined as cover that conceals 90% of an animal at a distance of less than 61 m (76 yards) (Canfield et al. 1986; Thomas et al. 1988). The value of hiding cover can change with the viewing angle. For example, young timber stands that provide 100% hiding cover when viewed on a horizontal plane may provide substantially reduced cover when viewed at an elevated angle from an opposing slope (Canfield et al. 1986). The presence of large downed woody structures enhances hiding cover for elk (Winn 1976, cited by Skovlin 1982).

Elk appear to require water and tend to concentrate their habitat use to areas within 0.8 km (0.5 mi) of a reliable water source (Skovlin 1982). Wisdom and Thomas (1996) speculated that elk were historically sparse or absent across vast areas of arid and semi-arid rangelands of western North America, owing to the scarcity of open water. More recent establishment of elk populations in these areas appear to be associated with water developments, such as the widespread construction of stock tanks and guzzlers on livestock allotments.

Adult females select calving grounds in ecotones where cover, forage, and water (<400 m or 0.25 mi) are interspersed (Peek

1982; Fitzgerald et al. 1994). Human disturbance of elk on calving areas has been shown to reduce reproductive success (Phillips and Alldredge 2000). In Idaho and Montana, sagebrush cover on slopes of 35-40% provided important hiding cover for calves; in other areas, calf hiding sites were within forested patches near the edges of openings (Skovlin 1982). Migratory Rocky Mountain cow elk usually calve on transitional spring range during the migration between winter and summer ranges, but calving locations can vary as a result of abnormal climatic conditions (Skovlin 1982). Shrubs, downed logs, and topographic irregularities appear to enhance concealment of calves (Skovlin 1982).

Preferred habitat for elk would include the above habitat attributes on both summer and winter ranges in areas devoid of domestic livestock, with limited human use, and with road densities less than 0.6 km/km² (1 mi/mi²).

FOOD HABITS / FORAGING BEHAVIOR

Elk are both grazers and browsers; grasses are important dietary components year-round; forbs increase in dietary importance in summer; and shrubs may increase in importance in fall and winter (Peek 1982; Fitzgerald et al. 1994; Verts and Carraway 1998). Snow depths of more than about 61 cm (24 in) cause elk to shift from herbaceous forage to browse (Skovlin 1982).

Mushrooms may constitute up to 75% (dry weight) of the diet during late summer and fall when they are most abundant (Launchbaugh and Urness 1992). They may provide important sources of proteins and phosphorous late in the growing season. Another potentially important food item of elk is lichens (Leckenby 1984, cited by Thomas et al. 1988).

POPULATION DYNAMICS

Life History. Elk are large members of the deer family (Cervidae)—only moose are larger. Adult males weigh about 300 kg (660 lb) on average and females weigh about 250 kg (550 lb) (Peek 1982; Fitzgerald et al. 1994; Verts and Carraway 1998). Large bulls may exceed 450 kg (990 lb) (Peek 1982). The following life history information is from Fitzgerald et al. (1994) unless otherwise cited. Elk breed in the fall (September and October). Males spar during the rut for the opportunity to breed females, which they gather into protected harems of 15-20 cows. Females may have up to three estrous cycles at 21-day intervals during the rut. Gestation lasts 240-255 days and calves weighing about 14 kg (31 lb) are born in late May or

early June. Twin calves are rare. Calves hide for the first 18-20 days of their lives, then join “nursery groups” and remain in cow-calf groups through the summer (Peek 1982). Yearling cows can breed, but less than a third maintain calves into the fall. Females are most productive between the ages of 4 and 10 (Verts and Carraway 1998). Elk are known to live beyond age 20 (Peek 1982). Because of competition for cows, most breeding bulls are at least 3 years old. Sex ratios are about 1:1 at birth. Elk are generally most active at twilight and during the night.

Population Density. Accurate censuses of elk populations are difficult to obtain and are rarely conducted. Population estimates are usually derived from mathematical models that rely on available scientific knowledge and various population indices, such as sex and age composition of herds, bull:cow and calf:cow ratios, and hunter harvest statistics. While elk population densities vary with habitat suitability and the quantity and quality of forage, a density of 1 elk/km² (2.6 elk/mi²) would probably be about average for suitable habitats within the NM Highlands planning area. Population density estimates are made more complex by the seasonal migration of many elk herds.

Home Range. Few data are published regarding the home ranges of elk. Bear (1989) observed home ranges of 8.5 to 18.1 km² (3.3 to 7.0 mi²) in Colorado with no significant differences between cows, calves, bulls, and yearlings. In northeastern Oregon, home ranges of female elk were inversely correlated to the availability of thermal cover (Verts and Carraway 1998). Being herd animals, many elk would share home ranges.

Causes of Death. Elk mortality is mostly due to predation on calves, hunting, and winter starvation (Peek 1982; Fitzgerald et al. 1994; Ballard et al. 2000). Poaching was an important cause of mortality for female Roosevelt elk (Cole et al. 1997). Important elk predators include black bear (*Ursus americanus*), mountain lion (*Puma concolor*), gray wolf (*Canis lupus*), grizzly bear (*Ursus arctos*), and coyote (*Canis latrans*) (Peek 1982; Taber et al. 1982). Generally, young, old, sick, malnourished, and physically debilitated animals suffer higher rates of predation. Severe winter weather causes increased elk mortality and may predispose the following year's calf crop to increased mortality rates because of low birth weights or late birth dates (Taber et al. 1982). If light- or late-born elk survive through fall, they often enter winter in poor condition making them more prone to die, especially during severe winters (Taber et al. 1982). Bulls deplete energy reserves during rut which make them more vulnerable than cows to winter starvation (Taber et al. 1982). Fatal injuries may result from combat between bulls

during rut (Peek 1982; Taber et al. 1982). In heavily hunted populations, most males die before the age of 2 and most females before the age of 5; and in less heavily hunted populations, about half of males die by age 3 and half of females by age 5 (Fitzgerald et al. 1994). Harassment by humans may force elk onto inferior winter range resulting in increased mortality (Taber et al. 1982). Many elk are killed from colliding with vehicles on roads and highways. While highway mortalities may be substantial, there is no evidence to suggest that this source of mortality limits elk populations.

Diseases and parasites are not considered to be major causes of mortality in elk, except in conjunction with malnutrition (Peek 1982). However, Miller et al. (2000) predict that chronic wasting disease could reach epizootic levels in northeastern Colorado and southeastern Wyoming at some unspecified, but distant (decades), future time.

Population Structure and Viability. Sex ratios are about 1:1 at birth, but even in unhunted herds adult cows may be more than twice as numerous as adult bulls (Peek 1982; Fitzgerald et al. 1994). Bulls are physically and physiologically stressed by the rut and often enter winter in poor condition resulting in greater winter mortality (Peek 1982; Fitzgerald et al. 1994). Ratios of adult bulls to adult cows range from about 40:100 in unhunted populations to 10:100 in heavily hunted populations (Peek 1982). Elk are polygynous with dominant bulls mating with harems of 15-20 cows (Fitzgerald et al. 1994). Eberhart et al. (1996) estimate the maximum rate of annual increase for elk populations to be about 28%.

Females with calves isolate themselves from the herd for the first 2-3 weeks following birth and then return to the herd. One adult female will keep watch over several young while their mothers forage. Adult bulls usually segregate themselves from cows and calves during spring and summer; and young bulls may form small bands or remain with cows and calves until the rut, when older bulls force them to separate from the cows. During winter, elk form large mixed herds (typically led by old cows) on favored winter range (Fitzgerald et al. 1994). Elk are not territorial.

Elk populations respond to forage quality and quantity (Peek 1982; Coughenour and Singer 1996); and survival of elk calves decreases as population density increases (Singer et al. 1997). Singer et al. (1997) documented both increased winter starvation and summer predation of elk calves in response to increased population size. Population regulation in elk appears to be both a food mediated, and density dependent phenomenon. When wolves were added (by natural recolonization) to a

multi-predator (mountain lion, grizzly bear, black bear, coyote, and human) and multi-prey (elk, white-tailed deer, and moose) ecosystem in northwestern Montana, declines in elk populations were documented (Kunkel and Pletscher 1999). While Kunkel and Pletscher (1999) theorized that predation would limit elk populations at low density equilibrium levels, they presented no data relating elk populations to carrying capacity of the habitat. Elk population limitation (i.e. below carrying capacity of the habitat) resulting from predation has not been clearly demonstrated, but the recent restoration of wolves to the Greater Yellowstone Ecosystem will provide an excellent opportunity for further examination of the effects of predation (Singer et al. 1997).

LaCava and Hughes (1984) calculated minimum viable population size for short-term conservation of elk to be 214 animals within discrete populations sharing the same range, where the bull:cow ratio was assumed to be 7:100. They cautioned that long-term conservation assurance would require populations about 10 times larger, citing Franklin (1980) as their source for this factor. A bull:cow ratio of 7:100 is indicative of a heavily hunted elk population. Using LaCava and Hughes' (1984) formulas, the minimum viable population size for short term conservation of an unhunted population with a bull:cow ratio of 40:100 would be 63 animals. LaCava and Hughes (1984) further caution that, because these are estimates, managers should strive to maintain higher population sizes as a precautionary measure. To be most conservative, we favor a long-term conservation approach and advocate increasing these estimates by a factor of 10. Thus, we propose that discrete hunted populations of elk contain at least 2000 animals and that unhunted populations contain at least 600 animals. Assuming an average density of 1 elk/km² (2.6 elk/mi²), minimum patch sizes of suitable elk habitat should be 600 km² (232 mi²) and 2000 km² (772 mi²) for unhunted and hunted populations, respectively.

Population Status. Elk were reduced to very low numbers, or possibly extirpated from New Mexico by the early 1900s (Bailey 1931; Findley et al. 1975). Reintroduction programs were begun in 1911 with the release of 62 animals from Colorado and Yellowstone National Park (Findley et al. 1975). Statewide elk population estimates were 60 in 1912, 750 in 1923, 4000 in 1934, and 11,000 in 1967 when most of the former elk range in the state was reoccupied. Elk populations have continued to expand in both numbers and distribution with the current population size estimated to be between 61,900 and 77,500 animals (New Mexico State Game Commission 2001).

MOVEMENTS

Dispersal. Expansion of elk herds into suitable habitats and the movement of individual elk among distinct populations has been documented (Adams 1982). Van Dyke et al. (1998) theorized that dispersal of elk to new ranges likely occurs in response to population growth as the density of the population approaches or exceeds the carrying capacity of the former range. Except for range expansion, the importance and effect of emigrations on the receiving populations have not been elucidated.

Migration. Elk populations exhibit a variety of migration patterns, including migrating from winter range to a transitional (spring-fall) range and then to summer range and vice versa; migrating directly from winter to summer range and vice versa; and remaining on the same range (usually the winter range) year round (Adams 1982; Peek 1982; Bear 1989; Fitzgerald et al. 1994). Elk migrate in response to snow depth and leave high-elevation summer ranges when snow depth exceeds 40-50 cm (16-20 in) (Skovlin 1982; Fitzgerald et al. 1994). They tend to follow the snow line when migrating down slope during the onset of winter and the regrowth of green vegetation when migrating up slope in the spring (Verts and Carraway 1998). Distances covered by migrating elk herds vary greatly (2.4-129 km or 1.5-80 mi) depending on local conditions and availability of suitable habitat (Skovlin 1982). South-facing and grassy wind-blown slopes may be used by elk in winter in lieu of down-slope migration Peek (1982). Some members of mostly migratory populations do not migrate with the main population (Adams 1982; Peek 1982).

Barriers to Movement. Snow depths greater than 76 cm (30 in) substantially impede the movement of elk (Skovlin 1982). Human settlements have caused elk to alter traditional migration routes (Skovlin 1982). When traveling, elk often cross small openings (<150 m or 164 yards) but skirt the edges of larger openings (Winn 1976, cited by Skovlin 1982). Elk are often hit by vehicles on highways, but this apparently does not deter remaining elk in the herd from crossing.

Use of Linkages. Elk tend to use forested travel lanes adjacent to open meadows to move through semi-open habitat complexes. Riparian areas and drainage bottoms also appear to be preferred travel routes for elk (Skovlin 1982). Elk often travel parallel and below exposed ridgelines on wooded north-facing slopes. Travel routes between drainage basins often pass through low saddles (Skovlin 1982). Use of travel linkages is likely enhanced by the availability of forage. The value of linkages as elk habitat is potentially maximized if they are at least 366 m (400 yards) wide (Thomas et al. 1988).

ECOLOGY

Interspecies Interactions. In most areas elk coexist with other large herbivores, both wild and domestic. Forage consumption by one species may reduce the amount of forage that potentially could have been consumed by another species, an interaction referred to as “exploitive competition” (Nelson 1982). Competition cannot occur unless two or more species share the same range. Theoretically, such competition would limit populations of competing species where significant dietary overlap occurs and shared food resources are not overly abundant (Nelson 1982). Other herbivores in the NM Highlands planning area that may compete with elk for food include mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*), pronghorn (*Antilocapra americana*), bighorn sheep (*Ovis canadensis*), and domestic cattle and sheep. Singer and Norland (1994) compared the diets of elk, bison (*Bison bison*), bighorn sheep, mule deer, and pronghorn on the northern range of Yellowstone national park and found that the diet of elk significantly overlapped only with bighorn sheep. Another type of interaction that can affect ungulate species access to food resources is social tolerance or intolerance between or among species (Nelson 1982).

Both elk and cattle eat grasses, sedges, forbs, and browse on both summer and winter ranges (Nelson 1982; Peek 1982; Fitzgerald et al. 1994; Verts and Carraway 1998). However, the greatest potential for competition probably occurs between elk and cattle on foothill ranges (primarily winter-spring range for elk), with grass being the major shared food resource (Nelson 1982; Peek 1982; Fitzgerald et al. 1994). While grazing of elk and cattle may be temporally separated, competition for available food resources, nevertheless, occurs. If cattle graze on elk winter range from spring through autumn, little grass may remain for elk in the winter, and elk winter ranges can be significantly degraded (Nelson 1982). The level of competition between elk and cattle is generally not as serious on summer range, which is usually more abundant and diverse (Nelson 1982). However, overstocking on summer ranges and stocking of cattle in elk calving areas can adversely affect elk populations (Nelson 1982). The preponderance of reported studies suggest that elk are generally intolerant of and tend to avoid cattle, except on limited winter ranges where they may have no viable options (Nelson 1982; Wallace and Krausman 1987; Christensen et al. 1993). The combined effects of food competition and avoidance of cattle by elk on spring, summer, and fall ranges would further adversely affect elk populations. On the other hand, Hobbs et al. (1996b) found that use of the same range by both elk and cattle reduced cattle production (they did not measure elk production), especially below some threshold

of available forage biomass, which serves to corroborate the existence of competition between the species.

Considerable overlap occurs in the diets of elk and domestic sheep, and elk apparently abandon areas being grazed by domestic sheep (Nelson 1982; Peek 1982; Fitzgerald et al. 1994). Competition between elk and domestic sheep occurs primarily on high-elevation summer ranges (Nelson 1982). The additional disturbance of herdsman, dogs, horses, and camps for tending sheep may further displace elk (Nelson 1982). Elk prefer to remain at least 0.8 km (0.5 mi) from bands of domestic sheep (Nelson 1982).

Apparently, differences in primary food and habitat preferences between elk and deer (mule and white-tailed) limit competition between these species to relatively insignificant levels, except perhaps where winter range is severely limited (Nelson 1982). However, Johnson et al. (2000) found that mule deer avoided areas used by elk. Dietary preferences of elk and bighorn sheep substantially overlap and competition for limited food resources may occur on shared winter ranges (Nelson 1982). On summer ranges, bighorn sheep tend to avoid areas being used by elk (Fitzgerald et al. 1994). Thus, bighorn sheep populations may be adversely affected by the presence of elk; although, significant population effects have not been demonstrated.

Ecological Effects. Large herbivores are capable of substantially modifying vegetation distribution, abundance, and productivity; and, thus, are similarly capable of modifying ecosystems (Hobbs 1996; Riggs et al. 2000). Changes in the vegetative components of ecosystems will almost certainly, but not necessarily predictably, cause changes in faunal components as well (Truett 1996; Rambo and Faeth 1999). Excessive browsing and bark feeding by elk can detrimentally affect survival, and regeneration of aspens (*Populus tremuloides*) and willows (*Salix* spp.) (Basile 1979; Fitzgerald et al. 1994; Baker et al. 1997; Singer et al. 1998; Suzuki et al. 1999; Maschinski 2001; Peinetti et al. 2001). Herbivory by elk raises nitrogen concentrations and increases digestibility of various forage plants and enhances the rate of nitrogen cycling between plants and herbivores (Coughenour 1991; Hobbs et al. 1996a; Singer and Harter 1996; Peinetti et al. 2001). A study in Yellowstone National Park found that aboveground productivity of grazed vegetation was 47% higher than that of ungrazed vegetation (Frank and McNaughton 1993). Moser and Witmer (2000) found that foraging by elk and cattle reduced shrub and small mammal biological diversity. Coughenour (1991) concluded that elk compete directly with decomposers for plant biomass. Riggs et al. (2000) found that herbivory by elk, mule deer, and

cattle dramatically reduced the biomass of shrubs in forest understories, favoring an understory of grasses and forbs.

Elk are an important source of food for many predators. The presence of elk in multi-ungulate-prey systems may result in complex interrelationships among ungulate species (Kunkel and Pletscher 1999). For example, in northwest Montana, wolves prefer elk over deer as prey but will readily switch to deer when elk populations decline. Mountain lions also exert constant predation pressure on both elk and deer. Deer have a higher reproductive potential than elk and can withstand greater predation pressure. This serves to maintain high populations of wolves, which tend to further suppress elk populations. Kunkel and Pletscher (1999) hypothesize that where deer occur at high densities compared to elk, and both wolves and cougars are present, deer populations will cycle and elk populations will remain suppressed at a low density equilibrium (i.e., below carrying capacity based on primary production).

Dead elk are fed on by many scavengers, which enhances biological diversity in ecosystems. For example, elk carrion was found to be an important food source for wintering bald eagles (*Haliaeetus leucocephalus*) in northern Arizona (Grubb and Lopez 2000).

Launchbaugh and Urness (1992) speculated that elk, by eating mushrooms, may serve as symbiotic vectors of fungal spores to initiate important mycorrhizal associations with several conifer trees, such as ponderosa pine (*Pinus ponderosa*).

JUSTIFICATION AND FOCAL VALUE

Flagship. The elk is a majestic, charismatic animal with considerable public appeal among both hunters and nonhunters. Indeed, the elk is the focus of a major foundation—the Rocky Mountain Elk Foundation—and the symbol of a major civic organization—the Benevolent and Protective Order of the Elks (BPOE). In 1867, founders of the BPOE chose the elk as their symbol because they desired a readily identifiable creature of stature, indigenous to America. The restoration of elk from their near extirpation in the early 1900s is a conservation success story. Conservation programs that benefit elk should readily capture the public's attention and interest.

Prey. Elk are important prey for gray wolf, mountain lion, black bear, and grizzly bear. As carrion, they support a wide range of scavengers and decomposers in the ecosystem.

Umbrella. Because elk range over large areas, migrate season-

ally, and use a variety of habitats, protection of habitat for a viable population of elk will potentially protect the habitats of many other species. Elk prefer ecotones, which tend to support greater biological diversity; thus, elk conservation will likely benefit many other species and key ecological processes. However, elk can adapt to a wide variety of environmental conditions, and thus may not function well as a true umbrella species for ecotone species. Other ecotone species may have more stringent environmental requirements than elk. Assumptions about the benefits of elk management for other species have not been studied extensively, and further research is warranted. Elk strongly avoid human disturbances and may function as an umbrella for a wide variety of other species that respond negatively to roads and other indices of human disturbance.

MANAGEMENT RECOMMENDATIONS

Preserve and Enhance Habitat. Available winter forage limits elk populations (NMDGF 2001). Probably the most critical management need for elk is the establishment, preservation, and enhancement of winter ranges with adequate forage. Domestic livestock management on critical elk winter ranges should support the goal of providing adequate winter forage for elk populations. This may require the establishment of some livestock-free winter ranges. Winter recreationists should be restricted to locations >650m (0.4 mi) from elk wintering areas and concentrated in small areas with abundant topographic relief. Drainages adjacent to winter recreation areas should be designated as security areas for elk (Cassirer et al. 1992).

Protected calving areas are also critically important for the persistence and viability of elk populations. Roads should be closed and activities known to disturb elk (e.g., logging, mining, livestock grazing, off-trail hiking) should be prohibited in critical calving areas from May 1 to June 30.

Timber harvests should be planned and scheduled to ensure a continuum of adequate amounts of elk thermal and hiding cover and foraging areas for elk (Canfield et al. 1986).

Management or resource extraction activities should, to the extent possible, be conducted on seasonal ranges when elk are not present.

Establish Refugia. Elk behavior and use of the landscape are abnormal and restricted in the vicinity of roads, human activities, and domestic livestock. Population dynamics, especially adult sex ratios and age structure are significantly altered by

human hunting. A network of protected refuges is necessary to allow elk populations the opportunity to be influenced by natural ecological and evolutionary processes, or, in other words, to preserve their “naturalness.” Each refuge area should be at least 1000 km² (386 mi²) in size and linked by suitable habitat to allow for interchange of elk among these natural populations. For migratory elk herds, refugia should include at least 600 km² (232 mi²) each of summer and winter range and a protected migration route between seasonal ranges. Also, critical calving areas for the population should be within the refuge boundary.

Restore Native Large Predators. Introduction and proper management of extirpated predators such as the wolf and grizzly bear will likely benefit elk populations by checking population growth, improving overall population health, and creating a more natural ecological/wilderness condition (Peek 1982). Modern ecological theory and empirical evidence strongly suggest that top predators exert a controlling influence over entire ecosystems through a cascade of regulatory effects across the various trophic levels of the ecosystem—the so called “top-down effect” (Palomares et al. 1995; Terborgh et al. 1999; Miller et al. 2001). A major effect of top predators is the limitation and stabilization of populations of large herbivores such as elk (Kunkel and Pletscher 1999). This, in turn, prevents overutilization of primary producers (primarily plants) and may promote a greater diversity of primary consumers (e.g., insects and small mammals), smaller predators, and omnivores. Generally, biological diversity is positively correlated to resilience stability (i.e., ability to rapidly recover from disturbance) of ecosystems (Odum 1993).

Promote and Restore Natural Disturbance Regimes. The restoration of natural fire regimes, especially in wilderness areas, will restore natural dynamic process and a mosaic of vegetative communities that benefit elk (Peek 1982). Prescribed burning is a useful management tool for improving elk habitat.

Road Closures. Road traffic and various outdoor recreational activities (such as hunting, cross-country skiing, and off-road vehicle use) have been shown to adversely affect habitat suitability for elk, except at very low road densities. Human disturbances in wintering areas cause increased energy demands and in calving areas cause declines in reproductive success. Road closures may minimize dislocation of elk from prime habitat, enhance habitat effectiveness and suitability, and increase elk survival. In many areas, closures may only need to be implemented seasonally or road use may need to be limited to only light (<4 trips/day) administrative use. Managers should strive for road densities of less than 0.6 km/km² (1

mi/mi²) and, where possible, close roads and recreational trails that traverse critical habitats, such as calving and wintering areas during seasonal use by elk.

Monitoring. Monitoring is essential to determine the status and condition of herds and to establish management objectives and sustainable hunt quotas. Changes in range use by elk populations may occur in response to changes in population size or alterations of the habitat (van Dyke et al. 1998). They recommend systematic reevaluation of important elk population range boundaries at least every 10 years.

Multi-jurisdictional Planning. Conservation planning for large, migratory ungulates must be conducted over large spatial scales and must consider landscape connectivity. Land areas large enough to support a viable elk population will likely encompass a multitude of jurisdictions. Establishment of regional planning authorities through appropriate means (e.g., legislative or administrative) should be encouraged and pursued.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of elk ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about elk to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

REVIEWERS

Michael Wisdom, University of Idaho
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Beaver (*Castor canadensis*)

*Please see accompanying Excel chart of Beaver Habitat Characteristics in the folder titled "Focal Species Habitat Characteristics Charts" on the compact disk.

INTRODUCTION

The beaver (*Castor canadensis*) is the largest native rodent (order Rodentia) and only representative of its family (Castoridae) and genus (*Castor*) in New Mexico and North America. It is a riparian-obligate species that is widely distributed in low-gradient streams and rivers and lakes throughout the NM Highlands planning area. Ecologically, beavers are herbivores and potential prey for large and medium sized predators; but, most importantly, the beaver is a keystone species that significantly increases biological diversity and ecological productivity and stability through its activities. The beaver is a protected furbearer in New Mexico and may be hunted or trapped in unlimited numbers in many areas within the NM Highlands planning area. They are common in the NM Highlands planning area, but the potential exists for expansion of populations. The beaver's popularity with the general public and nature enthusiasts fosters considerable public interest and support for nature conservation.

DISTRIBUTION

Historic. Beavers historically occurred in streams, ponds, and lake shores throughout North America, except in the arctic tundra, Florida peninsula, and southwestern deserts (Jenkins and Busher 1979). Populations were extirpated or reduced to near extinction throughout most of the beaver's historic range by the early 1900s, but reintroduction programs have successfully restored beavers to many parts of their former range (Jenkins and Busher 1979; Naiman et al. 1988). However, present populations represent a small fraction of historical numbers of beavers (Naiman et al. 1988). The situation was similar in New Mexico where beavers were historically abundant in most watersheds, but were eliminated by trapping from many streams and rivers and reduced to extremely low numbers elsewhere by the end of the 19th century (Bailey 1931). The New Mexico Department of Game and Fish actively restocked beavers throughout the state between 1947 and 1958 (NMDGF 2001).

Current. Beavers are distributed throughout much of their former range in New Mexico, but most are found in the northern mountains (San Juan, Jemez, and Sangre de Cristo)

(NMDGF 2001). They also occur in the Mogollon Mountains, and along the Canadian, Gila, Pecos, and Rio Grande river drainages (NMDGF 2001).

Potential. Considerable potential to restore and expand beaver populations exists within the NM Highlands planning area.

HABITAT

General. The following attributes contribute to habitat suitability for beavers: (1) stable aquatic systems with adequate, permanent water; (2) channel gradients less than 15%; (3) wide valley floors, and (4) adequate supplies of quality food resources (Williams 1965, cited in Allen 1983; Suzuki and McComb 1998). Beavers are stimulated to build dams by the sound of running water (Jenkins and Busher 1979). Suitability of beaver habitat may be reduced by nearby railways, roads, and land clearing activities (Slough and Sadleir 1977, cited in Allen 1983). Streams and lakes exhibiting extreme annual fluctuation in flow volumes or water levels have little value as beaver habitat (Allen 1983).

Preferred. The following information is from Allen (1983), unless otherwise cited. Beavers require a permanent supply of water and stable water levels, which beavers can control on small streams, ponds, and lakes by constructing dams. Larger rivers and lakes where water level control is beyond the capability of beavers are usually unsuitable, as are swift streams that lack suitable dwelling sites during periods of high and low water.

In lotic systems, stream gradient is the most significant factor affecting habitat suitability. In Colorado, most beaver colonies occupied stream valleys with less than 6% gradient; 90% of all colonies observed were in reaches with less than 12% gradient; and no colonies were found in streams with gradients that equaled or exceeded 15% (Retzer et al. 1956, cited in Allen 1983). However, only streams with valleys that are wider than the stream channel provide suitable habitat for beavers. Valley widths that equal or exceed 46 m (50 yards) are considered most suitable. Flat flood plains allow for the construction of extensive canal systems which aid beavers in accessing and transporting food. Marshes, ponds, and lakes associated with adequate food resources also provide suitable habitat for beavers.

Food availability is an important determinant of habitat suitability, especially palatable woody plants that can be cached for winter consumption (Allen 1983; Jenkins and Busher 1979). Food preferences are discussed in more detail below.

Special Features. Beavers construct lodges or burrows which provide secure cover for escape from predators, resting, thermal regulation, and reproduction (Jenkins and Busher 1979). Lodges may be constructed against a shoreline (sometimes over a bank burrow) or in open water, and entrances are below the water line (Allen 1983). Beavers often modify their habitats by constructing dams and extensive canal systems. Generally, dam building is limited to 1st to 4th order streams that are 15 m (50 ft) or less wide and have gradients of 4% or less (Gurnell 1998). Hydrogeomorphological effects of dam building by beavers include: stream flow stabilization, attenuation of peak flood flows; raised water tables; increased hydrologic complexity in riparian areas; increased stream channel complexity; increased sediment storage and decreased sediment yields by streams; sorting of bed sediments creating greater benthic substrate diversity; increased decomposition of organic matter and release of nutrients within the stream ecosystem; and, generally, increased lotic and riparian habitat diversity and stability (Gurnell 1998).

FOOD HABITS / FORAGING BEHAVIOR

Beavers appear to prefer herbaceous vegetation (such as forbs, grasses, and aquatic vegetation, including their roots and tubers) over woody plants when it is available (Allen 1983; Howard and Larson 1985). They eat the bark, buds, leaves, and twigs of a wide variety of woody plants and may show strong local preferences for particular plant species (Fitzgerald et al. 1994). Commonly preferred woody species include quaking aspen (*Populus tremuloides*), willows (*Salix* spp.), cottonwoods (*Populus* spp.), and alders (*Alnus* spp.) (Allen 1983). These species sprout vigorously after fires (NMDGF 2001). Both terrestrial and aquatic herbaceous plants are eaten during twilight and darkness in spring and summer (Fitzgerald et al. 1994). Beavers are coprophagous and re-ingest their feces to achieve more complete digestion of foods consumed (Fitzgerald et al. 1994). Most feeding by beavers occurs within 30 m (100 ft) of water, but foraging can extend out to about 100 m (328 ft) (Allen 1983; Fitzgerald et al. 1994). Harvested food is typically cached and consumed under the ice near lodges or burrows in winter (Fitzgerald et al. 1994).

Beavers cut trees year round, but tree cutting peaks in late fall, when herbaceous vegetation is less available and food is being stored under water for winter, and again in early spring (Allen 1983).

POPULATION DYNAMICS

Life History. The beaver is the largest rodent in North America and weighs 15-30 kg (33-66 lbs), with males being slightly larger than females (Jenkins and Busher 1979; Fitzgerald et al. 1994). These semi-aquatic mammals build dams, lodges, and canal systems to create a secure, aquatic environment which provides shelter and a system for floating and storing food and moving construction materials (Fitzgerald et al. 1994). Beavers are social animals and live in colonies of 4-8 animals comprised of an adult pair, yearling offspring, and young of the year (Jenkins and Busher 1979; Fitzgerald et al. 1994). Beavers are monogamous, become sexually mature at 1.5 years of age but first reproduce at about 3 years of age, mate in late winter, and bear average litters of 3-4 kits (Jenkins and Busher 1979). Beavers may live up to 21 years, but most do not live beyond 10 years in the wild (Jenkins and Busher 1979).

Population Density. Typical population densities for beaver are 0.4-0.8/km² (1-2/mi²) (Jenkins and Busher 1979).

Home Range. In suitable habitats, beaver colonies tend to be spaced about 1 km (0.6 mi) apart along streams, and colonies defend a territory of about 2-3 ha (5-7.5 acres) (Fitzgerald et al. 1994).

Causes of Death. Predators of beavers include wolf (*Canis lupus*), coyote (*Canis latrans*), bears (*Ursus* spp.), river otter (*Lutra canadensis*), mink (*Mustela vison*), lynx (*Lynx canadensis*), bobcat (*Lynx rufus*), and mountain lion (*Puma concolor*) (Jenkins and Busher 1979; Fitzgerald et al. 1994). Other causes of death include hunting and trapping by humans, starvation, drowning during floods, and epizootics of tularemia (Fitzgerald et al. 1994). Average annual mortality is about 30% (Jenkins and Busher 1979; Fitzgerald et al. 1994).

Population Structure and Viability. Beaver populations consist of nonoverlapping colonies, each defending their territory (Allen 1983). Typically, a colonized area consists of a series of ponds of varying ages, sizes, and depths (Rutherford 1964, cited by Allen 1983). Colony density increases with watershed size (Howard and Larson 1985). Fryxell (2001) demonstrated that a regional population comprised of several individual beaver colonies is similar in many respects to a metapopulation, where individual colonies are analogous to subpopulations within a metapopulation structure. Colonies occupying the most suitable habitats are more reproductively successful and provide a source of dispersing beavers. Dispersers periodically recolonize vacant habitats where former resident colonies have gone extinct due to negative net production over time or vari-

ous stochastic events. Thus, while many individual colonies are randomly becoming extirpated and subsequently recolonized, other colonies tend to exhibit long-term persistence, and relative stability is maintained throughout the larger population (Fryxell 2001). Estimates of minimum viable population sizes for beavers are not available.

Population Status. The beaver is a protected furbearer in New Mexico and may be hunted or trapped in unlimited numbers from October 15 through April 30, except in the Valles Caldera National Preserve, Rio Grande Wild and Scenic River, Orilla Verde and Santa Cruz Lake National Recreation Sites, Valle Vidal/Greenwood area, state parks, national parks and monuments, national wildlife refuges, state wildlife management areas, most of Los Alamos County, and the following national forests: Gila, Cibola, Lincoln, and Apache (NMDGF 2002). During the 1999-2000 season, 91 beavers were reported to have been harvested in New Mexico, with half the reported take coming from San Juan County (NMDGF 2000).

MOVEMENTS

Dispersal. Typically, all beavers (most at two years of age) disperse from their natal colonies in April and May, but it may take several months for dispersers to ultimately settle in suitable habitats (Fitzgerald et al. 1994; Sun et al. 2000). In New York, the predominant direction of dispersal was downstream and the mean dispersal distance for females (10 km [6.2 mi]) was about three times that of males (Sun et al. 2000). Sun et al. (2000) found that most dispersers moved only short distances and 73% colonized the closest neighboring suitable site. Fitzgerald et al. (1994) noted an average dispersal distance is about 7 km (4.3 mi) and that most dispersal movements are less than 16 km (10 mi).

Migration. Beavers are nonmigratory. However, entire beaver colonies may move to new stream sections when local food supplies become scarce (Fitzgerald et al. 1994).

Barriers to Movement. No discussion of barriers to the movement of beavers was found in the literature.

Use of Linkages. Most authors suggest that beavers usually follow watercourses when dispersing.

ECOLOGY

Interspecies Interactions. Beaver ponds may be beneficial or harmful to various species of fish, and they provide or enhance habitats for waterfowl and amphibians (Jenkins and Busher 1979; Fitzgerald et al. 1994). Habitat mosaics created by beavers increase arthropod diversity, which may benefit insectivorous birds and mammals (Martinsen et al. 1998). Beavers influence plant and animal community composition and diversity both in the vicinity of their activities and downstream (Naiman et al. 1988). Water management by beavers aids recovery of overgrazed, eroded stream banks, which improves habitats for many species (Jenkins and Busher 1979; Albert and Trimble 2000). Beavers are prey for various large and medium sized predators (see above).

Beavers compete for food resources with elk and cattle, but the effects of these interactions have not been extensively researched and reported in the literature. Obviously, riparian areas where livestock or elk browsing has significantly eliminated woody riparian vegetation and prohibits its recovery have little habitat value for beavers.

Ecological Effects. Beavers are classic “keystone” species in that their activities significantly affect ecosystem function to a degree that is disproportionate to their numerical abundance (Miller et al. 1999). These effects stem from the beaver’s ability to physically alter aquatic systems and from their concentrated herbivory near water bodies (Jenkins and Busher 1979; Naiman et al. 1988; Gurnell 1998). Naiman et al. (1988) and Gurnell (1998) identified the following ecological effects: stabilization of stream flows; increased wetted surface area (i.e. benthic habitat); elevation of water tables causing changes in floodplain plant communities; creation of forest openings; creation of conditions favoring wildlife that depend upon ponds, pond edges, dead trees, or other new habitats created by beavers; enhancement or degradation of conditions for various species of fish; replacement of lotic invertebrate taxa (e.g., shredders and scrapers) by lentic forms (e.g., collectors and predators); increased invertebrate biomass; increased plankton productivity; reduced stream turbidity; increased nutrient availability; increased carbon turnover time; increased nitrogen fixation by microbes; increased aerobic respiration; increased methane production; reduced spring and summer oxygen levels in beaver ponds; and increased ecosystem resistance to perturbations. Beaver ponds undergo predictable succession over long time periods from open water ponds to marshes to seasonally flooded meadows (Naiman et al. 1988). Many mountain meadows in the NM Highlands area are the result of past beaver activities (NMDGF 2001). Removal of beavers from an area

can cause their dams to deteriorate, increasing runoff and gully formation, lowering water tables, and reducing biological diversity (NMDGF 2001). Beavers are being used as ecological restoration agents in degraded riparian ecosystems on the Zuni Indian Reservation in east-central New Mexico and elsewhere (Albert and Trimble (2000).

Martinsen et al. (1998) discovered a most unusual relationship between beavers and beetles. Resprout growth following the cutting of cottonwood trees by beavers contained twice the level of defensive (against mammalian herbivory) chemicals as normal juvenile cottonwoods. Leaf beetles fed preferentially on cottonwood resprout growth, thus bioaccumulating the protective chemicals (phenolic glycosides) which they used for their own defense.

JUSTIFICATION AND FOCAL VALUE

Habitat Quality Indicator. Beavers prefer aspens, willows, cottonwoods and alders, which occur in high-quality riparian ecosystems (Allen 1983). Degraded riparian ecosystems, usually the result of overgrazing by domestic ungulates, are generally unsuitable for beavers. Beavers enhance the quality of habitats they occupy for themselves and many other species (Naiman et al. 1988).

Keystone. The beaver is a keystone riparian species because its activities substantially alter landscapes and create new ecosystems (Collen and Gibson 2001). Ecosystem productivity and biological diversity are also enhanced by the activities of beavers (Naiman et al. 1988).

Prey. Beaver are prey for gray wolf, mountain lion, black bear, grizzly bear, coyote, lynx, river otter, and mink (Jenkins and Busher 1979; Fitzgerald et al. 1994).

MANAGEMENT RECOMMENDATIONS

The recommendations of Bailey (1931: 219) remain valid and appropriate today:

On almost all the mountain streams they should be protected and encouraged. A series of beaver ponds and dams along the headwaters of a mountain stream would hold back large quantities of mountain water during the dangerous flood season and equalize the flow of the streams so that during the driest seasons the water supply would be greatly increased in the valleys. Beaver ponds not only hold water but distribute

it through the surrounding soil for long distances, acting as enormous sponges as well as reservoirs. A series of ponds also increases the fishing capacity and furnishes a safe retreat for the smaller trout and protection from their enemies. In addition a protected beaver colony is one of the most interesting features of mountain or forest, as with protection the animals become less wary and more diurnal in their habits so that they can be readily observed and studied by those traveling and camping in wild regions.

Restore Beaver Colonies. Beavers should be restored to all drainages where they historically occurred, especially in headwater and low-order streams. Reintroductions should proceed immediately in stream reaches with suitable habitat.

Improve Potential Beaver Habitat. Trees and shrubs preferred by beavers for food and construction materials tend to sprout vigorously after fires. Restoration of natural fire regimes and prescribed natural fires should be incorporated into management recommendations for beavers. Domestic ungulates should be excluded from riparian areas or managed in ways that promote the recovery of woody riparian plants, especially aspen, willow, cottonwood, and alder.

Use Beavers as Agents for Restoring Streams and Watersheds. Beavers have been shown to be effective agents for restoring degraded streams and watersheds and should be used in this capacity wherever feasible. In critical reaches, managers should recognize the beaver's ability to improve its own habitat and consider supplemental feeding during the establishment period.

Close Additional Areas to Harvesting of Beavers. Legal beaver harvests in New Mexico are small and of little economic consequence compared to the potential ecological benefits and services provided by beavers. In addition to currently closed areas, beaver harvesting should be prohibited in the Santa Fe and Carson national forests and in all core and compatible use areas identified in this plan, except where significantly adverse economic impact may result from their activities.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of beaver ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about beavers to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

River Otter (*Lontra canadensis*)

Adapted from an unpublished report for the Wildlands Project by Paul J. Polechla Jr., A Review of the Natural History of the River Otter (*Lontra canadensis*) in the Southwestern United States with special reference to New Mexico.

*Please see accompanying Excel chart of River Otter Habitat Characteristics in the folder titled "Focal Species Habitat Characteristics Charts" on the compact disk.

HISTORY AND CURRENT STATUS

Otters (*Lontra canadensis*) were found in all major waterways of the United States and Canada until at least the 18th century, but they have been extirpated over large areas due to damming and settlement of rivers, draining of wetlands, hunting and trapping, siltation, pesticides and other forms of human encroachment (Toweill and Tabor 1982). Of the thirteen species of otters in the world, twelve are considered endangered and are listed on the CITES treaty on Appendix I (Foster-Turley et al. 1990). The Nearctic river otter (*Lontra canadensis*) is considered endangered in four (Colorado, Indiana, Kansas, and Nebraska) of the 49 states where it once occurred, and is still legally trapped in 28 states (Polechla in press). In New Mexico it is considered both extirpated and endangered, even though most of the state is unsurveyed and reports of otters continue to surface (Polechla 2000). Bailey (1931) claimed the subspecies occurred in three principal drainages in New Mexico: the Gila/San Francisco/San Juan/Colorado River, the Rio Chama/Pecos/Rio Grande, and the Canadian/Arkansas/Mississippi River drainage. The species saw a decline in New Mexico in the late 1890s to early 1900s due to the increase in human population, unregulated over-trapping, over-grazing, intensive irrigation, mining without reclamation, over-logging, and organic pollution.

The Southwestern river otter (or Sonoran river otter, as it is sometimes known) (*Lontra canadensis sonora*) is the most imperiled mammal (Polechla 2000; Melquist et al. in press) in the most endangered habitat in New Mexico and the southwestern United States: the riparian zone (Dick-Peddie 1993; Cole et al. 1996; Bogan et al. 1998). The subspecies was listed under the Natural Heritage Global Rank as "G5T1" (G5 = Demonstrably Secure; T1 = Very Rare) (Arizona Game and Fish Department 1995). No remnant wild population of southwestern river otters has been documented to date, though a number of reports from amateur naturalists and professional biologists are made annually. Very few of the tributaries of the Colorado River and Rio Grande have been surveyed and many river otters of other subspecies have been translocated from areas outside of these waters (Polechla 2000). Southward/east-

ward dispersal could have occurred from animals introduced in recent years in Colorado or Arizona (New Mexico Department of Game and Fish 1991).

ECOLOGY AND HABITAT

River otters occur in wetlands in every major biome including grasslands, temperate deciduous forest, temperate rain forest, semi-tropical forests, boreal forests, and deserts. River otters occupied virtually every estuarine, riverine, lacustrine, and palustrine aquatic habitats (Polechla In prep.). The essential aspect of river otter habitat is the presence of wetlands, the most imperiled habitat in the Southwest. Native fish fauna also constitute an endangered component of the region.

About 17 different semi-aquatic situations have been cited as being used by otters in the Southwest. The bodies of water inhabited by otters range from man-made ones (e.g. stock and hatchery ponds, reservoirs, spillways, and borrow ditches) to permanent and intermittent bodies of water that are either lotic (flowing) or lentic (pooled). Minimum estimated flows are 10 cubic feet (.28 m³) per second (Fitzgerald et al. 1994). Otters also require heavy bank cover (rocks, vegetation and debris piles), and haul-out sites suitable for leaving and entering the water. Otters take advantage of available natural crevices for shelter and for their natal dens, occasionally modifying it. Wild and reintroduced otters are known to use rock cavities, log jams, hollow logs, and burrows of other animals (e.g., muskrat and beaver) (Boone 1983; Foy 1984; Reeves 1988).

Several biologists have correlated otter presence with certain environmental parameters. In a translocation study in Colorado, Malville (1986) found that "otters were selecting pools with higher densities of beaver den entrances and high numbers of exposed boulders." Translocated otters in Arizona preferred areas with high canopy cover that shaded naturally created crevices and cavities (such as rock piles and old beaver and muskrat dens) that are close to deep pools of water (Christensen 1984). In northern Utah, Bich (1988) identified habitats (such as mountain valleys, beaver ponds, and wetlands) with abundant vegetative escape cover adjacent to streams with a low gradient as favorable otter habitat. The commonality among these studies is that otters inhabit areas with ample cover and water volume, and the presence of beaver. In New Mexico, otters have been documented occurring in permanent streams, beaver ponds, the confluence of two streams, large coldwater reservoirs, and prominent small reservoirs (Polechla 2000).

Otters require an abundant food base of fish and crustaceans, supplemented with semi-aquatic amphibians, insects, birds and small mammals. Information on specific prey species in New Mexico is not available, however it is speculated that all of the 69 native fish, and 46 introduced fish, as well as one salamander, crayfish, 12 species of frogs, 13 species of toads, 9 species of turtles, and some lizards, snakes and birds could be regarded as potential prey. In the past, the otter was condemned as a menace to trout fisheries, but studies have demonstrated that the bulk of its diet is made up of small nongame fish (Greer 1955).

POPULATION STRUCTURE AND VIABILITY

Mean home range estimates vary from 1 to 78 km linear distance along rivers and from 9 to 57 square km in marshes (Melquist and Dronkert 1987). After a period of acclimation, translocated otters in the arid climate of Arizona had a 14.4 to 15.7 km linear range (Christensen 1984). Reintroduced otters in Colorado settled down to a mean home range of 32 km (ranging from 5 to 71 km) (Mack 1985). Density estimates vary from one otter per 3.9 km waterway to one otter per 17 km waterway along rivers (Melquist and Dronkert 1987). Otters translocated in the mountains of Colorado occurred at a density of one otter per 10.2 km (Malville 1986).

Very little direct evidence of mating or reproduction has been recorded in the Southwest. The ultimate test of whether or not remnant populations can hold on or expand or if translocated populations can become established is if otters can reproduce and recruit their offspring into the next reproducing generation.

MOVEMENTS

Arid areas act as barriers to otter dispersal, whereas drainage systems act as primary avenues of dispersal and corridors of genetic flow (Pohle 1920; Van Zyll de Jong 1972). Overland dispersal is dependant on distance between the divide between two drainages and the nature of the intervening country (Van Zyll de Jong 1972). Overland dispersal across mesic areas of mountain passes is more likely than dispersal across arid deserts. Otters are so semi-aquatic that they physiologically cannot adapt to waterless conditions, making some arid areas true barriers (Reed-Smith 2001). In New Mexico, natural barriers to otter dispersal include the Llano Estacado, Jornada del Muerto, San Simeon Valley, Animas Valley, Playas Valley, Hachita Valley, and the Plains of San Agustin.

Little information is available on movement of otters in the Southwest, but in other parts of North America, river otters travel distances of up to 42 km per day for dispersing animals in established populations (Melquist and Hornocker 1983). Most daily movements of single animals and family groups are less than 10 km per day. This varies with season and they move less during the winter (Melquist et al. In press). The average daily distance moved by translocated otters on the Verde River in Arizona was only 0.63 km (Christensen 1984). Post-release movement averaged 12.8 km (range 1-47 km) for translocated otters in Rocky Mountain National Park in Colorado (Mack 1985).

THREATS AND SENSITIVITIES

Although the most important predators of river otters are humans, they are also preyed upon by bald eagle, gray wolf, coyote, domestic dog, red fox, bobcat, mountain lion and black bear (Frey and Yates 1996; Melquist et al. In press). Dogs may pose a serious threat to otter populations in developing or heavily used recreational areas (Melquist and Hornocker 1983). Habitat alteration (e.g., adverse modification of bank configurations along rivers, destruction of riparian vegetation, and draining of wetlands), indiscriminant shooting, overtrapping of beaver, overgrazing, lack of minimum stream flow, erosion and subsequent siltation, decline of fish population, possible contamination of prey species due to pesticides and heavy metals, and other types of human disturbances contributed to otter decline and ultimate extirpation (Jones and Schmitt 1997). Most mortality is thought to occur from trapping (Hoffmeister 1986) and road kills (Melquist and Hornocker 1983). Roads are deathtraps for otters; they do not like to go under bridges and will instead climb over the road to avoid the bridge, which results in numerous roadkills. They also tend to walk along roads and be killed.

Habitat destruction and water pollution, sometimes due to cattle grazing, have an impact as well (Christensen 1984; Fitzgerald et al. 1994). Surface mining and oil and gas development may also seriously impact otter population by affecting water quality, habitat and prey availability (Rudd et al. 1986). Residues of pesticides, including mercury, DDT and Mirex have all been reported from otter tissues (Toweill and Tabor 1982). River otters are susceptible to bioaccumulation of persistent toxins such as industrial pollutants, heavy metals, and chlorinated hydrocarbons (Melquist and Kronkert 1987; Ben-David et al. 2001). Otters are also at risk from human recreation activities, including power-boating, disturbance, and fisheries management practices (Waller et al. 1999).

JUSTIFICATION AND FOCAL VALUE

Umbrella. : Although not a classic umbrella species, the otter may function as one for long stretches of river or stream. If a river supports otters, it is likely to support other species dependent on healthy river ecosystems as well. The river otter has large area requirements (long stream stretches) and a defined habitat association (streams and rivers). The protection of otters in New Mexico requires the protection of watersheds and maintenance of water quality, which in turn require ecologically sound forestry and cattle ranching practices. Therefore, protection of the otter will protect a large number of species.

MANAGEMENT RECOMMENDATIONS

Conduct comprehensive otter surveys. The New Mexico Game and Fish Commission declared the river otter to be extirpated in the state before any comprehensive search had been conducted. Polechla (2000) surveyed the upper Rio Grande in New Mexico obtaining 22 records, 17 historical and 5 new hearsay records. Otter surveys need to be done in Los Pinos River, Navajo River, San Juan River, Gila River, San Francisco River, Vermejo River, More River, Canadian River, Pecos River, and Rio Chama, since otters were known to occur there historically or are reported to be there now.

Reintroduce otters, if *sonora* subspecies extinct. If any members of the southwestern river otter are present, full protection under the Endangered Species Act should be proposed, with a recovery plan and critical habitat designated. If the *sonora* subspecies is extinct, otters should be reintroduced in suitable sections of the rivers where they were known to occur historically, and when study shows adequate habitat exists. Factors that may have caused the actual or near extirpation of the otter in New Mexico may be too far advanced to allow for the species' successful restoration (Hubbard et al. 1979). Before reintroduction is considered, comprehensive habitat studies should be conducted.

Target future reintroduction efforts to lower elevation river segments. Some of the best, unpopulated otter habitat is the low elevation reaches of large rivers. Once reintroduced at lower elevations, otters can rapidly expand upstream as conditions dictate. Some conservationists oppose river otter reintroduction to stream segments that are critical habitat for endangered native fish species. However, otters are much more likely to prey on abundant fish species which many times are competitors with the endangered species.

Aquatic management. Management recommendations for the upper Rio Grande include (Polechla 2000):

- Conserve existing native riparian vegetation. Review prescribed burn and let burn policies to avoid crown fires.
- Restrict cattle grazing in or near riparian areas.
- Stabilize eroding banks and arroyos.
- Undertake extensive cleanups of river basins polluted by past mineral extraction, intensive agriculture or other resource uses.
- Clean up mine tailings on the Red River.
- Restore water flow on the Rio Costilla from intermittent to permanent stream status.
- Increase public information and education efforts regarding otters and other wildlife.
- Institute a comprehensive beaver management program.
- Model possible interactions between Rio Grande Cutthroat trout, whirling disease and river otters.
- Minimize disturbance in shoreline habitats: Locate trails, roads, picnic areas, campgrounds and other recreational facilities away from riparian and shoreline habitats.

Research and Monitoring. If an adequate native population is found or otters are translocated into New Mexico, research is needed to determine their movements, survivability and biology.

- Continue to monitor water quality on the Red River with more sophisticated techniques.
- Institute an assessment of methyl mercury in fish in the Red River.
- Monitor recreational activities that degrade riparian forests and marshes, erode the banks, or reduce beaver and fish populations.
- Monitor the concentrations of contaminants in water and bioaccumulation of toxic compounds in fish, reintroduced river otter and other semi-aquatic mammals.

Lynx (*Lynx canadensis*)

*Please see accompanying Excel chart of Lynx Habitat Characteristics in the folder titled "Focal Species Habitat Characteristics Charts" on the compact disk.

INTRODUCTION

The lynx (*Lynx canadensis*) is one of three members of the cat family (Felidae) that are native to the southern Rocky Mountains. It requires large expanses of high-elevation boreal forest habitat and is highly dependent upon snowshoe hares for prey. Ideal habitat for lynx is old-growth forest subjected to natural disturbance regimes resulting in a dynamic heterogeneous mosaic of various forest successional stages. The lynx requires habitat protection and forested linkages among patches of suitable habitat for its survival and persistence. Ecologically, the lynx is an important predator of snowshoe hares and red squirrels. It is a good indicator of wilderness habitat quality within the southern Rocky Mountain ecoregion. Lynx are federally listed as a threatened species and are fully protected throughout the region.

DISTRIBUTION

Historic. Lynx are primarily associated with forested habitats in arctic and boreal regions of North America, but also occurred in the northern contiguous United States (Koehler and Aubry 1994; McKelvey et al. 1999a). See McKelvey et al. (1999a:244) for a map of documented lynx occurrences from 1842 to 1998 in the contiguous United States. The southernmost extension of their historic range included the southern Rocky Mountains, with documented occurrences as far south as the Colorado-New Mexico border (McKelvey et al. 1999a). There are no specimens from or verified records of lynx occurrence in New Mexico (McKelvey et al. 1999a; NMDGF 2001). McKelvey et al. (1999a:230-231) provides the following account of documented lynx records in Colorado:

A thorough review of the history of documented lynx records in Colorado was conducted by Halfpenny et al. (1989, unpublished) and, except for the discovery of several more historical specimen records, little new information has become available since their analysis. Unlike other western montane regions..., boreal forest habitat in Colorado is insular in nature and isolated from similar habitat in Utah and Wyoming by more than 150 km [93 mi] of lower elevation habitats in the Green River Valley and

Wyoming Basin (Findley and Anderson 1956). All but a few specimen records are from the center of this island of boreal forest habitat in west-central Colorado. There are four specimens from the late 1800s: one without a specific collecting locality, one from Cumbres County near the New Mexico border, one from Breckenridge, Summit County, and one from Colorado Springs, El Paso County. Halfpenny et al. (1989, unpublished) reported that Edwin Carter's taxidermy notes in the Denver Museum of Natural History included a lynx trapped in Soda Gulch, Clear Creek County in 1878. Museum specimens were also found from Grand Lake, Grand County in 1904-1905; Jefferson, Park County in 1912; and southwestern Gunnison County in 1925. Terrell (1971) reported one lynx trapped at Red Cliff, Eagle County in 1929 and one at Marble, Gunnison County in 1931. Through interviews with trappers, Halfpenny et al. concluded that reports of three lynx being trapped in Eagle County in 1930 and 1936 were reliable.

After 1936, no lynx specimens or reports of kills are known until 1969, when a specimen was trapped near Leadville, Lake County, and others were reportedly shot on the Frying Pan River, Pitkin County (Terrell 1971) and on the south side of Vail Mountain, Eagle County (Halfpenny et al. 1989, unpublished). In 1972, a lynx specimen was trapped on Guanella Pass, Clear Creek County and, in 1974, two were trapped (one is preserved as a specimen) on the north side of Vail Mountain, Eagle County. Since that time, only tracks have been found, including three sets on the Frying Pan River, Eagle and Pitkin Counties and five sets near Mt. Evans, Clear Creek County (Halfpenny et al. 1989, unpublished). There are no verified records of lynx in Colorado since 1974, despite large-scale snow-tracking efforts (Carney 1993, unpublished).

Seidel et al. (1998) concluded that no viable populations of lynx occurred in Colorado and that they were most likely extirpated from the state. Koehler and Aubry (1994) theorized that viable lynx populations may never have occurred in the fragmented boreal habitats of the southern Rocky Mountains. They based this hypothesis on the paucity of historical records and the long-distance dispersal capabilities of lynx, suggesting that lynx periodically emigrated to Colorado from more northern lynx populations when these populations were at high levels.

Current. In 1999 and 2000, 96 lynx were released on the San Juan and Rio Grande National Forests in southwestern Colorado. From 39 to 56 of these animals were alive as of March 2002; most were in or near the release area (CDOW 2002).

Potential. The Colorado Division of Wildlife plans to identify and map potentially suitable lynx habitat remaining in Colorado (Seidel et al. 1998).

HABITAT

General. The following information regarding lynx habitat is taken from Koehler and Aubry (1994), unless otherwise cited. Throughout their distribution, lynx occur in boreal forest vegetative communities. Suitable habitats in the southern Rocky Mountains are at high elevations and are characterized by Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), aspen (*Populus tremuloides*), and lodgepole pine (*Pinus contorta*). Typical lynx habitat is characterized by continuous forest stands of varying ages with low topographic relief. Late successional or old-growth forests provide denning sites and hiding cover for kittens. Early successional forests support high prey densities and provide important foraging areas for lynx. Snowshoe hare (*Lepus americanus*) presence and abundance is a major determinant of suitable lynx habitat. Intermediate successional forests provide cover for travel by lynx and provide important connectivity among favorable patches of denning and foraging habitat. Frequent, small-scale disturbances tend to improve habitat for lynx.

Habitat conditions near roads may favor snowshoe hares and, thus, attract lynx, increasing their vulnerability to illegal killing and vehicle collisions. Roads may also increase access of competing carnivores (e.g., coyote and bobcat) to winter habitats used by lynx (Aubry et al. 1999a). Limited field research suggests that high-use roads may affect the spatial distribution of lynx by truncating home ranges (Aubry et al. 1999a). Apparently, lynx can tolerate moderate levels of human disturbance (Aubry et al. 1999a).

Preferred. Preferred foraging habitat for lynx is early successional forests where snowshoe hares are abundant. These habitats may result from natural or human-set fires, logging, windthrow, or tree diseases. Such stands provide preferred habitat for snowshoe hares from about 10 to 30 years following their establishment. Dense conifer stands are also an important habitat component for snowshoe hares and, therefore, lynx.

Shrubs and small-diameter trees provide important habitat for snowshoe hares. A shrub stage is generally lacking in the regeneration cycle of southern boreal forests, but occurs in canopy gaps within old-growth forest and in riparian areas (Buskirk et al. 1999). Red squirrels, the second most important prey item of lynx, require mature, cone-producing conifers which are also abundant in old-growth forests (Buskirk et al. 1999). Since the two most important prey species of lynx are likely to be most abundant in old-growth forests in the southern Rocky Mountains, it follows that lynx will be most abundant in old-growth forests as well.

Lynx tend to not cross openings greater than 100m in width and do not hunt in open areas. Travel corridors are important for lynx movements within their home ranges and for dispersal movements. Suitable travel corridors are characterized by woody vegetation greater than 2 m (6.6 ft) in height with a closed canopy and close proximity to foraging habitat.

Den Sites. Preferred denning habitats are mature or old-growth coniferous forests stands greater than 1 ha (2.5 acres) in size with abundant large woody debris on the forest floor. Fallen trees and upturned stumps enhance the vertical and horizontal complexity of the habitat and provide important thermal and hiding cover for kittens. Other important attributes of denning habitat are multiple suitable denning sites, minimal human disturbance, proximity to preferred foraging habitat, and connectivity among preferred use areas. Suitable denning areas also provide safe havens for adult lynx. Den sites include caves and cavities under ledges, trees, and logs (Fitzgerald et al. 1994).

FOOD HABITS / FORAGING BEHAVIOR

The following information is taken from Aubry et al. (1999a) unless otherwise cited. Lynx depend heavily on snowshoe hares as their primary prey, as evidenced by the nearly complete overlap of their respective distributions in North America. Lynx tend to occupy habitats where snowshoe hares are most abundant. Even though snowshoe hare densities are typically low in southern boreal forests, they remain the predominant prey species in the lynx diet. However, alternative prey may be continuously important to the overall diet of lynx in the southern Rocky Mountains. The most important alternative prey species is the red squirrel (*Tamiasciurus hudsonicus*) followed most likely by cottontails (*Sylvilagus* spp.), blue grouse (*Dendragapus obscurus*) and a variety of small mammals (Aubry et al. 1999a; Best 2002). Ungulates represent an insignificant proportion of lynx diets, but lynx do occasionally kill young ungulates and scavenge on ungulate carcasses (Fitzgerald et al. 1994; Koehler and Aubry 1994).

POPULATION DYNAMICS

Life History. Lynx are one of three wild members of the cat family (Felidae) found in the southern Rocky Mountains, the other two being the bobcat (*Lynx rufus*) and mountain lion (*Puma concolor*) (Fitzgerald et al. 1994; Koehler and Aubry 1994; Cook et al. 2000). The lynx is a medium-sized cat weighing from 5-15 kg (11-33 lbs) (Fitzgerald et al. 1994). Males are slightly larger than females, with males averaging 10 kg (22 lbs) and females 8.5 kg (19 lbs) (Koehler and Aubry 1994). Their paws are large, offering greater support on snow, and their legs are relatively long—also an advantage in deep snow. Lynx are primarily nocturnal and solitary, except females with kittens and when adults come together to breed (Fitzgerald et al. 1994; Koehler and Aubry 1994). Lynx become sexually mature as yearlings, but breeding may be delayed if prey is scarce (Fitzgerald et al. 1994). Breeding occurs in the spring and litters average about 3 young which are raised by the mother (Fitzgerald et al. 1994). Few wild lynx live beyond the age of 6 years, but some individuals may live as long as 11 years (Aubry et al. 1999a). Aubry et al. (1999a) characterized lynx population dynamics in southern boreal forests as being generally similar to those of lynx populations in northern boreal forests during times of snowshoe hare scarcity. Generally, these population characteristics include low yearling pregnancy rates and litter sizes, low overall kitten production, high kitten mortality, and low lynx densities. Koehler and Aubry (1994) attribute the lack of dramatic snowshoe hare cycles and corresponding lynx population cycles in southern boreal forests to additional predators and competitors of snowshoe hares, the presence of alternative prey species, and increased patchiness of suitable habitat in southern boreal forests.

Population Density. Aubry et al. (1999a) note that lynx population densities in southern boreal forests (2-3 lynx/100 km² [39 mi²]) are similar to those in northern forests when snowshoe hare populations are low. They speculate that relatively stable and low snowshoe hare populations in southern boreal forests cause generally low lynx populations that lack the characteristic dramatic population fluctuations of northern populations. In contrast, densities in excess of 35 lynx/100 km² (39 mi²) have been observed in northern boreal forests following peaks in snowshoe hare populations (Koehler and Aubry 1994).

Home Range. Unless otherwise cited, the following information is from Aubry et al. (1999a), who reviewed and summarized studies of lynx home ranges in southern boreal forests. Home ranges of lynx in southern boreal forests tend to be about 1.5 times the size of lynx home ranges in northern forests dur-

ing periods of low snowshoe hare densities. Average home range sizes for males and females in southern habitats are 151 km² (58 mi²) and 72 km² (28 mi²), respectively. Male home ranges overlapped with those of 1-3 females in most studies. Ridges, major rivers, and major highways may define home range boundaries. Aubry et al. (1999a) advise caution in interpretation of these results because of multiple incongruities among studies.

Causes of Death. Causes of mortality in lynx include trapping or shooting (illegal in the southern Rocky Mountains) by humans, vehicle collisions (especially for translocated lynx), starvation (often significant), and predation (Fitzgerald et al. 1994; Aubry et al. 1999a). Known predators of lynx include gray wolf (*Canis lupus*), mountain lion (*Puma concolor*), wolverine (*Gulo gulo*), and adult male lynx (Fitzgerald et al. 1994). Because of their low population densities and solitary nature, disease is probably not a major mortality factor in lynx populations (Fitzgerald et al. 1994). The average annual natural mortality rate for adult lynx is about 27%, but kitten mortality may approach 90% when snowshoe hares are at low densities—the normal situation in southern boreal forests (Koehler and Aubry 1994).

Population Structure and Viability. In the southern Rocky Mountains, habitats suitable for lynx and their primary prey (snowshoe hares and red squirrels) are high-elevation boreal forests, which tend to be isolated and fragmented in their distribution (Aubry et al. 1999). Human disturbances have increased isolation and fragmentation of lynx habitat (Seidel et al. 1998; Buskirk et al. 1999). This disjunct spatial arrangement of suitable habitat favors a metapopulation structure for lynx populations (Buskirk et al. 1999; McKelvey et al. 1999b). Metapopulations are characterized by several subpopulations, each occupying a suitable patch of habitat, that are linked (genetically and demographically) by individuals that move between subpopulations (Levins 1969, cited in Meffe and Carroll 1997).

Patches of lynx habitat vary in their size and quality. Larger, high-quality habitats may support “source” populations whose combined reproductive and survival rates produce excess individuals that become dispersers (Meffe and Carroll 1997:211). Smaller or low-quality habitats are more likely to support “sink” populations that would eventually go extinct without the influx of immigrants from other subpopulations (Meffe and Carroll 1997:211-212). For the metapopulation to persist, the colonization rate of extinction-prone subpopulations must greatly exceed their extinction rate (McKelvey et al. 1999b). If source populations are few, the loss of one or more of these pop-

ulations could theoretically destabilize the entire metapopulation and lead to its eventual extinction (McKelvey et al. 1999b). McKelvey et al. (1999b) speculate that many southern lynx populations exist near the source-sink threshold, and may change status with small changes in the size or quality of habitat patches.

Connectivity that affords sufficient rates of dispersal among isolated habitats is the key to metapopulation health and persistence (Koehler and Aubry 1994; McKelvey et al. 1999b). Lynx populations in the southern Rocky Mountains are probably effectively isolated from larger populations to the north by broad expanses of unsuitable lynx habitat (McKelvey et al. 1999b). This isolation increases the importance of maintaining sufficient habitat to support source populations in the southern Rockies.

Determination of viable population size requires information on age structure, reproductive parameters, and survival rates that is lacking for southern lynx populations (McKelvey et al. 1999b). Thus, reliable minimum population viability analyses for lynx in southern boreal forests are not available, and we must resort to general conservation biology theory to develop minimum population and conservation area recommendations.

Because ideal situations rarely exist in nature, the “genetically effective population size” (N_e) is probably always smaller than the actual census size (N_c) of the population (Meffe and Carroll 1997:172). A genetically effective population is generally defined as an ideal, stable population with randomly mating individuals, even sex ratio, equal birth rates among females, and non-overlapping generations (Meffe and Carroll 1997). While empirical data for determining N_e of southern lynx populations is not available, general knowledge of lynx population dynamics would suggest an effective population size in the range of 0.5-1.0 of the census population size. We will base subsequent calculations on $N_e = 0.75 N_c$.

Franklin (1980) recommends that subpopulations should not be allowed to fall below an effective population size of 50 ($N_c = 67$). However, a major goal of the southern Rocky Mountains conservation plan is to preserve or restore evolutionary processes in natural systems. Genetic variation within and among individuals comprising populations of animals is the currency of evolution. In short, genes that confer advantages to individuals are selected over time (because their carriers are more fit and survive longer) and non-advantageous genes are not selected as often. As a general rule, the preservation of “evolutionarily important amounts of quantitative genetic variation” requires effective population sizes of “at least several hundreds

of individuals” (Lande and Barrowclough 1987:119; but see Lande 1995). Franklin (1980) recommended an effective population size of 500 for long-term conservation. Following the “precautionary principle” (Meffe and Carroll 1997:546), we recommend protected habitat complexes for lynx that are large enough to support an effective population of at least 500 lynx ($N_c = 667$).

Using density estimates of 2 lynx/100 km² (39 mi²) interconnected complexes of suitable lynx habitat totaling about 33,000 km² (12,750 mi²) would be necessary to support a lynx metapopulation with an effective population size of about 500 animals (i.e., a source population). We recommend two or more such lynx refugia within the southern Rocky Mountains ecosystem to increase the probability of lynx population persistence. Ideally, lynx refugia should be comprised of old-growth forest subject to natural disturbance regimes, which create and sustain a dynamic habitat heterogeneity favored by lynx and their primary prey (Buskirk et al. 1999). Any patch of suitable or potentially suitable lynx habitat of at least 1,000 km² (386 mi²) in size should be considered of potential conservation significance for lynx (McKelvey et al. 1999c), especially if similar patches occur within 100 km (62 mi). Habitat complexes smaller than 33,000 km² (12,750 mi²) that are greater than 100 km (62 mi) from a source population may require periodic augmentation by translocations of lynx to ensure metapopulation persistence.

Population Status. As of March 2002, 39 to 56 reintroduced lynx were free-ranging in southeastern Colorado, but reproduction by reintroduced lynx had not been documented (CDOW 2002). Lynx are federally listed as threatened and are fully protected in Colorado, Utah, and Wyoming. Except for the reintroduced population in Colorado, lynx are considered extirpated from the remainder of the southern Rocky Mountain ecosystem (Shinneman et al. 2000)

MOVEMENTS

Dispersal. The following information is from Aubry et al. (1999a), unless otherwise cited. Dispersal is the movement of an organism from a place of residence to its first successful breeding location (Shields 1987, cited in Aubry et al. 1999a). No successful dispersal movements of lynx in southern boreal forest have been reported in the literature. Dispersal distances in excess of 100 km (62 mi) are considered typical for lynx; and dispersal success is a function of the dispersal capability of the species, the size of habitat patches, and the distance between habitat patches. The chances of successful dispersal diminish as

population islands decrease in size and the distances separating them increase (McKelvey et al. 1999b). Aubry et al. (1999a:386) suggest that lynx inhabiting the more fragmented habitats found in southern boreal forests make occasional “exploratory” movements prior to actual dispersal. Movements of up to 38 km (24 mi) have been observed, after which the lynx returned to their previous home ranges. Similar movements have not been observed in the more contiguous northern boreal forests.

Long-distance Movements. When snowshoe hares are scarce (< 0.5/ha [0.2/acre]), lynx may abandon their home ranges and move long distances, presumably in search of new territories with more abundant prey (Koehler and Aubry 1994). Long-distance movements exceeding 1,000 km (621 mi) have been reported. Results of such movements may be the re-establishment of lynx in vacant suitable habitats or the augmentation of marginal populations near the southern edge of the lynx’s range (Koehler and Aubry 1994). However, Ruggiero et al. (1999) note the lack of hard evidence that long-distance movements actually result in successful dispersal or the augmentation of distant populations.

Barriers to Movement. Obstacles that may impede the movement of lynx across the landscape include paved roads, human developments, large rivers, and large expanses of open or unsuitable habitat (Koehler and Aubry 1994; Aubry et al. 1999a; Ruggiero et al. 1999). Koehler (1990, cited in Aubry et al. 1999a) observed that lynx traveled the edges of meadows, but only crossed openings of less than 100 m (109 yards) in width. However, for some documented lynx movements, successful crossings of busy, paved highways and large rivers had to have occurred (Aubry et al. 1999a).

Use of Linkages. Aubry et al. (1999a) define adequate travel cover as wooded linkages with 420-640 trees/ha (170-260 trees/acre) and possibly shrub-dominated habitats during snow-free periods. Aubry et al. (1999b) caution that there is no empirical evidence of the use of linkages by lynx.

ECOLOGY

Interspecies Interactions. The distribution and abundance of the snowshoe hare, directly affects the lynx’s geographic distribution, habitat selection, foraging behavior, reproductive success, and population density (Koehler and Aubry 1994). Other predators that compete with lynx for snowshoe hares include coyote and bobcat and, to a lesser extent, red fox (*Vulpes vulpes*), great horned owl (*Bubo virginianus*), marten (*Martes*

americana), fisher (*Martes pennanti*), and wolverine (Koehler and Aubry 1994). Because of their large paws and long legs, lynx are more effective than other predators in deep snow (Koehler and Aubry 1994). However, maintained roads and snowmobile trails may offer increased access to deep-snow habitats by coyotes and bobcats, increasing competition for scarce food resources during times of high energy demands (Koehler and Aubry 1994; Seidel et al. 1998; Buskirk et al. 1999). Mountain lion predation of lynx has been documented in southern boreal forests, which may add to factors limiting lynx populations (Aubry et al. 1999a). The combination of exploitation competition from bobcats, coyotes, and other mesopredators and interference competition from mountain lions may significantly affect lynx population in southern boreal forests, but sufficient data to demonstrate such an effect are not available (Aubry et al. 1999a; Buskirk et al. 1999). However, Buskirk et al. (1999) note that the presence of gray wolves will likely reduce overall competition for preferred prey of lynx and they suggest, for example, that the Greater Yellowstone Ecosystem should improve as lynx habitat in response to the re-establishment of wolves in the ecosystem.

Ecological Effects. The ecological effects and relationships of lynx have not been elucidated with any clarity in southern boreal forests. Because lynx are heavily dependent upon snowshoe hare and red squirrel for prey, lynx populations are expected to respond directly to changes in populations of these prey species (Koehler and Aubry 1994; Aubry et al. 1999a). However, lynx are not the only predator of snowshoe hares and red squirrels. There is no evidence suggesting that lynx exert a regulatory effect on prey populations in southern boreal forests, except perhaps as a member of the larger suite of predators of snowshoe hares, red squirrels, and other small prey species.

JUSTIFICATION AND FOCAL VALUE

Flagship. Wild predatory cats like the lynx fascinate a wide range of people. Conservation efforts focusing on lynx should generate popular support.

Umbrella. Lynx have very large area requirements for population persistence. They require a dynamic mosaic of various boreal forest successional stages mimicking that resulting from natural disturbance regimes. Protection of habitat for viable populations of lynx will, by inclusion, protect the habitats of many other southern boreal forest-dwelling species.

Wilderness Quality Indicator. The lynx needs old-growth, unlogged forests subject to dynamic natural disturbance

regimes for optimum habitat. These parameters are the essence of “wilderness.” The best protection for such forests is formal designation as Wilderness Areas.

MANAGEMENT RECOMMENDATIONS

“Our challenge, from the perspective of maintaining lynx and their prey in the context of ecosystem management, is to design management strategies that result in dynamic, sustainable landscapes that approximate the composition of natural systems” (McKelvey et al. 1999c:428).

Establish Refugia. Koehler and Aubry (1994) recognize the importance of protected areas to the persistence of lynx populations in southern boreal forests. Areas designated for lynx conservation should contain a mixture of forest age classes and structural conditions similar to the habitat configuration that would result from natural disturbance regimes (Koehler and Aubry 1994). This complex of habitat types can be approximated by a negative exponential stand-age distribution model based on the estimated natural fire regime for the area, which mimics large-scale stochastic disturbance regimes (McKelvey et al. 1999c). Such an approach would provide a continuum of stand ages in a variety of spatial configurations that will benefit lynx, their primary prey species, and other boreal forest-dwelling species (McKelvey et al. 1999c). However, McKelvey et al. (1999c) caution that if management prescriptions include planned or artificial disturbance mechanisms, planned disturbance rates should be somewhat less than the historically derived estimate. This would prevent potential under-representation of old-growth forests, which are important for lynx and take a long time to recover. We recommend the establishment of at least two protected refugia for lynx. Refugia should each contain at least 33,000 km² (12,750 mi²) of interconnected habitats that collectively provide critical ecological needs of lynx and support a viable source population.

Protect Subpopulations. The fragmented arrangement of suitable habitat in the southern Rocky Mountains favors a metapopulation structure for lynx populations (Buskirk et al. 1999; McKelvey et al. 1999b). Metapopulations are comprised of several smaller subpopulations whose individual viability is critical to the long-term persistence of the metapopulation. Management practices should consider the vulnerability of local, isolated lynx populations to extinction. Management considerations should include habitat protection and enhancement, protection or creation of linkages, prohibition of take, and population augmentation through translocation of lynx from source populations.

Protect or Restore Linkages. Because lynx tend to avoid openings and travel in areas with relative dense woody cover, direct links of forest cover among suitable habitat blocks may be essential to lynx persistence (Koehler 1990, cited in Aubry et al. 1999a; Aubry et al. 1999a). Management practices should result in the protection, restoration, and enhancement of potential movement linkages for lynx.

Reintroduce Lynx to Suitable Habitats. The southern Rocky Mountains are effectively isolated from potential source populations (McKelvey et al. 1999b). Reintroduction is the only feasible method for restoring lynx populations to this region. One reintroduction project is underway (CDOW 2002). Pending completion, monitoring, and assessment of the current project, additional reintroductions should be considered for areas of suitable habitat of sufficient size to support subpopulations of lynx.

Road Closures. Roads and recreation trails in otherwise suitable lynx habitat will likely result in human disturbances that detrimentally affect lynx populations, especially during critical times such as the denning season and winter (Aubry et al. 1999a). During winter, roads and trails may facilitate access by competing predators, especially coyotes and bobcats, to deep snow areas, where lynx would otherwise have a competitive advantage (Aubry et al. 1999a). Roads should be eliminated, reduced, or seasonally closed in critical lynx habitat areas. We do not advocate the closing of important thoroughfares or paved roads or highways.

Monitoring. Because of their rather specific habitat preferences (southern boreal forests) and large area requirements (~2 lynx/100 km² [39 mi²]), lynx populations are very sensitive to habitat loss or degradation. Resource managers must implement effective monitoring strategies to detect effects of land management practices on habitat quality and numerical abundance and density of lynx and other biota, especially snowshoe hares and red squirrels (McKelvey et al. 1999c). Monitoring of hare populations with pellet transects and lynx populations with snow tracking or other effective techniques are critical components of monitoring strategies (McKelvey et al. 1999c). Monitoring plans should have defined criteria for assessing results and include adaptive management provisions for modifying management plans and their implementation if necessary (McKelvey et al. 1999c).

Promote Natural Disturbance Regimes. Fire is an important agent in creating forest diversity. Additionally, factors such as windthrow, disease, and insect infestations create import microhabitats for lynx, especially within old-growth

forests (McKelvey et al. 1999c). McKelvey et al. (1999c) believe that the dynamic mosaic of habitat types resulting from natural disturbance regimes operating at the landscape scale in southern boreal forests provide high-quality habitat for lynx, their primary prey, and other members of the biotic community. With few exceptions (e.g., threats to human life and property), natural disturbance regimes should be allowed to operate freely in large protected reserves. To the extent practical and feasible, natural disturbance patterns should be mimicked by land use management actions in compatible use areas.

Multi-jurisdictional Planning. Conservation planning for broad-ranging carnivores, such as lynx, must be conducted over vast spatial scales and must consider connectivity among sub-populations (Noss et al. 1996). Large areas required for lynx population persistence, will most certainly span the jurisdictions of multiple land management agencies and private lands. Establishment of regional, multi-jurisdictional planning authorities or arrangements through appropriate means (e.g., legislative or administrative) should be encouraged and pursued.

Educate Managers and the Public. Wildlife managers and policy makers need a thorough understanding of lynx ecology in order to establish appropriate policies and make sound management decisions. In addition, the public needs accurate information and knowledge about lynx to inform their opinions and values and their understanding of appropriate management measures. Knowledge is the key to informed conservation actions and advocacy by both agencies and the public.

Conduct Meaningful Research. Knowledge of lynx ecology in southern boreal forests is extremely limited (Koehler and Aubry 1994; Aubry et al. 1999a). Research is needed in the following areas: foraging ecology, habitat use, den site characteristics, optimum habitat composition and structure (for lynx and key prey species), and effects of forest management practices and natural or artificial disturbance events on the quality and quantity of lynx habitat (Koehler and Aubry 1994).

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