

BIOLOGICAL FOUNDATIONS
FOR MANAGEMENT OF
MAMMALIAN PREDATORS
IN ARIZONA

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Arizona Game and Fish Department

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PREFACE

Arizona Game and Fish Department recognizes and endorses the need to manage mammalian predators to maintain historic distribution in the state and to allow for maximum aesthetic, economic, and recreational uses of these species commensurate with existing wildlife. Mammalian predators in Arizona have diverse life strategies and occupy various habitats ranging from arid, low deserts to wetter, higher elevation forested regions. Given the present state of knowledge about mammalian predators in the state, many unanswered questions remain regarding fundamental biology of these species. Monitoring trends of populations and developing effective management prescriptions for these species thus remain formidable tasks.

This book was conceived to provide an overview and synthesis of information about various aspects of life histories and management of mammalian predators relevant to Arizona. Prior to publication of this book, there was no single source available to provide this perspective. Our goal was to provide a brief introduction to available information that provides biological foundations for management of mammalian predators in Arizona.

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Chapter 1

Black Bear

Stan Cunningham, Research Branch
Arizona Game and Fish Department

Common name - Black Bear
Scientific name - *Ursus americanus*

Life History

Description

Black bears (*Ursus americanus*) in the Southwest average between 2 and 2.5 feet tall at the shoulders when standing on all 4 legs, and are about 6 feet tall when standing on hind legs. Adult females and males average about 150 and 275 pounds, respectively. Some males might weigh more than 400 pounds, but this is uncommon.

Natural life expectancy of black bears varies regionally (Bunnell and Tait 1981), but they often live more than 20 years in the wild; a 27-year old female and a 23-year old male were captured in New Mexico (Costello et al. 2001). Two different females in New Mexico produced litters at 22 years of age.

Several color phases of black bears exist indicating a number of gene pools. Lighter-colored bears occur in arid regions (Cowan 1938, Piekielek and Burton 1975). Among 105 bears captured in the Pinaleno Mountains of Arizona, 10% were black, 42% were dark brown, 45% were brown, and 3% were blond (Waddell and Brown 1984). In New Mexico, 75% of black bears captured were brown phase (Costello et al. 2001).



A black bear in a coniferous montane forest meadow.

Distribution

Before European settlement, black bears occurred in all forested habitats in North America, including Canada and Mexico. They still occur in 38 states, 11 Canadian provinces, and 7 Mexican states. Black bears in Arizona occur from northwest to southeast portions of the state (Figure 1).

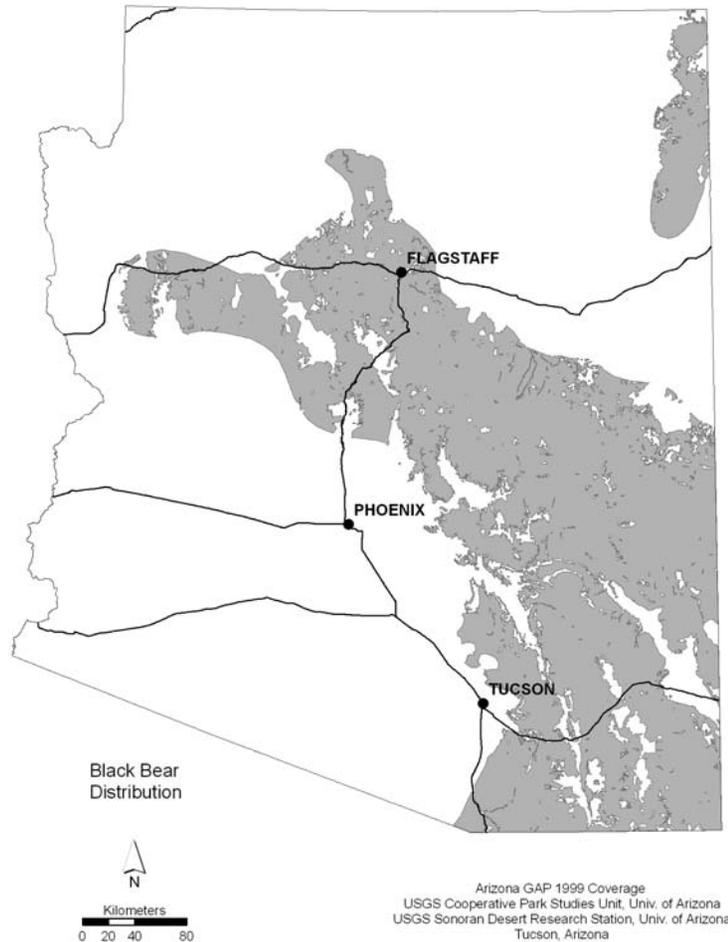


Figure 1. Distribution of black bears in Arizona.

Black Bear

In 1999, the statewide population of adult black bears in Arizona was estimated to be 2,500 animals occupying about 12,600 square miles of habitat (Arizona Game and Fish Department 2000). Black bears occur in most woodland habitats, including Madrean evergreen woodlands, mixed conifers, spruce-fir (*Picea* spp.–*Abies* spp.), ponderosa pine (*Pinus ponderosa*), and pinyon-juniper (*Pinus edulis*–*Juniperus* spp.), and are common in interior chaparral habitats adjacent to forests.



During prickly pear (*Opuntia engelmannii*) fruiting season, black bears also occupy habitats in the Sonoran Desert, often along river corridors. Black bears rarely are seen in areas of coniferous forests north of the Colorado River in Arizona.

Arizona black bear densities are greatest in central and southeastern regions (Figure 1). Diversity of vegetation in desert sky islands (remnant stands of mixed conifers, Madrean evergreen woodlands, and ponderosa pine at higher elevations, surrounded by interior chaparral and Sonoran Desert scrub vegetation at lowest elevations) and just south of the Mogollon Rim provides more reliable fruit and nut food sources than other areas. Black bear densities in Arizona are similar to or exceed those documented elsewhere in southwestern North America, and are relatively high in comparison to other populations in the United States (range 0.03 to 1.93/square mile). In regions of more homogeneous ponderosa pine and mixed conifers north of the Mogollon Rim, densities are one-half to one-third of densities in central and southeastern Arizona. Fewer bears inhabit areas north of the Mogollon Rim because of reduced plant diversity and decreased availability of food resources (LeCount and Yarchin 1990).

Habitat

Black bears in the Mazatzal Mountains and south of the Mogollon Rim in Arizona prefer Madrean evergreen woodlands and interior chaparral habitats, and avoid desert grasslands (LeCount et al. 1984, Mollohan 1985). Females north of the Mogollon Rim avoid open areas of ponderosa pine, and select smaller areas (less than 5 acres) of maple (*Acer* spp.) and mixed conifer “maternity sites” that provide adequate cover, water, green grass, and large trees (Mollohan 1985). Black bears in east-central Arizona prefer steep slopes (more than 20%) in old growth mixed conifer stands with multi-storied canopies and greater horizontal cover near the ground (LeCount and Yarchin 1990). Black bears avoid ponderosa pine forest, unless it provides structural characteristics similar to mixed conifer forests.



Black bears in Arizona prefer madrean evergreen and mixed conifer vegetation.

Regardless of vegetation type, black bears prefer horizontal cover up to 6.5 feet, less than 50% canopy overstory, presence of trees more than 2 feet in diameter, and slopes more than 20% (LeCount et al. 1984, Mollohan 1985, LeCount and Yarchin 1990, Cunningham et al. 2001). Density of shrubs at mid-day bedding and feeding sites is 5 times greater than density at random points. Black bears prefer trees greater than 2 feet in diameter in bedding areas (Mollohan 1985, Cunningham et al. 2001), likely to reduce predation on cubs (Herrero 1972). Black bears select habitats with tall cover near food sources (LeCount et al. 1984, Mollohan 1985, Cunningham et al. 2001). All research studies suggest cover is more important than food in selecting habitats.

Hibernation

Hibernation likely evolved to avoid scarcity of food (Pelton 1982). Timing of all other black bear activities is probably an evolutionary consequence of hibernation. Physiological state of bears during winter is considered to be true hibernation (Hellgren 1998); some biologists consider it to be the most refined response to starvation of any mammal (Nelson 1980). A hibernating black bear generally does not drink, eat, defecate, or urinate for up to 7 months (Nelson 1980, Hellgren 1998). Energy stored in body fat generates metabolic activity. Bears are unique in recycling waste products of fat metabolism, whereas small mammals must periodically become physically active during dormant periods to eat and to eliminate wastes. During hibernation, bears save energy by reducing metabolic rate as much as 50% and by lowering heart rate from 40 to 50 beats per minute to 8 to 10 (Hellgren 1998). Unlike small mammals that lower body temperature to 50 degrees Fahrenheit (°F) or less during hibernation, the body temperature of a hibernating black bear drops from a range of 98.6 °F to 100.4 °F to 87.8 °F to 95 °F (Hellgren 1998). Maintenance of near-normal body temperatures allows black bears to arouse quickly if disturbed.



Biologists return this black bear to its hibernation den following capture and collection of data to better understand its biology and habitat needs.

Diets

Black bears are primarily herbivorous, and eat mostly grass for the first 2 months after hibernation, switch to early maturing fruits and animal matter (primarily insects) in mid-summer, then eat hard and soft nuts produced by trees (mast) during late summer and autumn. Areas of the Southwest with several species of oaks (*Quercus* spp.) provide premium habitat for bears; production of acorns varies among years and oak species (Mollohan and LeCount 1989).

Oaks and junipers produce the most consistently eaten mast species in the Southwest. Manzanita (*Arctostaphylos* spp.), serviceberry (*Amelanchier bakeri*), prickly pear, juniper, raspberries (*Rubus strigosus*), and 6 species of oaks were eaten most in Madrean evergreen woodland and interior chaparral habitats (LeCount et al. 1984, Mollohan 1985, Cunningham et al. 2001). Black bears in ponderosa pine habitats moved each autumn into Madrean evergreen woodland at lower elevations to take advantage of more abundant mast (Mollohan 1985). Black bears ate grasses and insects in east-central Arizona, but switched to oaks, squawroot (*Conopholis mexicana*), and gooseberry (*Ribes pinetorum*) as these foods became available in late autumn (LeCount and Yarchin 1990). In New Mexico,



A black bear in mixed conifer habitat.

oaks, juniper, squawbush (*Ribes* spp.), and chokecherry (*Prunus virginiana*) were the most important plants in diets of black bears (Costello et al. 2001).

Although black bears are capable of surviving on a primarily carnivorous diet (Schwartz and Franzmann 1991), vegetation comprises more than 75% of their diets in the Southwest. Black bears

in the Southwest sometimes prey on mule deer (*Odocoileus hemionus*), javelina (*Pecari tajacu*), elk (*Cervus elaphus*), and domestic livestock, as well as smaller animals.

Nutrition and Reproduction

Annual variation in availability of food resources contributes to positive correlations between abundance of food and reproduction by black bears in North America (Rogers 1976, Eiler et al. 1989, Elowe and Dodge 1989, Miller 1994, Costello et al. 2003), or differences in availability of food resources between study areas or home ranges (Elowe and Dodge 1989, Schwartz and Franzmann 1991). Productivity of females has been linked to nutritional condition (Noyce and Garshelis 1994, Samson and Huot 1995). Production of oak mast does not solely affect reproduction. Perhaps only a minimum threshold of high quality food is needed for successful reproduction (Costello et al. 2003).

Black bears in the wild have low reproduction rates because of small litter sizes, 2–3 year intervals between litters, food shortages caused by climatic events, attainment of sexual maturity at 4–7 years of age, and stress caused by disturbances or higher densities of bears, both of which can affect reproductive physiology (Jonkel 1978). Age of first reproduction and breeding interval can be influenced by nutritional condition; earlier breeding occurs when nutrition is high (Bunnell and Tait 1981). Reproductive rates are not affected by density (i.e., reproduction will not lessen if abundance of bears increases) of a population (Beecham 1980, Bunnell and Tait 1981). Removal of adult females from a population does not contribute to increased productivity of remaining females (LeCount 1987a). Also, removal of males and females from a population does not correspond with higher reproductive rates (Beecham 1980).

Female black bears give birth in dens during hibernation in winter and must meet requirements of gestation and lactation, as well as other metabolic processes. Bears developed 2 strategies to meet requirements of reproduction that

Black Bear

likely are linked to hibernation (Costello et al. 2001). Black bears breed during May and June and experience delayed implantation, in which fertilized eggs remain in the uterus for several months before implanting in the uterine wall. Although breeding occurs in spring or summer, development of fetuses usually does not begin until a female starts to construct or enters her den (Jonkel 1978). If a female is nutritionally sound at onset of hibernation, embryos (called blastocysts at this stage) implant in the uterine wall and a gestation period of about 90 days begins. Cubs usually are born in January or early February (Hellgren 1998) and weigh only 8–10 ounces. Failure to produce litters has been linked to poor nutritional condition (Noyce and Garshelis 1994).

Mating and Cub Rearing

During the May to June breeding season, males travel long distances to find and breed different females. Barren females and females with yearlings generally will be receptive to males. Frequent fighting might occur among males



Arizona Game and Fish Department biologist Stan Cunningham with young black bear cubs taken temporarily from the den to collect important data to help biologists better understand the life history of black bears in Arizona.

in areas with dense populations of bears. Adult males often kill cubs; females with new cubs likely avoid males by selecting different habitats (LeCount et al. 1984). Females with young cubs rarely breed (LeCount 1983). A minimum interval between successful births usually is 2–3 years.

Males probably kill cubs because estrus can be induced in females if lactation abruptly ceases for 2 to 3 days (Jonkel 1978). Thus, if a male kills cubs and remains with the mother for a short time, he likely can breed with her. Predation by males likely is the primary cause of mortalities of cubs (LeCount 1987b). Although male black bears might recognize and not kill their own cubs (Jonkel 1978), this has not been proven.

Cubs in the southwestern United States generally remain with their mothers until 14 to 18 months old (LeCount 1983, Costello et al. 2001), and den

with their mothers during their second winter. When an adult female comes into estrus, she severs contact with her cubs, and the family group bond is broken (Schwartz and Franzmann 1992). Young bears, especially immature males, then disperse, often for long distances, perhaps to avoid encounters with adult males. Conversely, young females often establish home ranges adjacent to home ranges of their mothers (Schwartz and Franzmann 1992, Costello et al. 2001).

Litter Size

Litter sizes range from 1 to 5 cubs, and sex ratios usually are equal (LeCount 1987a). Mean size of litters in studies in the Southwest ranged from 1.7 to 2.3 cubs (Table 1). Size of individual litters ranged from 1 to 3 cubs. In North America, mean size of litters decreases at more eastern longitudes and increases at higher latitudes (Bunnell and Tait 1981). Although presence or absence of a litter may be affected by nutrition, size of a litter is not, and likely is genetically fixed (Bunnell and Tait 1981).

Table 1. Locations, major reproductive characteristics, and references for studies of black bears in southwestern North America.

Location	Mean birth interval (years)	Mean litter size	Mean age at first litter (years)	References
Arizona	Unknown	1.9 to 2.3	1.8 to 4.7	LeCount 1982, 1987a, 1990; Cunningham et al. 2001
Colorado	2.2	2.0	4.7	Beck 1991
New Mexico	1.7 to 2.0	1.7 to 1.9	5.6 to 5.8	Costello et al. 2001

Birth Intervals

Mean number of years between producing cubs (birth interval) in the Southwest ranged from 1.7 to 2.2 years (Table 1). Birth interval has been linked to female nutritional condition. Litters in successive years likely indicate no cubs survived from the previous year because females with cubs rarely breed (LeCount 1983); short birth intervals might indicate relatively poor growth of a population. For example, the lowest birth interval in central Arizona followed a wildfire (Cunningham and Ballard 2004), when no cubs survived. Average interval between successful rearing of cubs to dispersal likely is a more useful statistic than birth interval (Carrel 1987).

Age at First Reproduction

Mean age at first reproduction in southwestern North America ranged from 1.8 to 5.8 years (Table 1). Scarce forage or poor quality habitat might result in an older mean age at first reproduction (Bunnell and Tait 1981). Age at first reproduction is correlated with body weight in wild populations (Stringham 1990).

Age and experience of female bears also can affect survival of cubs. In New Mexico (Costello et al. 2001), females with first litters experienced significantly lower survival of cubs, compared to survival of cubs in subsequent litters (38% and 62%, respectively). Among 22 litters in Massachusetts (Elowe and Dodge 1989), 4 of 7 females with first litters lost entire litters, whereas none of 15 experienced females lost whole litters.

Distribution and Movements

Home Range.—Density of black bears is influenced by size of areas traveled by bears during normal activities. Smallest home ranges in the Southwest were in Madrean evergreen woodland/interior chaparral in the Mazatzal Mountains and just south of the Mogollon Rim in Arizona. Home ranges of females and males in these areas were 5.6 to 6.7 square miles and 13.5 to 44.4 square miles, respectively (LeCount et al. 1984, Mollohan 1985, Cunningham et al. 2001). Home ranges of females and males in ponderosa pine habitats were



A young black bear on the move, likely looking for a new place to establish a territory.

about 40.1 and 235 square miles, respectively. In general, male home ranges in the Southwest are similar to those in other areas, are 5 times larger than those of females, and often overlap 7 to 15 female home ranges (Bunnell and Tait 1981). Overlap of home ranges of black bears varies from slight (Jonkel and Cowan 1971, Rogers 1976) to extensive (Lindzey and Meslow 1977, Reynolds and Beecham 1980, Costello et

al. 2001). Extensive overlap of home ranges occurs in areas with patchy and unpredictable food sources (Reynolds and Beecham 1980, Costello et al. 2001).

Larger home ranges of black bears in areas with less available mast (ponderosa pine, spruce-fir, and even mixed conifer) result from movements to locate seasonally available food sources (Mollohan 1985, LeCount and Yarchin 1990, Costello et al. 2001). During years of food shortages, annual home ranges might greatly increase, even in good habitat. Home ranges of females in the Mazatzal Mountains in 2000 increased from 5.8 to 48 square miles when females left resi-

dent territories to search for food in nearby mountain ranges. Most long-range movements, particularly during times of food shortages, occur during late summer and autumn, making female black bears more susceptible to hunters (Costello et al. 2001).

Territory.—Territories (defended areas) of black bears are optimally spaced and evenly distributed where resources are plentiful. It is not beneficial for wildlife to establish and defend territories in areas where availability of food is patchy or unpredictable (Horn 1968, Wiens 1976).

Defense of territories generally is greater between females than males, suggesting that relative abundance of forages essential to reproduction affects territoriality. Populations of bears inhabiting areas of the West with concentrated



This black bear is sporting a new radiocollar in chaparral and Madrean evergreen habitat. Biologists hope to track the movements of this bear to better understand biological needs throughout the year.

food sources, such as berry patches, oak stands, and salmon (*Oncorhynchus* spp.) streams, generally have a dominance hierarchy with a high degree of home range overlap (Bunnell and Tait 1981) and less defense of territories.

Dispersal.—Dispersal of black bears 1 to 3 years old (subadults) begins during the breeding season. Males more often disperse further than females (Schwartz and Franzmann 1992). Dispersal increases the probability of hunters harvesting

black bears; sex-related behavioral differences among subadults should be considered in managing harvested populations (Schwartz and Franzmann 1992). Black bears might slowly repopulate a burned area, and transplants might be required to re-establish populations after large-scale fires (Cunningham et al. 2001).

Translocation.—Regulatory agencies responsible for management of black bears frequently are required to respond to calls related to “nuisance” bears. Nuisance bears often are captured and relocated in attempts to abate unacceptable behaviors and place animals in other areas where they might establish new home ranges. Probability of relocated bears returning to original capture sites is inversely related to distance moved (Sauer et al. 1969, Beeman and Pelton 1976, McArthur 1981, Rogers 1986). Bears that were moved more than 35 miles from capture sites seem less likely to return.

In New Mexico, 8 bears were translocated from 16 to 52 miles from capture sites, and 73% of bears returned to areas where they were captured within 1 to 328 days (Costello et al. 2001). In Arizona, 17 cubs were translocated after

being captured within the urban area of Phoenix during September 2000. After gaining sufficient weight and receiving medical attention, they were released in dens or in known bear habitat the following March to May. No cubs returned to the urban area of Phoenix, and some dispersed distances up to 124 miles.

Survival

Cubs.—Survival of cubs is extremely variable between populations and among years, and ranged from 0% to 90%. Survival rates of cubs in Arizona were similar to those for other populations in the Southwest (Table 2), but were lower than those in northern latitudes and central and eastern states. Cub mortalities usually are estimated by changes in sizes of litters between cubs and yearlings, but this method might underestimate mortality of cubs (Bunnell and Tait 1981, 1985).

Table 2. Locations, annual cub survival rates, number of cubs, and references for studies of black bears in western North America.

Location	Survival rate	Number of cubs	References
Arizona	0 to 0.90	16 to 43	LeCount 1982, 1987b, 1990; Cunningham et al. 2001
Colorado	0.41 to 0.73	39	Beck 1991
Mexico	0.81	25	Doan-Crider and Hellgren 1996
New Mexico	0.55	148	Costello et al. 2001

Nutrition seems to regulate the ability of a female to produce offspring independently of bear density. However, predation of cubs and subadults by adult males (density dependent mortality factors) is more likely to occur at high densities. Predation by bears is the second highest cause of mortalities of cubs and subadults, and is exceeded only by human harvest (Bunnell and Tait 1981). Adult males might limit or regulate densities of populations by killing or evicting younger males (Bunnell and Tait 1981), assuming mortality and dispersal of subadult males and females increase with increasing density of adult males.

Removal of adult males from a population might result in higher local abundance of black bears (Kemp 1976). In another study, removal of adult

males did not result in a local population increase (Miller 1990). If resident males recognize and do not kill their own cubs and kill fewer cubs than immigrant bears, over-harvest of males could reduce density of a population (Rogers 1977, LeCount 1987a). Fifty two percent of cubs survived in central Arizona (LeCount 1987a), but no cubs born after a catastrophic wildfire in the Mazatzal Mountains in Arizona survived their first year of life (Cunningham et al. 2001, Cunningham and Ballard 2004).

Subadults.—Survival of subadults is difficult to estimate because of long-range movements and prolonged period of time it might take for a subadult to establish a resident home range (Bunnell and Tait 1985). Hunting likely is the primary cause of mortality, and dispersing subadult males likely are particularly vulnerable to harvest (Elowe and Dodge 1989, Schwartz and Franzmann 1992). Survival of subadults in the Southwest varied among studies (Table 3).

Table 3. Locations, estimated annual survival rates, numbers (male and female) of subadults (1 to less than 3.5 years of age), and references for black bears studied in southwestern North America.

Location	Survival rate	Males/females	References
Arizona	0.57	30 (sexes not analyzed separately)	LeCount 1987b
Colorado	0.76 to 0.94	29/25	Beck 1991
New Mexico	0.85 to 0.95	97/159	Costello et al. 2001

Adult Males.—Hunting is a major cause of adult black bear mortality, and hunters generally kill more males than females among all age groups (Cowen 1972, Bunnell and Tait 1981). Although adult males will fight each other, they usually are killed by hunters or die from age-related problems (Bunnell and Tait 1981). Males are more vulnerable than females to hunters because of larger home ranges and more frequent long-range movements (Bunnell and Tait 1985). Long-range movements increase vulnerability of bears to deaths from a variety of factors.

Mean annual survival rates of adult males in the Southwest (Table 4) ranged from 0.7 to 1.0 and generally were greater than survival rates reported for populations in central and eastern United States (Bunnell and Tait 1985, Costello et al. 2001). Adult black bears in ponderosa pine habitats in Arizona had greater (35%) mortality from hunting (LeCount 1981). In comparison, lower mortality of adult males was recorded in a lightly exploited population in the Mazatzal



A black bear has 5 toes; the inner toe is the smallest toe.

Mountains (LeCount 1982, Cunningham et al. 2001). Biologists in New Mexico were surprised by high survival of adult males (Costello et al. 2001), because harvest is strongly correlated with density of hunters (Bunnell and Tait 1985).

Table 4. Locations, estimated annual survival rates, sample size (number studied) of adults (more than 3.5 years old), and references for studies of male black bears in southwestern North America.

Location	Survival rate	Number studied	References
Arizona	0.65 to 1.00	47	LeCount 1990, Cunningham et al. 2001
Colorado	0.70	21	Beck 1991
New Mexico	0.91	157	Costello et al. 2001

Adult Females.—Birth and mortality rates of adult females are critical data for management of black bears (Bunnell and Tait 1981, 1985; Costello et al. 2001). Habitat quality and abundance of breeding age adult females influence reproductive potential and growth of populations, whereas abundance of males influences maximum size of a population or social carrying capacity in an area. High mortality rate of female black bears, particularly bears more than 3 years old, can decrease population size (Costello et al. 2001). Similar to adult and subadult males, the greatest source of mortality of females generally is hunting, but females often appear to be killed by other bears, presumably adult

males (Bunnell and Tait 1985, Costello et al. 2001, Cunningham et al. 2001). Researchers commonly report equal sex ratios from capture data, but an equal sex ratio for harvested bears can indicate over-exploitation of a population (Bunnell and Tait 1981, 1985; Costello et al. 2001).

Sex composition of black bear harvest data in Arizona from 1964 to 1978 was 60% males and 40% females (LeCount 1981); from 1998 through 2002, the harvest ratio was 54% males and 46% females (Arizona Game and Fish Department 2003). Food shortages heighten vulnerability of females to hunters because of increased long-range movements (Bunnell and Tait 1981, Elowe and Dodge 1989, Costello et al. 2001).

Survival rates for adult female black bears in the Southwest (Table 5) generally are higher than survival rates (0.8 to 0.92) estimated in a population model of southwestern bears (Costello et al. 2001). Survival rates near the upper end of estimations (0.92) likely correspond with potential for increase of a population, whereas survival rates near the lower level (0.80) might eventually result in decline.

Table 5. Locations, estimated annual survival rates, sample size (number studied), and references for studies of adult (more than 3.5 years old) female black bears in western North America.

Location	Survival rate	Number studied	References
Arizona	0.85 to 0.98	27	LeCount 1990, Cunningham et al. 2001
Colorado	0.96	25	Beck 1991
Mexico	0.94	12	Doan-Crider and Hellgren 1996
New Mexico	0.92	250	Costello et al. 2001

Sources of Mortality.—Legal sport hunting is the major cause of black bear mortality, followed by intra-specific (among bears) fighting. Other causes of mortality include livestock depredation and nuisance bear harvest, poaching, collisions with automobiles, predation, disease, and starvation. Causes of black bear cub mortality include hunting, orphaning, disease, collisions with automobiles, and predation by black bears, mountain lions (*Puma concolor*), and bobcats (*Lynx rufus*).

Management and Conservation

Management Since European Settlement

Black bears were considered non-game animals in Arizona as late as 1928, and could be shot or trapped at any time. However, a new “game code” classified bears as big game in 1929, provided a month-long season, and prescribed a bag limit of 1 bear per hunter. Bears could not be trapped, but could be taken with the aid of dogs. Subsequently, cubs were protected in 1934 and bear hunting was closed south of the Gila River in 1936, although the season was later reopened in Cochise and Graham counties. Month-long seasons during autumn and spring were authorized in 1944, and black bears lost designation as big game animals in 1946. Yearlong hunting seasons were opened until 1954, except for closure in 1950. A tag was required to harvest a bear between 1954 and 1968, and hunting seasons were reinstated during autumn and spring. Black bears now are classified as game animals in Arizona.

Population Trends

It is difficult to determine trends of black bear populations. Harvest data alone are of limited use in this respect (Bunnell and Tait 1981, Costello et al. 2001), but harvest data from 1998 through 2002 (Arizona Game and Fish Department 2003) indicated abundance of black bears was similar to the level in 1980 (LeCount 1981), except for increased harvest (particularly of females) in 2000. However, trends in age data collected from bears harvested by hunters since 2000 indicate recruitment has been poor. If density of hunters remains similar, a decreasing trend in harvest levels for a 4-year period might indicate a declining population (Costello et al. 2001).



Biologists use radiotelemetry to track locations of radiocollared bears using hand-held antennae and receivers.

Monitoring and Surveys

Mark-recapture studies presently offer the only optimal method for determining black bear population densities, but the method provides unreliable estimates of sex and age ratios if fewer than 25 animals are captured per year. Although aerial surveys of previously marked grizzly bears (*Ursus arctos Linnecan*) provide reliable estimates of population size (Miller et al. 1997), aerial surveys

are not applicable for studying black bears because of black bear preference for thick cover. Newer techniques using deoxyribonucleic acid (DNA) fingerprinting might allow use of hair or fecal matter in conjunction with harvest data to estimate and monitor populations of black bears.

Scent-post surveys have been used to index abundance of black bears in the Mazatzal Mountains, but data were not published (J. Wegge, Arizona Game and Fish Department, personal communication). Triggered camera sightings of previously marked bears at baited sites provided estimates of sizes of grizzly bear populations (Mace et al. 1994), but biologists had to document different routes of visitation to bait sites and model for "bait shyness" to estimate sizes of populations accurately.

Harvest data can provide managers with useful information for monitoring and modeling populations, but harvest data alone might fail to detect short-term population changes (Bunnell and Tait 1981, Costello et al. 2001). Data on harvest should be evaluated in conjunction with data on age structure, as is currently done in Arizona. Ages and intervals of successful cub rearing can be estimated by determining the number and distance between annuli found in bear teeth (Carrel 1987).

Current Regulations

Increased interest by hunters prompted reclassification of black bears to big game status in 1968, when monitoring of harvest was implemented. A mandatory system to check all black bears harvested by hunters began in 1980. Other recent changes in regulations include eliminating baiting as a method of harvest. Harvest is legal for any bear except sows with cubs. Successful bear hunters are required to contact the Arizona Game and Fish Department within 48 hours of killing a bear. Within 10 days of killing a bear, successful hunters must physically present the bear skull, hide, and attached proof of sex for inspection, and provide a premolar tooth (Arizona Game and Fish Department 2006). Data collected by biologists will include age, sex, reproductive status, hunting method, harvest date and location, condition of animals, evidence of disease, hunter effort, and presence of biological markers, tattoos, and radio transmitters.



Black bears are omnivorous (they eat both plants and animals). More than 75% of their diet is comprised of vegetation.

Harvest

Success of black bear hunters varies with weather conditions and characteristics of populations. Harvest varied from 131 to 313 bears between 1964 and 1980 (LeCount 1981). Harvested male bears averaged 6.8 years of age, 33% of males were more than 7 years old, and mean age of female bears harvested was 6.3 years. Sport harvest of black bears in Arizona between 1990 and 2002 averaged about 188 per year, and about 11 bears per year were killed for depredation or died in vehicle accidents, captures, and illegal harvests. Most (57.6%) black bears harvested in Arizona between 1998 and 2002 were killed in areas located primarily south of the Mogollon Rim, and most (55%) were harvested in September.

Currently, surveys of black bear hunters in Arizona are conducted to assess hunter demographics. Data from 1995, 1998, and 2001 indicated fairly consistent hunter effort, with greatest hunter effort in areas where harvest was highest. Most hunters harvest bears using a combination of techniques, including searching with binoculars, calling, stalking, and hunting with the aid of dogs.

Mean age of male and female black bears harvested in Arizona from 1998 to 2002 was 5.9 and 6.5 years, respectively; about 21% were more than 10 years old. An average of 38% was classified as subadults. Breakdown of age classes by year suggests a declining trend in recruitment, probably caused by drought conditions. Between 1998 and 2000, black bear cubs and yearlings comprised 10% and 42%, respectively, of total harvest. In comparison, cubs and yearlings comprised only 3% of harvest in 2001 and 2002, indicating a cohort was missing. These values, along with low percentage of subadults in 2002 (22%), indicated recruitment or survival of cubs has been low since 2000.

Inferring population trends from harvest data alone can fail to detect declining or increasing population trends (Bunnell and Tait 1981, Miller 1990, Garshelis 1991). A stable age and sex composition in both the live population and the harvest can occur in stable and declining populations (Miller 1990). A predominantly male harvest has been documented in a chiefly female population (Bunnell and Tait 1981). Harvest rate can be constant in a declining population, particularly if hunter density increases (Bunnell and Tait 1981, Garshelis 1991). Capture or harvest data indicate that lightly exploited populations generally have fewer than 30% subadults and a large proportion of males greater than 8 years old (Bunnell and Tait 1981, LeCount 1982, Cunningham et al. 2001). Reduced numbers of subadults in Arizona during 2002 probably was caused by a lack of reproduction.

Effects of Drought

Given potential effects of differences in mast production on movements and reproduction of black bears, drought might influence relative abundance. However, mast production failures often do not correlate directly with precipita-

tion. Several climatic factors and attributes of plants, such as age, can control production of oak or juniper mast (Harper et al. 1985). Date of last frost has been correlated with production of juniper mast (Costello et al. 2001), but no predictive relationship has been established.

Drought in Arizona in 2000 likely affected movements of black bears (Cunningham et al. 2001), which in turn influenced harvest of black bears and abandonment and movement of at least 20 cubs into the urban area of Phoenix.

Although precipitation is the primary factor influencing production of acorns (Harper et al. 1985), harvest of black bears in Arizona was not correlated with precipitation between 1996 and 2002. If inadequate precipitation was the primary predictor of increased movements and harvest of black bears, the 2002 harvest should have exceeded harvest in other years. Only complete failures of mast production seem to cause the longest movements, and surveys of mast and prickly pear cactus fruit production, not measurements of precipitation or temperature, will allow development of useful predictive modeling (Costello et al. 2001). Better analyses can be achieved regarding effects of drought on reproduction as more data are obtained on successful rearing of cubs.

Habitat Fragmentation and Fire

Home ranges of black bears in logged areas of ponderosa pine with less than 6.5 vertical feet of cover often are 3 to 5 times greater than those in denser mixed conifer or Madrean evergreen woodland habitats (Mohr 1947, Mollohan 1985, LeCount and Yarchin 1990). Fragmentation of habitat by logging or fires makes it difficult for bears to travel to important food patches, increases vulnerability to hunters and other bears, and lowers overall bear density (Mollohan 1985). It is increasingly important to identify and preserve travel corridors; loss of travel corridors might reduce ability of subadult females to refill vacant territories (Mollohan 1985). Fragmentation of habitats also might increase vulnerability of bears to hunting by concentrating them and allowing easier access by hunters.

Black bears occupied primarily unburned islands of vegetation after a large-scale wildfire. Shift in local distribution resulted in post-fire density 3 times that of pre-fire levels, a male-dominated sex ratio, and no survival of cubs (Mollohan 1985, Mollohan and LeCount 1989, Cunningham and Ballard 2004). Fire may fragment habitats of contiguous populations of black bears in Arizona.

Research Needs

Forest Restoration

Dense stands of ponderosa pine are more prevalent today than during pre-settlement times (Cooper 1960). A broad scientific, social, and political consensus recently emerged, stating restoration of ecological function in southwestern pine forests is necessary and urgent (Covington and Moore 1994, Covington

et al. 1997, Allen et al. 2002). Restoration treatments were proposed for more than 200,000 acres annually in Arizona and New Mexico (Anonymous 2001). Numerous treatments and techniques are planned to reduce densities of trees and shrubs, from controlled burning to mechanical manipulations. Restoration treatments could negatively affect black bear habitats and increase fragmentation if more open habitat is created because of bear dependence on shrubs and large trees (Mollohan 1985, LeCount and Yarchin 1990). Research is needed to determine patch size, shape, and the number of patches necessary to maintain suitable black bear habitat while reducing fire risk.

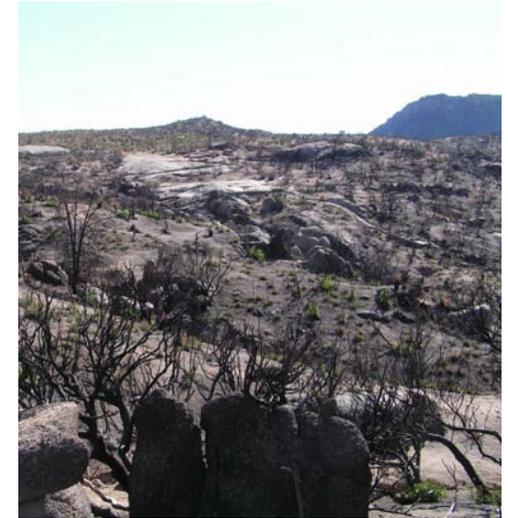
Monitoring Populations

A cost-effective method is needed to monitor black bear populations. Mark-recapture indices using hair snares to identify specific individuals without expensive capture operations might be possible, but research is needed to evaluate sampling costs, estimate adequate sample sizes, and to assess overall utility of this approach. Research also is needed to evaluate use of biomarkers such as tetracycline.

Low rates of birth or survival of black bear cubs correspond strongly with failures of mast production (Costello et al. 2003). Surveys of mast production could be conducted and have predictive value for populations of black bears. Data on production of mast improved New Mexico's population model (Costello et al. 2001, 2003), particularly in identifying years of poor reproduction and potential effects on future harvests. Determining availability of mast could help explain missing cohorts and identify areas of concern.

Geographic Information Systems (GIS) Technology

Based on vegetation data from New Mexico's Gap Analysis Project (GAP), there is an estimated 22,756 square miles of suitable habitat for black bears in that state (Costello et al. 2001). Using density data, the statewide population in New Mexico likely is about 5,947 bears more than 1 year old, almost



Fires can enhance wildlife habitat if burned areas are not large and are interspersed with unburned areas, creating a diversity of food and habitat.

double the previous estimate of 3,000. New estimates of population size do not reflect an actual increase in number of bears, but reflect a more efficient procedure of estimation based on remote sensing (Costello et al. 2001).

As of 1999, the population of black bears in Arizona was estimated at 2,500 adults occupying 12,598 square miles of habitat. Remote sensing data, based on new vegetation maps and information specific to management units, ostensibly would yield a more accurate estimate of the population and provide more useful information to wildlife managers. Presently, Arizona GAP vegetation data are being ground-checked and are not yet available (S. Boe, Arizona Game and Fish Department, personal communication).

Literature Cited

- Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecological Applications* 12:1418–1433.
- Anonymous. 2001. Proposed ten year federal/state/tribal/local partnership programs for restoration of Arizona/New Mexico urban interface and wildland forest areas. Prepared for the Western Governor's Association, Phoenix, Arizona.
- Arizona Game and Fish Department. 2000. *Wildlife 2006*. Arizona Game and Fish Department, Phoenix.
- Arizona Game and Fish Department. 2003. *Hunt Statistics 2003*. Arizona Game and Fish Department, Phoenix.
- Arizona Game and Fish Department, 2006. *Arizona Hunting and Trapping Regulations*. Arizona Game and Fish Department, Phoenix.
- Beck, T. D. I. 1991. Black bears of west-central Colorado. Colorado Division of Wildlife Technical Publication No. 39, Denver.
- Beecham, J. J. 1980. Some population characteristics of two black bear populations in Idaho. Pages 201–204 in C. J. Martinka and K. L. McArthur, Eds. *Bears—Their biology and management*. Bear Biology Association Conference, Series 3.
- Beeman, L. E., and M. R. Pelton. 1976. Homing of black bears in the Great Smokey Mountains National Park. International Conference on Bear Re-

search and Management 3:87–95.

- Bunnell, F. L., and D. E. N. Tait. 1981. Population dynamics of bears—implications. Pages 75–98 in C. W. Fowler and T. D. Smith, Eds. *Dynamics of Large Mammal Populations*. John Wiley and Sons, New York, New York.
- Bunnell, F. L., and D. E. N. Tait. 1985. Mortality rates of North American bears. *Arctic* 38:316–323.
- Carrel, W. K. 1987. Reproductive history of female black bears from dental cementum. International Conference on Bear Research and Management 9:205–212.
- Cooper, C. F. 1960. Changes in vegetation, structure and growth of southwestern pine forests since European settlement. *Ecological Monographs* 30:129–164.
- Costello, C. M., D. E. Jones, K. A. Green-Hammond, R. M. Inman, K. H. Inman, B. C. Thompson, T. A. Deitner, and H. B. Quigley. 2001. A study of black bear ecology in New Mexico with models for population dynamics and habitat suitability. Final Report, Federal Aid in Wildlife Restoration Project W-131-R, New Mexico Department of Game and Fish, Santa Fe.
- Costello, C. M., D. E. Jones, R. M. Inman, K. H. Inman, B. C. Thompson, and H. B. Quigley. 2003. Relationship of variable mast production to American black bear reproductive parameters in New Mexico. *Ursus* 14:1–16.
- Covington, W. W., and M. M. Moore. 1994. Southwestern ponderosa pine forest structure: changes since Euro-American settlement. *Journal of Forestry* 92:39–47.
- Covington, W. W., P. Z. Fulé, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. *Journal of Forestry* 95:23–29.
- Cowan, I. M. 1938. Geographic distribution of color phases of the red fox and black bear in the Pacific Northwest. *Journal of Mammalogy* 19:202–206.
- Cowan, I. M. 1972. The status and conservation of bears (*Ursidae*) of the world—1970. International Conference on Bear Research and Management 2:343–367.
- Cunningham, S. C., L. M. Monroe, L. Kirkendall, and C. L. Ticer. 2001. Effects

Biological Foundations for Management of Mammalian Predators in Arizona

of the catastrophic Lone Fire on low, medium, and high mobility wildlife species. Technical Guidance Bulletin No. 5. Arizona Game and Fish Department, Phoenix.

- Cunningham, S. C., and W. B. Ballard. 2004. Effects of wildfire on black bear demographics in central Arizona. *Wildlife Society Bulletin* 32:928–937.
- Doan-Crider, D. L., and E. C. Hellgren. 1996. Population characteristics and winter ecology of black bears in Coahuila, Mexico. *Journal of Wildlife Management* 60:398–407.
- Eiler, J. H., W. G. Wathen, and M. R. Pelton. 1989. Reproduction in black bears in the southern Appalachian Mountains. *Journal of Wildlife Management* 53:962–968.
- Elowe, K. D., and W. E. Dodge. 1989. Factors affecting reproductive success and cub survival. *Journal of Wildlife Management* 53:962–968.
- Garshelis, D. L. 1991. Monitoring effects of harvest on black bear populations in North America: a review and evaluation of techniques. *Proceedings of the Eastern Workshop on Black Bear Research and Management* 10:120–144.
- Harper, K. T., F. J. Wagstaff, and L. M. Kunzler. 1985. Biology and management of the Gambel Oak vegetative type: A literature review. General Technical Report INT-179. Intermountain Forest and Range Experiment Station, Ogden, Utah.
- Herrero, S. 1972. Aspects of evolution and adaptation in American black bear (*Ursus americanus Pallas*) and brown and grizzly bears (*Ursus arctos Linné*) of North America. Pages 221–231 in S. Herrero editor. *Bears—Their biology and management*. Bear Biology Association Conference Series 2.
- Hellgren, E. C. 1998. Physiology of hibernation in bears. *Ursus* 10:467–477.
- Horn, H. S. 1968. The adaptive significance of colonial nesting in the Brewer's Blackbird (*Euphagus cyanocephalus*). *Ecology* 49:692–694.
- Jonkel, C. J., and I. M. Cowan. 1971. The black bear in the spruce-fir forest. *Wildlife Monograph* 27.
- Jonkel, C. 1978. Black, brown (grizzly), and polar bears. Pages 227–248 in *Big game of North America ecology and management*. J. L. Schmidt and D. L.

Black Bear

Gilbert Eds. Stackpole Books, Harrisburg, Pennsylvania.

- Kemp, G. A. 1976. The dynamics and regulation of black bear, *Ursus americanus*, populations in northern Alberta. Pages 191–197 in M. R. Pelton, J. W. Lentfer, and G. E. folks, editors. *Bears—Their biology and management*. IUCN Publication No. 23, Morges, Switzerland.
- LeCount, A. L. 1981. An analysis of the black bear harvest in Arizona (1968–1978). Arizona Game and Fish Department Special Report No. 12, Phoenix.
- LeCount, A. L. 1982. Characteristics of a central Arizona black bear population. *Journal of Wildlife Management* 46:861–868.
- LeCount, A. L. 1983. Evidence of wild black bears breeding while raising cubs. *Journal of Wildlife Management* 47:264–268.
- LeCount A. L., R. H. Smith, and J. R. Wegge. 1984. Black Bear habitat requirements in central Arizona. Federal Aid Final Report. Arizona Game and Fish Department, Phoenix.
- LeCount, A. L. 1987a. Causes of black bear cub mortality. *International Conference of Bear Research and Management* 7:75–82.
- LeCount, A. L. 1987b. Characteristics of a northern Arizona black bear population. Final Report, Federal Aid in Wildlife Restoration Project W-78-R, Arizona Game and Fish Department, Phoenix.
- LeCount, A. L. 1990. Characteristics of an east-central Arizona black bear population. Arizona Game and Fish Department Technical Report 2, Phoenix.
- LeCount, A. L., and J. C. Yarchin. 1990. Black bear habitat use in east-central Arizona. Arizona Game and Fish Department Technical Report No. 4, Phoenix.
- Lindzey, F. G., and E. C. Meslow. 1977. Population characteristics of black bears on an island in Washington. *Journal of Wildlife Management* 41:408–412.
- Mace, R. D., S. C. Minta, T. L. Manley, and K. E. Aune. 1994. Estimating grizzly bear population size using camera sightings. *Wildlife Society Bulletin* 22:74–83.

Biological Foundations for Management of Mammalian Predators in Arizona

- McArthur, K. L. 1981. Factors contributing to effectiveness of black bear transplants. *Journal of Wildlife Management* 45:102–110.
- Miller, S. D., G. C. White, R. A. Sellers, H. V. Reynolds, J. W. Schoen, K. Titus, V. G. Barnes Jr., R. B. Smith, R. R. Nelson, W. B. Ballard, and C. C. Schwartz. 1997. Brown and black bear density estimation in Alaska using radio telemetry and replicated mark-resight techniques. *Wildlife Monographs* 133.
- Miller, S. D. 1990. Impact of increased bear hunting on survivorship of young bears. *Wildlife Society Bulletin* 18:462–467.
- Miller, S. D. 1994. Black bear reproduction and cub survivorship in south-central Alaska. *International Conference of Bear Research and Management* 9:263–273.
- Mohr, C. O. 1947. Table of equivalent populations of North American small mammals. *American Midland Naturalist* 37:223–249.
- Mollohan, C. M. 1985. Adult Female Black Bear Habitat Use in Northern Arizona. MS Thesis. Arizona State University, Tempe.
- Mollohan, C. M., and A. L. LeCount. 1989. Problems of maintaining a viable black bear population in a fragmented forest. Conference on Multi-resource Management of Ponderosa Pine Forests. Proceedings of the symposium; 1989 November 14–16; Flagstaff, AZ. General Technology Report RM-185. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.
- Nelson, R. A. 1980. Protein and fat metabolism in hibernating bears. *Federation Proceedings* 39:2955–2958.
- Noyce, K. V., and D. L. Garshelis. 1994. Body size and blood characteristics as indicators of condition and reproductive performance in black bears. *International Conference on Bear Research and Management* 9:481–496.
- Pelton, M. R. 1982. Black Bear (*Ursus americanus*). Pages 504–514 in J. A. Chapman and G. A. Feldhammer, editors. *Wild mammals of North America: biology, management, and economics*. John Hopkins University Press, Baltimore, Maryland.
- Piekielek, W., and T. S. Burton. 1975. A black bear population in northern California. *California Fish and Game* 61:4–25.

Black Bear

- Reynolds, D. G., and J. J. Beecham. 1980. Home range activities and reproduction of black bears in west-central Idaho. *International Conference on Bear Research and Management* 4:181–190.
- Rogers, L. L. 1976. Effects of mast and berry crop failure on survival, growth, and reproductive success of black bears. *Transactions of the North American Wildlife and Natural Resources Conference* 41:431–438.
- Rogers, L. L. 1977. Social relationships, movements, and population dynamics of black bears in northeastern Minnesota. Ph.D. Thesis, University of Minnesota, Minneapolis.
- Rogers, L. L. 1986. Homing by radio-collared black bears, *Ursus americanus*, in Minnesota. *Canadian Field-Naturalist* 100:350–353.
- Samson, C., and J. Huot. 1995. Reproductive biology of female black bears in relation to body mass in early winter. *Journal of Mammalogy* 76:68–77.
- Sauer, P. R., S. Free, and S. Browne. 1969. Movement of tagged black bears in the Adirondacks. *New York Fish and Game Journal* 16:205–223.
- Schwartz, C. C., and A. W. Franzmann. 1991. Interrelationship of black bears to moose and forest succession in the northern coniferous forest. *Wildlife Monographs* 113.
- Schwartz, C. C., and A. W. Franzmann. 1992. Dispersal and survival of subadult black bears from the Kenai Peninsula, Alaska. *Journal of Wildlife Management* 56:426–431.
- Stringham, S. F. 1990. Grizzly bear reproductive rate relative to body size. *Proceedings International Conference on Bear Research and Management* 5:140–151.
- Waddell, T., and D. E. Brown. 1984. Weights and color of black bears in the Pinaleno Mountains, Arizona. *Journal of Mammalogy* 65:350–351.
- Wiens, J. A. 1976. Population responses to patchy environments. *Annual Review of Ecology and Systematics* 7:81–120.

Chapter 2

Bobcat

Ted McKinney, Research Branch
Arizona Game and Fish Department

Common name- Bobcat
Scientific name - *Lynx rufus*

Life History

Description

Arizona's most abundant wild feline, the bobcat (*Lynx rufus*) is one of the smallest members of the cat family in North America. Adults of these medium-sized predators in Arizona range from 24 to 30 inches long, and weigh from 12 to 30 pounds. Throughout North America, males weigh between about 14 and 57 pounds, and females weigh between about 8 and 33 pounds (Young 1958, McCord and Cardoza 1982, Anderson and Lovallo 2003). Average weight for sexes combined is about 13.6 pounds (Gittleman 1985).

Bobcats generally are reddish to yellowish brown with black or dark brown streaks or spots. They have white underparts, and the inside of the forelegs is marked with several black bars. Ears are prominent, with a tuft of black hair at tips. Perhaps the most distinguishing physical characteristic is a short, stubby tail, which is no more than about 5 inches long and always less than one quarter of the length of head and body (McCord and Cardoza 1982, Arizona Game and Fish Department 2003). Although bobcats seldom are seen, tracks commonly are found and are distinctive. Well-defined tracks of adults in dirt or dust often show 4 toes with no claw marks. Tracks are about 1.9 inches wide by 1.7 inches long. Anterior borders of heel pads are two-lobed, and posterior borders are three-lobed (Murie 1974).



Bobcat tracks show no claw marks and are more rounded than tracks from members of the dog family. The anterior or front portion of the heel pad has two lobes instead of the single lobe found in the dog family.

Distribution

Bobcats inhabit only North America, throughout the United States, most of Mexico, and into southern Canada, and usually are most abundant below about 6,000 feet



elevation (Young 1958, McCord and Cardoza 1982, Anderson 1987). Bobcats are distributed throughout Arizona (Figure 1). Estimating densities of resident bobcats is difficult, even on small areas, and can be achieved with extensive effort, but estimating densities of transients is more difficult (McCord and Cardoza 1982).

Bobcat densities in the Southwest and West have ranged from about less than 1 to 5 per square mile, depending on categories measured, e.g., total population, adults, or residents (McCord and Cardoza 1982, Anderson 1987).

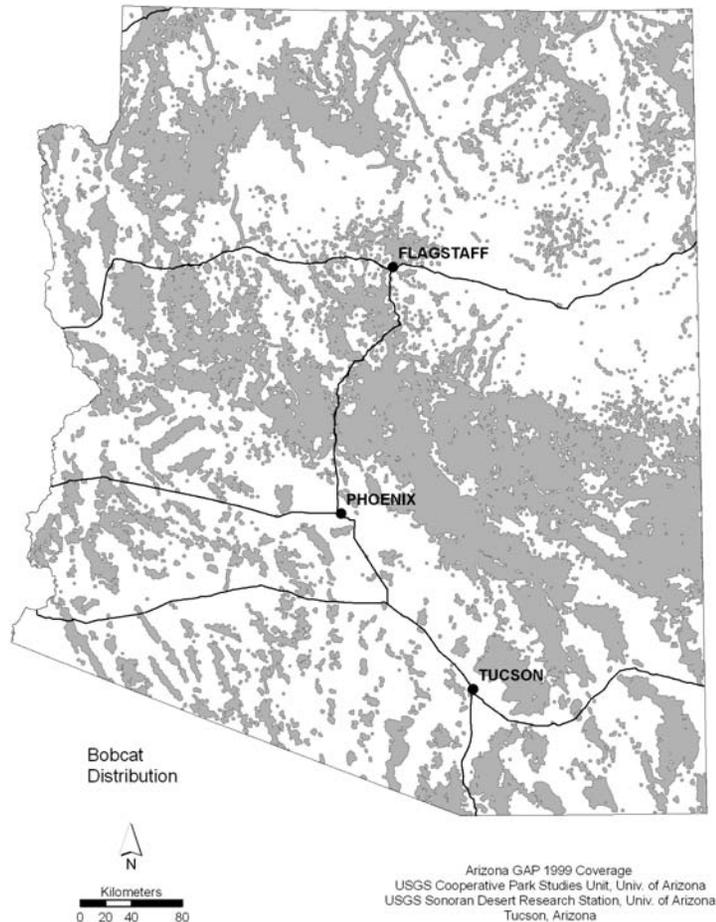


Figure 1. Distribution of bobcats in Arizona.

Minimum densities on Three Bar Wildlife Area in central Arizona ranged from 0.15 to 0.17 bobcats per square mile (Jones and Smith 1979). Scent-station transects, which use tracking bed stations about 3 feet diameter cleared of debris, smoothed, and covered with a thin bed of lime or sifted dirt, indicate relative abundance. Scent-station surveys potentially provide a method for monitoring long-term trends of populations of bobcats (Brady 1979, Hon 1979). Counts of scats along transects also might indicate change in abundance of populations (Conner et al. 1983). One unpublished research project estimated densities of bobcats by comparing catch rates by trappers between areas with known sizes of bobcat populations.

Fossil remains of bobcats have been found in Pleistocene deposits in southern North America (Hemmer 1976). Archaeologists also have found bone remains of bobcats in Native American ruins, indicating their presence in the Southwest more than 2,000 years ago (Young 1958).

Habitat

Bobcats inhabit widely varying terrain and vegetation types throughout their range of distribution, including swamps, deserts, and mountain ranges (Young 1958, McCord and Cardoza 1982, Arizona Game and Fish Department 2003). Bobcats tend to avoid open areas, and generally prefer mixed shrub and forested habitats. In Arizona and the West, bobcats tend to inhabit broken, brushy country. Abundance of bobcats in Arizona might be highest in Sonoran Desert scrub and Great Basin conifer woodland habitats (Anonymous 1982). However, highest densities in the West likely occur in regions of chaparral and desert scrub/grassland habitat (Anderson 1987). Recent studies (T. McKinney, Arizona Game and Fish Department, unpublished data) suggest abundance in Arizona might be higher in interior chaparral than other habitats. Bobcats avoided large, open areas such as frozen lakes and bogs, and selected mature softwood and mixed forest areas in western Maine (Major and Sherburne 1987).



Bobcat habitat in dense chaparral vegetation. Bobcats prefer mixed shrub and forested habitats versus open areas.

Bobcats in Oregon inhabited forested and open habitats in proportion to availability, and occupied areas with forest cover more during the day (Witmer

and deCalesta 1986). Bobcats in Idaho commonly were associated with areas where stalking cover was present, and most often occupied rocky and timbered terrain (Koehler and Hornocker 1989, 1991). Bobcats used open areas more than timbered terrain in winter and timbered habitats more than open terrain in summer, and spent only about 10% of the time in alpine areas (Koehler and Hornocker 1991).

Diets and Prey

Little research has focused on bobcats and their principal prey in Arizona, but bobcats primarily eat small prey (Rosenzweig 1966, Beasom and Moore 1977, Leopold and Krausman 1986, Witmer and deCalesta 1986, Major and Sherburne 1987). Life histories of cottontail rabbits (*Sylvilagus* spp.) in Arizona are poorly understood, and only 1 study of desert cottontails (*S. audubonii*) has been conducted in the state (Arizona Game and Fish Department 2003). Bobcats, antelope jackrabbits (*Lepus alleni*), black-tailed jackrabbits (*L. californicus*), and cottontail rabbits co-exist on desert mountain ranges and bajadas (outwash slopes with long straight profiles) in the Southwest. These species of rabbits might occupy different types or portions of habitats, and landscape management practices might differentially affect abundance (Brown and Krausman 2003), possibly influencing abundance and diets of bobcats.

Rainfall in deserts, particularly during winter, influences primary production and relative availability of small prey (Beatley 1969, Reichman and Van De Graaff 1975, Jones and Smith 1979). Availability of prey influences diets of bobcats (Beasom and Moore 1977, Jones and Smith 1979, Leopold and Krausman 1986). Based on analyses of stomach contents, diets of bobcats in Texas varied with availability of prey; scarcity of prey was associated with a more varied diet (Beasom and Moore 1977). Dynamics and trends of populations of bobcats and predator-prey relationships, and implications of these variables for management are poorly understood.

Analysis of 3,538 stomachs of bobcats collected in 30 states (70% were from 15 western states) indicated rodents and rabbits provided 46% and 45% of diets, respectively (Young 1958). Remains of rabbits occurred more often than remains of rodents in scats (feces) of bobcats in Utah/Nevada (Gashwiler et al. 1960), and rabbits seem to dominate diets of bobcats over broad regions of North America (McCord and Cardoza 1982). However, rodents appear to be more predominant than rabbits in diets of bobcats in the southwestern and western United States (Delibes et al. 1997, Anderson and Lovallo 2003).

Although rabbits are thought to be the principal prey of bobcats in Arizona (Arizona Game and Fish Department 2003), this can vary. Based on analysis of scats, small prey (rabbits, rodents, birds, reptiles) comprised from 82% to 98% of diets of bobcats in the Mazatzal Mountains of central Arizona between 2000 and 2003; rodents comprised 58% to 78% of diets, whereas rabbits com-

prised 9% to 21% of diets (McKinney and Smith 2007). Other studies based on analysis of scats from the same area found rodents comprised 46% to 67% of diets of bobcats, whereas rabbits comprised 38% to 40% of diets (Jones and Smith 1979, Cunningham et al. 2001). In general, occurrence of rabbits in diets of bobcats has differed among studies, and predation likely varies with availability of prey (Delibes and Hiraldo 1987, Delibes et al. 1997, Baker et al. 2001).

Bobcats are strict carnivores and occasionally prey on domestic sheep and goats, deer (*Odocoileus* spp.), and pronghorn (*Antilocapra americana*), but more commonly kill smaller prey. Among remains of ungulates in bobcat scats in Arizona, deer predominated (0% to 8% occurrence), whereas no remains of desert bighorn sheep (*Ovis canadensis*) were identified, and occurrence of remains of cattle and javelina (collared peccary; *Pecari tajacu*) each comprised from 0% to less than 2% (McKinney and Smith 2007). In contrast to results of studies in Arizona (Jones and Smith 1979, McKinney and Smith 2007), remains of ungulates in bobcat scats in Idaho occurred more often during winter than other seasons (Koehler and Hornocker 1991). When mule deer (*O. hemionus*) declined in Texas (Leopold and Krausman 1986), occurrence of remains of rabbits in scats of bobcats tended to increase, whereas occurrence of remains of mule deer and javelina tended to decrease. In general, bobcats in the Southeast and Southwest likely tend to eat less ungulates than do bobcats in more northern latitudes of North America (Anderson and Lovallo 2003).

Bobcats also can be scavengers. In Idaho, bobcats scavenged 5% of deer carcasses (Koehler and Hornocker 1991), and scavenged kills made by mountain lions (*Puma concolor*) but not kills made by coyotes (*Canis latrans*). In comparison, bobcats scavenged carcasses of white-tailed deer (*O. virginianus*) possibly killed by coyotes in western Maine (Major and Sherburne 1987).

Visual examination of remains of prey in scats has been used widely in field studies of diets of predators, but visual identification of bobcat feces might not always be conclusive (Johnson et al. 1984). Recent studies suggest removal of carnivore scats by local animals might bias diet studies based on fecal analysis (Sanchez et al. 2004, Livingston et al. 2005). Studies of digestion and metabolism potentially contribute to understanding nutritional ecology of bobcats (Golley et al. 1965, Johnson and Aldred 1982, Mautz and Pekins 1989).

Disease

Diseases in populations of wild bobcats are poorly understood. No die-offs of populations have been associated with epizootics or parasite infestations. Nonetheless, 12 infectious diseases might produce lesions or antibodies in wild bobcats, including feline distemper, leukemia, brucellosis, and rabies. Bobcats are susceptible to various internal and external parasites (Samuel et al. 2001), and possibly to infections of anthrax, parvovirus, plague, and vesicular stomatitis (Williams and Baker 2001). Bobcats likely are a definitive host for at least 13 par-

asite species, and deer, mice, and rabbits are alternative hosts for several of these parasites. Samples of blood, lung, and muscle tissues of bobcats demonstrated high incidences of parasitism (McCord and Cardoza 1982).

Reproduction

Female bobcats breed between 1 and 2 years of age, and males produce few viable sperm before they are 2 years old. Bobcats breed with more than 1 partner, and might mate during any month. Courtship behavior is complex and involves chasing, running, ambushing, and bumping before and after multiple copulations (Rolley 1987). Females ovulate seasonally, breeding seems to be most intensive during late winter to spring, and some females might reproduce in alternate years. Breeding season in Arizona is poorly documented, but males might be reproductively active throughout the year (Young 1958). Although uncertain, bobcats might be induced ovulators (the female does not ovulate until bred). In contrast, results of most recent studies suggest spontaneous ovulation occurs in bobcats. Females experience estrous or heat cycles up to 3 times per season if they are not fertilized, and ovulation rates might range from 3.4 to 5.5 ova.

Bobcats generally have only 1 litter per year (McCord and Cardoza 1982, Anderson 1987). Gestation lasts 50 to 70 days, litters are born mainly during spring to early summer, size of litters averages about 2.6 kittens (range = 1 to 4), and sex ratios (male:female) at birth generally are 1:1 (McCord and Cardoza 1982, Anderson 1987). Litter size might increase slightly with increased availability of prey. Kittens weigh about 0.66 pounds at birth, and by autumn of the year of birth



Bobcats are territorial, using urine, feces, and anal gland secretions to delineate home ranges. A successful male's home range overlaps with those of several females, and may also overlap the territory of another male. Female home ranges are generally smaller, and do not overlap one another.

Bobcat

can weigh between about 9 and 15 pounds (Young 1958, McCord and Cardoza 1982).

Mothers rear kittens alone and suckle them for 2 to 3 months, but kittens might remain with their mother for nearly a year (Young 1958, Towell 1979, McCord and Cardoza 1982, Arizona Game and Fish Department 2003). Bobcat kittens seem to experience high rates of natural mortality (Anderson 1987). Survival of kittens (birth to withdrawal of maternal support) and juveniles (withdrawal of maternal support to establishment of a home range) depends primarily on availability of food resources (McCord and Cardoza 1982, Anderson 1987). Most bobcats in free-ranging populations likely do not survive beyond about 3 years of age (McCord and Cardoza 1982).

Behavior

Hunting.—Methods of hunting vary considerably, but bobcats often stop, sit, and wait if hunting prey such as rabbits, rodents, and squirrels. Conversely, bobcats move more frequently if hunting larger prey, such as deer, and generally attempt to locate bedded animals (McCord and Cardoza 1982). Bobcats tend to kill adult deer by biting the trachea when deer are lying down, and generally kill fawns by biting the head or neck. Bobcats tend to consume mainly hindquarters, neck, and shoulder regions of deer they have killed, but they rarely eat rumens (stomachs and contents). Bobcats may cover kills of large prey with debris gathered from areas near a carcass (Dolbeer et al. 1994). Bobcats might prey on young javelinas (Knipe 1956), but low incidence of remains in samples of stomach contents of bobcats suggests javelinas are not major prey (Day 1985). Similarly, remains of javelinas occurred in less than 2% of 320 bobcat scats in the Mazatzal Mountains of central Arizona (McKinney and Smith 2007).

Activity.—Bobcats are solitary predators, but diurnal and intraspecific patterns of behavior are not well understood. Direct social interactions likely are infrequent, and individuals tend to avoid direct interaction with other bobcats (intraspecific aggressive behavior). Bobcats may tend to be most active at night (Anderson 1987). However, male and female bobcats might exhibit similar patterns of activity, and tend to be most active in evening and morning, and least active during mid-day (Witmer and deCalesta 1986). In contrast, some studies suggest daily activities of bobcats lack any particular pattern or rhythm (Gittleman 1985).

Home Range.—Home ranges of bobcats reportedly vary widely in size from about 1 to 126 square miles, depending on sex, age, density of populations, density of prey, season, and method of survey (Bailey 1974, McCord and Cardoza 1982, Anderson 1987). Smallest home ranges in the West likely correspond with highest densities in mixed chaparral/desert grassland habitats (Anderson 1987). Home ranges tend to be smaller in exploited than in unexploited populations, and tend to be larger in more open, flatter regions. Home ranges of females are

smaller than those of males, and might be smaller in winter, possibly related to breeding and denning activities, whereas home ranges of males might increase in size during winter and the spring breeding season (Witmer and deCalesta 1986, Anderson 1987).

Home ranges of male and female bobcats in central Arizona were similar in size, and ranged between about 1.2 and 2.5 square miles (Lawhead 1984). Six male bobcats in Maine had an average home range of 88 square miles, whereas 1 female had a home range of 17 square miles (Major and Sherburne 1987). Average home ranges for 5 females and 1 male in Oregon were about 0.6 and 1.2 square miles, respectively (Witmer and deCalesta 1986). Bobcats with adjoining or overlapping home ranges tend to maintain random spacing (separation distances between simultaneously monitored animals were neither greater nor less than expected).



Bobcats can be found in riparian areas of chaparral vegetation. Chaparral vegetation burns frequently and is comprised of woody plants that grow well after fires.

Older bobcats of both sexes usually have fairly well-defined territories that vary in size depending on density of prey, sex, season, and climate. Core areas might be shared among resident animals, but residents almost always exclude transients. Some studies reported bobcats avoid one another (Witmer and deCalesta 1986), and aggressive behavior between residents and transients has not been observed (McCord and Cardoza 1982). A dominance hierarchy based on a land tenure system might function most effectively on transient animals (Bailey 1974, Hornocker and Bailey 1986). In general, social structure of bobcats might involve resident females and males occupying fixed home ranges, and transient animals perhaps associated with resident populations (McCord and Cardoza (1982). Home ranges of females tend not to overlap, but home ranges of males likely overlap each other and overlap home ranges of females (Anderson 1987). Social structure of bobcats seems to be maintained, in part, by a complex system of scent marking involving urine, feces, anal glands, and enhancement of excretions by scraping with the feet (McCord and Cardoza 1982).

Dispersal.—Transient, dispersing bobcats likely are young or sexually immature, and they can make erratic local or long-range movements. Dispersal

of young bobcats might occur primarily between fall and late winter. Transients have been observed to move more than 25 miles, but residents also might move long or short distances on a daily basis. One study found 92% of marked bobcats were recaptured within 10 miles of original capture site (McCord and Cardoza 1982). Young bobcats in south-central Idaho dispersed into suitable habitat “far” from the natal area (Hornocker and Bailey 1986). Adult females and males moved average minimum distances ranging from 0.6 to 1.6 and 1.2 to 3.0 miles/day, respectively (Witmer and deCalesta 1986). Maintaining suitable travel corridors (Beier and Loe 1992, Beier 1993) might be an important element of conservation and management of bobcats.

Competition with Coyotes.—Competition between bobcats and coyotes has been discussed widely among researchers, but little research has addressed this issue (McCord and Cardoza 1982), and the topic remains controversial. Competition between the species might be minimal in some cases (Witmer and deCalesta 1986), but substantial in some ecosystems (Anderson and Lovallo 2003). Bobcats and coyotes likely partition food resources in central Arizona’s Sonoran Desert, suggesting little competition for food (McKinney and Smith 2007). Bobcats were more specialized in their diets (diet diversity index range = 1.79 to 2.47) than were coyotes (diet diversity index range = 4.96 to 5.57), and overlap of diets between the species was moderate (diet overlap index range = 0.44 to 0.53).

In contrast, abundance of bobcats in eastern Maine declined concurrent with an increase in abundance of coyotes, and coyotes possibly reduced carrying capacity of bobcats by diminishing availability of prey (Litvaitis and Harrison 1989). Experimental removal of coyotes in Texas resulted in higher abundance of bobcats and common gray foxes (*Urocyon cinereoargenteus*) on treatment sites (Henke and Bryant 1999).

Survival

Comparatively little is known regarding the lifespan of free-ranging bobcats, but captive bobcats have lived up to 23 years (Young 1958, Anderson 1987). Based on analysis of cementum annuli in teeth (Crowe 1972), maximum ages of bobcats harvested in Wyoming ranged from 16 to 17 years (M. Zornes,



Coyotes may compete with bobcats for food or space in some areas, but more than likely share natural resources with bobcats in Arizona’s deserts.

Arizona Game and Fish Department, unpublished data). About 80% of bobcats in free-ranging populations likely are less than 3 years old, and composition of this age group has ranged from 54% to 87% (Crowe 1975, Crowe and Strickland 1975, Johnson et al. 1981, Knick et al. 1985, Rolley 1985).

Management and Conservation

Bounties were paid intermittently for bobcats in some areas of the United States as early as 1727. By the 1940s and 1950s, bounties were considered an ineffective management tool and were terminated in most states (Young 1958). Efforts by government agencies to control predators on the Kaibab Plateau of northern Arizona from 1906 to 1963 focused on removal of coyotes and mountain lions to benefit abundance of mule deer, but resulted in incidental harvest of 1,550 bobcats (McCulloch 1986).

Management of bobcats relies on assessment of indirect indices of abundance to evaluate range of distribution, selection of habitats, size and structure of populations, and effects of harvest. Spatial distribution or range of bobcats might be monitored using several habitat-specific indices or methods, including: 1) locations of harvest, 2) scent-station surveys, 3) monitoring radio collared animals, and 4) observations of sign such as scats and tracks. Chemical immobilization using projected darts or syringes likely provides the best method for capturing bobcats and placement of radio collars, but care must be taken when darting at close ranges (Pond and O'Gara 1994).

Indices related to population dynamics of bobcats have received considerable attention in recent years, including sex and age structure, recruitment, and rates of mortality. Annual harvest should be tempered with measures of habitat-specific harvests and hunter effort. Key elements to be identified before implementing management practices are: 1) direction of population change, if any, 2) rate of population change, and 3) factors affecting population change (McCord and Cardoza 1982).



Arizona Game and Fish Department personnel examine a scent station tracking bed to determine if predators visited the site. Biologists monitor abundance and distribution of bobcats, coyotes, and foxes in Sonoran Desert scrub habitat using scent tracking stations.

Removal of predators has been used as an experimental element to study population-level effects of predation, but no studies have evaluated specific influences of predation by bobcats on populations of prey. Intensive, nonspecific methods of predator control in Texas resulted in removal of 120 bobcats and 337 other sympatric predators during February to June over about a 2-year period (Beasom 1974a,b). Although populations of predators rebounded each year when control efforts ceased, intensive control efforts might have enhanced short-term reproductive success of wild turkeys (*Meleagris gallopavo*), white-tailed deer, and bobwhite quail (*Colinus virginianus*). Calling and shooting, or use of trained trail hounds, provide selective means for control or sport harvest of predators (Dolbeer et al. 1994).

Interaction with Humans

Bobcats in Arizona often occur in urban areas, contributing to potential for conflicts with humans (M. Zornes, Arizona Game and Fish Department, personal communication). Human developments likely have minimal effect on activities of or use of habitats by bobcats (McCord and Cardoza 1982). Little information is available regarding interactions between bobcats and humans, but several attacks or near-attacks of humans by rabid bobcats have been reported (Young 1958). Rabies in bobcats in the southeastern United States corresponded with apparent epizootics of the disease in dogs and foxes. Bobcats in some areas occasionally prey on domestic livestock, particularly sheep (McCord and Cardoza 1982), contributing to conflicts with humans.



Bobcats can be found in urban areas in Arizona. They are attracted to backyards with abundant wildlife, birds, small pets, water, shade, and shelter.

Population Trends

Abundance of bobcats in the western United States might have increased dramatically during the early 1950s, possibly corresponding with decline in abundance of coyotes caused by Compound 1080. Abundance of bobcats in the West might have increased after this time, but declined between the mid-1960s and late 1970s (Knowlton and Tzilkowski 1979, Towell 1979). Development of an accurate, non-destructive method for monitoring habitat-specific trends of populations of bobcats is a critical need for effective management (Anderson 1987).

Harvest

Sport and illegal harvest combined likely is the most prevalent cause of mortality of bobcats (Anderson 1987, Anderson and Lovallo 2003). Bobcat harvest in Arizona presently is not limited during open hunting and trapping seasons from August through March. Reported annual sport harvest between 1981 and 2001 ranged from 408 to 3,235 bobcats. Harvest data are presumed to be useful only in evaluating long-term harvest trends. Harvest by trapping declined dramatically after about 1989. A public initiative passed in 1994 banned use of leg-hold steel traps on public lands in Arizona. Sport harvest by means other than trapping tended to increase above previous levels between 1997 and 2001 (Arizona Game and Fish Department 2003, 2006). Although sex ratios at birth usually are equal, exploited populations might show a preponderance of males.

Sex ratios of adult bobcats in populations usually ranged between about 0.9 and fewer than 2.0 males per female, but hunting and trapping appeared to have no clear effect on sex ratios (Anderson 1987). Sex ratios in about 15 states ranged widely from 0.40 to 2.99 males per female (Bailey 1974, McCord and Cardoza 1982). Survival of bobcats might be higher among age classes 0 to 1, 1 to 2, and 2 to 3 years, and lower within age classes 3 to 4 years and older (McCord and Cardoza 1982). In contrast, harvest of bobcats in Wyoming tended to be proportionally greater among age classes 0 to 1, 1 to 2, 2 to 3, and 3 to 4 years than for older animals (M. Zornes, Arizona Game and Fish Department, unpublished data). Sport harvest in Maine accounted for 55% of annual mortality of bobcats at least 1 year old (Litvaitis et al. 1987). In general, proportion of bobcats less than 2 years old in a population is closely related to intensity of harvest, possibly reflecting high mortality of adults or a high level of reproduction (Anderson 1987). Currently proposed bobcat management guidelines for Arizona include:

- 1) Develop and implement standard, statewide survey methods to index relative abundance;
- 2) Evaluate age structure of harvest through jaws submitted by hunters and trappers; and
- 3) Develop an annual harvest survey using expanded format to enhance data gathering.

Predator/prey Relationships

Predation by bobcats was closely associated with abundance of prey, in that the number of prey species in diets increased as abundance of the principal prey species decreased (Baker et al. 2001). No study has implicated bobcats as a major source of mortality for populations of deer (Ballard et al. 2001). Predation by bobcats in New Brunswick caused only 4% of mortalities of white-tailed deer

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fawns during the first 6 months of life (Ballard et al. 1999). Bobcats killed 10% of 67 mule deer examined in Idaho (Koehler and Hornocker 1991). Bobcats likely cause little mortality of white-tailed deer in Arizona (Knipe 1977). Although bobcats potentially prey on desert bighorn sheep, no evidence indicates predation by bobcats affects survival in populations (Kelly 1980, Krausman et al. 1999, Sawyer and Lindzey 2002).

Effects of Fire

Two bobcats remained in a burned area and survived during a wildfire in the Sonoran Desert in central Arizona, but abundance of bobcats declined in a burned area within 6 months post-fire, and returned to pre-fire levels of abundance within 2 years. Abundance of small prey tended to be similar or greater on burned than on unburned areas, despite loss of cover. Bobcats returned to the burned area as vegetation cover increased, and dispersal might have been linked to avoidance of the burned area because of lack of cover. In comparison, other studies in Arizona, California, and Utah indicated bobcats increased following fires (Cunningham et al. 2001).

Research Needs

Gaps in knowledge about bobcats and their populations in Arizona occur in several areas relevant to needs of management. Managers and researchers need the following studies and information to improve prescriptions for and assessments of management actions.

- 1) Develop methods to determine and monitor statewide, habitat-specific relative abundance and trends of populations. Scent stations might be particularly applicable in this respect, but other methods, including scat transects should be investigated.
- 2) Evaluate environmental variables potentially influencing habitat-specific abundance, dynamics, and structures of populations, including intensity of harvest, rainfall, abundance of prey, and abundance and distribution of coyotes.
- 3) Evaluate habitat-specific aspects potentially linked to dispersal of young and long-range movements of adults, including intensity of harvest, fragmentation of habitats, travel corridors, human developments, and areas of intensive recreational activities.

Literature Cited

Anderson, E. M. 1987. A critical review and annotated bibliography of literature on the bobcat. Special Report No. 62. Colorado Division of Wildlife, Denver.

Biological Foundations for Management of Mammalian Predators in Arizona

- Anderson, E. M., and M. J. Lovallo. 2003. Bobcat and lynx. Pages 758–786 in G. A. Feldhamer, B. C. Thompson, and J. A. Chapman, editors. *Wild mammals of North America*. Johns Hopkins University Press, Baltimore, Maryland.
- Anonymous. 1982. Arizona status report. Page 6 in S. J. Stiver, editor. *Proceedings of the Western States Bobcat Workshop*. Reno, Nevada.
- Arizona Game and Fish Department. 2003. *Hunt Arizona*. Arizona Game and Fish Department, Information Branch, Phoenix.
- Arizona Game and Fish Department. 2006. *Hunting and Trapping Regulations*. Arizona Game and Fish Department, Phoenix.
- Bailey, T. N. 1974. Social organization in a bobcat population. *Journal of Wildlife Management* 38:435–446.
- Baker, L. A., R. J. Warren, D. R. Diefenbach, W. E. James, and M. J. Conroy. 2001. Prey selection by reintroduced bobcats (*Lynx rufus*) on Cumberland Island, Georgia. *American Midland Naturalist* 145:80–93.
- Ballard, W. B., H. A. Whitlaw, S. J. Young, R. A. Jenkins, and G. J. Forbes. 1999. Predation and survival of white-tailed deer fawns in northcentral New Brunswick. *Journal of Wildlife Management* 63:574–579.
- Ballard, W. B., D. Lutz, T. W. Keegan, L. H. Carpenter, and J. C. deVos, Jr. 2001. Deer-predator relationships: a review of recent North American studies with emphasis on mule and black-tailed deer. *Wildlife Society Bulletin* 29:99–115.
- Beasom, S. L. 1974a. Intensive short-term predator removal as a game management tool. *North American Wildlife Conference* 39:230–240.
- Beasom, S. L. 1974b. Relationships between predator removal and white-tailed deer net productivity. *Journal of Wildlife Management* 38:854–859.
- Beasom, S. L., and R. A. Moore. 1977. Bobcat food habit response to a change in prey abundance. *Southwestern Naturalist* 21:451–457.
- Beatley, J. C. 1969. Dependence of desert rodents on winter annuals and precipitation. *Ecology* 50:721–724.
- Beier, P., and S. Loe. 1992. A checklist for evaluating impacts to wildlife move-

Bobcat

- ment corridors. *Wildlife Society Bulletin* 20:434–440.
- Beier, P. 1993. Determining minimum habitat areas and habitat corridors for cougars. *Conservation Biology* 7:94–108.
- Brady, J. R. 1979. Preliminary results of bobcat scent station transects in Florida. *National Wildlife Federation Scientific and Technical Series* 6:101–103.
- Brown, C. F., and P. R. Krausman. 2003. Habitat characteristics of 3 leporid species in southeastern Arizona. *Journal of Wildlife Management* 67:83–89.
- Conner, M. C., R. F. Labinsky, and D. R. Progulske, Jr. 1983. Scent-station indices as measures of population abundance for bobcats, raccoons, gray foxes, and opossums. *Wildlife Society Bulletin* 11:146–152.
- Crowe, D. M. 1972. The presence of annuli in bobcat tooth cementum layers. *Journal of Wildlife Management* 36:1330–1332.
- Crowe, D. M. 1975. Aspects of ageing, growth, and reproduction of bobcats from Wyoming. *Journal of Mammalogy* 56:177–198.
- Crowe, D. M., and M. D. Strickland. 1975. Population structures of some mammalian predators in southeastern Wyoming. *Journal of Wildlife Management* 39:449–450.
- Cunningham, S. C., L. M. Monroe, L. Kirkendall, and C. L. Ticer. 2001. Effects of the catastrophic Lone Fire on low, medium, and high mobility wildlife species. *Technical Guidance Bulletin No. 5*, Arizona Game and Fish Department, Phoenix.
- Day, G. I. 1985. *Javelina: research and management in Arizona*. Arizona Game and Fish Department, Phoenix.
- Delibes, M., and F. Hiraldo. 1987. Food habits of the bobcat in two habitats of the southern Chihuahuan Desert. *Southwestern Naturalist* 32:457–461.
- Delibes, M., M. C. Blazquez, R. Rodriguez-Estrella, and S. C. Zapata. 1997. Seasonal food habits of bobcats (*Lynx rufus*) in subtropical Baja California Sur, Mexico. *Canadian Journal of Zoology* 74:478–483.
- Dolbeer, R. A., N. R. Holler, and D. W. Hawthorne. 1994. Identification and control of wildlife damage. Pages 272–506 in T. Bookhout, editor. *Research and management techniques for wildlife and habitats*. The Wildlife

Biological Foundations for Management of Mammalian Predators in Arizona

Society, Allen Press, Lawrence, Kansas.

- Gashwiler, J. S., W. L. Robinette, and O. W. Morris. 1960. Foods of bobcats in Utah and Nevada. *Journal of Wildlife Management* 24:226–229.
- Gittleman, J. L. 1985. Carnivore body size: Ecological and taxonomic correlates. *Oecologia* 67:540–554.
- Golley, F. B., G. A. Petrides, E. L. Rauber, and J. H. Jenkins. 1965. Food intake and assimilation by bobcats under laboratory conditions. *Journal of Wildlife Management* 29:442–447.
- Hemmer, H. 1976. Fossil history of living Felidae. *The World's Cats* 3:1–14.
- Henke, S. E., and F. C. Bryant. 1999. Effects of coyote removal on the faunal community in western Texas. *Journal of Wildlife Management* 63:1066–1081.
- Hon, T. 1979. Relative abundance of bobcats in Georgia: survey techniques and preliminary results. *National Wildlife Federation Scientific and Technical Series* 6:104–106.
- Hornocker, M., and T. Bailey. 1986. Natural regulation in three species of felids. Pages 211–220 in S. D. Miller, and D. D. Everett, editors. *Cats of the world: biology, conservation, and management*. National Wildlife Federation, Washington, D.C.
- Johnson, M. K., and D. R. Aldred. 1982. Mammalian prey digestibility by bobcats. *Journal of Wildlife Management* 46:530.
- Johnson, M. K., R. C. Belden, and D. R. Aldred. 1984. Differentiating mountain lion and bobcat scats. *Journal of Wildlife Management* 48:239–244.
- Johnson, N. F., B. A. Brown, and J. C. Bosomworth. 1981. Age and sex characteristics of bobcat canines and their use in population assessment. *Wildlife Society Bulletin* 9:203–206.
- Jones, J. H., and N. S. Smith. 1979. Bobcat density and prey selection in central Arizona. *Journal of Wildlife Management* 43:666–672.
- Kelly, W. E. 1980. Predator relationships. Pages 186–196 in G. Monson, and L. Sumner, editors. *The desert bighorn*. University of Arizona Press, Tucson.

Bobcat

- Knick, S. T., J. D. Brittell, and S. J. Sweeney. 1985. Population characteristics of bobcats in Washington state. *Journal of Wildlife Management* 49:721–728.
- Knipe, T. 1956. Javelina in Arizona. *Wildlife Bulletin No. 2*. Arizona Game and Fish Department, Phoenix.
- Knipe, T. 1977. The Arizona whitetail deer. *Special Report No. 6*. Arizona Game and Fish Department, Phoenix.
- Knowlton, F. F., and W. M. Tzilkowski. 1979. Trends in bobcat visitations to scent-station survey lines in western United States, 1972–1978. *National Wildlife Federation Scientific and Technical Series* 6:8–12.
- Koehler, G. M., and M. G. Hornocker. 1989. Influences of seasons on bobcats in Idaho. *Journal of Wildlife Management* 53:197–202.
- Koehler, G. M., and M. G. Hornocker. 1991. Seasonal resource use among mountain lions, bobcats, and coyotes. *Journal of Mammalogy* 72:391–396.
- Krausman, P. R., A. V. Sandoval, and R. C. Etchberger. 1999. Natural history of desert bighorn sheep. Pages 139–191 in R. Valdez, and P. R. Krausman, editors. *Mountain sheep of North America*. University of Arizona Press, Tucson.
- Lawhead, D. N. 1984. Bobcat (*Lynx rufus*) home range, density, and habitat preferences in south-central Arizona. *Southwestern Naturalist* 29:105–113.
- Leopold, B. D., and P. R. Krausman. 1986. Diets of 3 predators in Big Bend National Park, Texas. *Journal of Wildlife Management* 50:290–295.
- Litvaitis, J. A., J. T. Major, and J. A. Sherburne. 1987. Influence of season and human-induced mortality on spatial organization of bobcats (*Felis rufus*) in Maine. *Journal of Mammalogy* 68:100–106.
- Litvaitis, J. A., and D. J. Harrison. 1989. Bobcat-coyote niche relationships during a period of coyote population increase. *Canadian Journal of Zoology* 67:1180–1188.
- Livingston, T. R., P. S. Gipson, W. B. Ballard, D. M. Sanchez, and P. R. Krausman. 2005. Scat removal: a source of bias in feces-related studies. *Wildlife Society Bulletin* 33:172–178.

Biological Foundations for Management of Mammalian Predators in Arizona

- Major, J. T., and J. A. Sherburne. 1987. Interspecific relationships of coyotes, bobcats, and red foxes in western Maine. *Journal of Wildlife Management* 51:606–616.
- Mautz, W. W., and P. J. Pekins. 1989. Metabolic rate of bobcats as influenced by seasonal temperatures. *Journal of Wildlife Management* 53:202–205.
- McCord, C. M., and J. E. Cardoza. 1982. Bobcat and lynx. Pages 728–766 in J. A. Chapman, and G. A. Feldhamer, editors. Johns Hopkins University Press, Baltimore, Maryland.
- McCulloch, C. Y. 1986. A history of predator control and deer productivity in northern Arizona. *Southwestern Naturalist* 31:215–220.
- McKinney, T., and T. W. Smith. 2007. Diets of sympatric bobcats and coyotes during years of varying rainfall in central Arizona. *Western North American Naturalist* 67:8–15.
- Murie, O. J. 1974. A field guide to animal tracks. Second edition. Houghton Mifflin Company, New York.
- Pond, D. B., and B. W. O’Gara. 1994. Chemical immobilization of large mammals. Pages 125–139 in T. Bookhout, editor. Research and management techniques for wildlife and habitats. The Wildlife Society, Allen Press, Lawrence, Kansas.
- Reichman, O. J., and K. M. Van De Graaff. 1975. Associations between ingestion of green vegetation and desert rodent production. *Journal of Mammalogy* 56:503–506.
- Rolley, R. E. 1985. Dynamics of a harvested bobcat population in Oklahoma. *Journal of Wildlife Management* 49:283–292.
- Rolley, R. E. 1987. Bobcat. Pages 671–681 in M. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, editors. Wild Furbearer Management and Conservation in North America. Ministry of Natural Resources, Ontario, Canada.
- Rosenzweig, M. L. 1966. Community structure in sympatric Carnivora. *Journal of Mammalogy* 47:602–612.
- Samuel, W. M., M. J. Pybus, and A. A. Kocan. 2001. Parasitic diseases of wild mammals. Second edition. Iowa State University Press, Ames.

Bobcat

- Sanchez, D. M., P. R. Krausman, T. R. Livingston, and P. S. Gipson. 2004. Persistence of carnivore scat in the Sonoran Desert. *Wildlife Society Bulletin* 32:366–372.
- Sawyer, H., and F. Lindzey. 2002. A review of predation on bighorn sheep (*Ovis canadensis*). Wyoming Cooperative Fish and Wildlife Research Unit, Laramie.
- Toweill, D. E. 1979. Bobcat populations—a review of available literature. Oregon Department of Fish and Wildlife Information Report Series, No. 79–2.
- Williams, E. S., and I. K. Baker. 2001. Infectious diseases of wild mammals. Iowa State University Press, Ames.
- Witmer, G. W., and D. S. deCalesta. 1986. Resource use by unexploited sympatric bobcats and coyotes in Oregon. *Canadian Journal of Zoology* 64:2333–2338.
- Young, S. P. 1958. The bobcat of North America. Stackpole Company, Harrisburg, Pennsylvania, and The Wildlife Management Institute, Washington, D.C.

Chapter 3

Mexican Wolf

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Common Name - Mexican wolf
Scientific Name - *Canis lupus baileyi*

Life History

Description

Mexican wolves (*Canis lupus baileyi*) generally weigh between about 50 and 90 pounds, or 2 to 3 times as much as coyotes (*Canis latrans*). Adult Mexican wolves are about 4.5 feet in overall length and about 32 inches tall. Coloration varies widely, but tends to be brindled, grizzled, or mottled, and ranges from shades of gray and black to brown, buff, and red. Ears and muzzles are shorter than those of coyotes, and the head is larger and blockier.

Tracks of coyotes and Mexican wolves seem similar in configuration, and often show 4 toes with claw marks, and triangular-shaped heel pads. Tracks of adult Mexican wolves generally are about 3.8 inches wide and 4.8 inches long, and are larger than those of coyotes and most domestic dogs. Small scats (fecal droppings) of wolves potentially overlap in size with large scats of coyotes, hindering definitive visual identification, but scats of canids more than 1.1 inches maximum diameter likely are from wolves (Reed et al. 2004).



The presence of wolf tracks allows wildlife biologists to identify wolf habitat and conduct population surveys.

Distribution

Gray wolves (*C. lupus*) have been present in the Southwest since at least the late Pleistocene (Lindsay and Tessman 1974). Although subject to considerable



debate, there may have been as many as 3 subspecies of gray wolves in Arizona, including Mexican wolves (Hall and Kelson 1959). Mexican wolves were released into the Blue Range Wolf Recovery Area in Arizona and New Mexico beginning in 1998 to restore the subspecies in a portion of its historic range (Reed et al. 2004). Mexican wolf distribution in Arizona is limited by the absence of any naturally occurring, fully protected wolves. Distribution is also limited through the Non-essential Experimental Special Rule, which ensures that the species is protected from intentional harm, yet doesn't allow the population to restrict current or potential land development or activities. The rule designates the wolf population as experimental and non-essential to the survival of the species. Wolves in Arizona are currently allowed to establish home ranges only within the Apache National Forest and the White Mountain Apache Reservation.



Mixed-conifer forests provide habitat for Mexican wolves in Arizona.

There are presently no formal, agency-sponsored evaluations of other reintroduction sites within Arizona, although the North Kaibab Plateau has been proposed by some environmental groups and might provide potential habitat for Mexican wolves. Mexican wolves do not seem to select for roadless wilderness areas or steep, inaccessible rocky terrain. Access to areas within the reintroduction area varies from a day's mule ride on rough mountain trails or cross-country to near roads.

Habitat

Mexican wolves inhabited a variety of vegetation types in Arizona prior to extirpation in the early 1900s. Wolves were never abundant in the state, and densities likely were lowest on the North Kaibab National Forest (Hoffmeister 1986). Early naturalists often might have confused wolves with coyotes, confounding early estimates of abundance and distribution. Most reports of historical distribution indicated wolves inhabited areas of woodlands above 4,500 feet elevation (Girmendonk 1994) that included oak (*Quercus* spp.), pinyon pine (*Pinus edulis*), juniper (*Juniperus* spp.), ponderosa pine (*P. ponderosa*), and mixed-conifer forests (Bailey 1931, McBride 1980).

Other subspecies of gray wolves inhabit vegetation types ranging from

spruce-fir (*Picea* spp.-*Abies* spp.) alpine woodlands to pinyon-juniper woodlands (Mech 1970). Wolves in the northern hemisphere potentially inhabit almost every vegetation type except arid deserts and tropical rainforests. Vegetation type affects availability of prey, and thus indirectly influences density and distribution of wolves. Vegetation type might not be limiting if an area supports healthy populations of ungulates (hoofed mammals), but wolves have not persisted in areas that do not support large ungulate prey.

During pre-settlement times, wolves inhabited mostly montane woodlands and were extirpated gradually from the most accessible areas following increases in human populations and production of livestock. Most wolves were extirpated from Arizona by the 1950s, but occasional sightings continued to be reported, primarily along the border between Mexico and United States (Brown 1983). The last known wolf in Arizona likely was killed in 1970 (Girmendonk 1994).

Important characteristics of habitats affecting abundance and distribution of wolves other than vegetation type include road densities and human developments, current land uses, and sizes of relatively intact patches of suitable habitat. Studies in Wisconsin indicated density of roads can be used to index and evaluate quality of wolf habitat (Mladenoff et al. 1997), but interactions between humans and wolves are more complex than simply road density. Different types of human activities during various times of year have variable effects on wolves. Mere disturbance by humans might not negatively affect wolves, but purposeful killing or accidental collisions with vehicles can have significant effects. As sizes of intact, remote patches of habitat decline, interactions between humans and wolves will increase.

Diets and Prey

Historical records indicate Mexican wolves may prey on several species of large ungulates, as well as rabbits and rodents, but there have been few detailed studies of food habits. Preliminary analyses of scats from reintroduced Mexican wolves indicated



Mexican wolves prey on large ungulates such as elk.

about 75% of fecal samples contained hair from elk (*Cervus elaphus*). Remains in samples of scats also included 10% deer (*Odocoileus* spp.), 4% livestock, and 11%

small mammals and unknown items (Reed et al. 2006). In comparison, wolves in Yellowstone National Park in 2002 killed mostly elk (84%), and preyed less on bison (*Bison bison*; 6%) and deer (1%) (Smith et al. 2003). On average, a Mexican wolf would likely consume about 2,800 pounds of prey per year, comprised primarily (80%) of deer and elk (Parsons 1994).

Observations of wolves attacking live prey suggest predation primarily of elk, possibly due to greater availability of elk than other large prey. Elk calves and deer fawns seem to be primary food sources of Mexican wolves during summer. Mexican wolves also commonly feed on offal left by hunters and wounded game during autumn, but no studies have evaluated importance of scavenging as a source of food. Mexican wolves need to eat only a couple of times per week, because their highly elastic stomachs allow gorging on up to 20 pounds of food in a single feeding. Biomass of available prey directly influences relative abundance and densities of wolves (Fuller 1989). The formula $y = 0.041x$ (where y = numbers of wolves, and x = biomass of prey available) theoretically might describe the relationship between number of wolves potentially supported in an area and biomass of available prey (Paquet et al. 2001).

Behavior

Wolves that are fed artificially can become habituated to humans, similar to behavior of bears (*Ursus* spp.), coyotes, and foxes (*Urocyon cinereoargenteus*, *Vulpes* spp.). Fortunately, distribution of Mexican wolves in Arizona does not overlap major human population centers. However, potential for habituation and related problems remain at developed, dispersed campgrounds within the area inhabited by reintroduced Mexican wolves. Wolves pose a greater risk of injuring or killing humans, domestic pets, and other wolves if they are habituated to humans, but they generally pose little if any threat to human safety, unless they are sick, injured, or cornered (Mech 1990).

Captive wolves, domestic dogs, and wolf-dog hybrids, especially if released into the wild, pose a much more significant risk to human safety than do wild wolves. Although free-ranging Mexican wolves have never injured humans, wolves as well as other wildlife need to be actively managed to limit potential dependencies on humans or prevent negative interactions with humans. Even intensive hazing failed to discourage some Mexican wolves that became habituated to humans and frequented campsites. These animals had to be captured and returned to captivity. Other wolves that moved to areas near humans experienced high mortality rates and died before they could be recaptured.

Diseases

Gray wolves might be subject to infections of brucellosis, distemper, canine hepatitis, parvovirus, and rabies, as well as infestations of cestode parasites and ticks (Samuel et al. 2001). Mexican wolves thus might be susceptible to a

number of diseases and parasites likely already circulating in other canids in the reintroduction area.

Canine parvovirus and distemper are the most significant diseases wolves encounter. At least 1 pup in a relatively remote area died from parvovirus. Widespread exposure of pups to parvovirus, corresponding with their low survival, was thought to suppress a colonizing population of wolves in Wisconsin (Wydeven et al. 1996). Infectious agents of parvovirus probably exist throughout potential wolf habitat, and wolves must develop natural immunity to the virus, or recruitment of pups could be compromised. Wolves can contract rabies and potentially infect humans with the disease, but this rarely has been documented (Johnson 1992). Although wolves might act as host or vector for rabies, their low densities and remotely located habitats minimize potential for transmitting the disease to humans, especially when compared to potential for transmission of rabies by bats, skunks (*Mephitis mephitis* and related forms), and foxes.

Density and Size of Packs

Densities of Mexican wolves vary widely because individuals often travel together as a pack and seem to make disproportionate use of locations within landscapes. Wolves in some cases seem to use areas near prime calving grounds for elk more than areas such as established foot and horse trails often used by humans. Wolves also might use valley bottoms and other readily accessible areas in higher proportions than their availability (Singleton 1995).

Densities of colonizing Mexican wolves likely vary between about 1.6 and 5 per 100 square miles. Densities of other subspecies of wolves in northern Ontario ranged from 33 per 100 square miles on Coronation Island to between 0.8 and 1 per 100 square miles (Merriam 1964, Pimlott et al. 1969). Excluding pups born in 2004, there currently are about 12 free-ranging wolves on non-Tribal lands in Arizona. Estimated carrying capacity for Mexican wolves within the reintroduction area of Arizona is at most about twice the current population.

Mexican wolf packs average between 4 and 5 individuals; they historically averaged between 3 and 8 animals (Bailey 1931, McBride 1980). In Yellowstone National Park, where a northern subspecies of gray wolf has been reintroduced, size of packs in 2002 averaged 10.6 animals (Smith et al. 2003).

Reproduction

Wolves primarily are monogamous, even though a pack can include more than 1 sexually mature female (Mech 1970), but polygyny (more than 1 mate at a time) occurs occasionally (Mech and Nelson 1989). Behavioral and physiological adaptations usually prevent more than 1 female per pack from breeding, which usually occurs in February (McBride 1980). If a reproductively active wolf dies or is removed from a pack, another wolf from within or outside the pack can immediately become reproductively active and mate prior to the next breeding

season. However, removal of an alpha (dominant) or reproductively active female or male can disrupt a pack, resulting in dissolution of the pack and independent distribution and movements of members of the pack.

Wolves in the wild usually do not mature sexually and breed until they are at least 2 years old (Rausch 1967), but 1 female in the wild bred at 10 months of age (Mech and Seal 1987). Male Mexican wolves in the wild can be reproductively active until at least 8 years of age, whereas females can breed until at least 7 years of age, but maximum age for reproduction is unknown. Courtship behavior begins during winter months, when potential mates remain closer and interact physically with each other more often. Scent marking activity increases during courtship, and an alpha female eventually exhibits raised leg urinations alongside her mate, expelling urine tinged with pre-estrus blood (Mech 1970).

Mexican wolves dig dens into soft soil or locate them under various objects, including logs and rock ledges (Bailey 1931). Dens might be reused, but reintroduced Mexican wolves likely move dens annually, even if just a short distance. After a gestation period of about 63 days, a single litter of 4 to 7 pups usually is born in April. Average size of 16 litters of wild Mexican wolves



Biologists attach a radiocollar to a Mexican wolf to learn about distribution and movement of wolves in Arizona.

was 5.6 pups (McBride 1980). Average size of litters of gray wolves during mid-winter in Yellowstone National Park in 2002 was 4.3 pups (Smith et al. 2003).

Pups are weaned at 5 to 6 weeks of age and remain totally dependent on adults until at least 9 to 10 months old. All members of packs care for pups. About 6 weeks after pups are born, wolves move them away from a den site to another area near water, called a rendezvous site. Pups and other pack members use rendezvous sites as centers of activity during summers, and pups usually begin traveling with adults on short forays by October. Sex ratios of pups (male:female) reportedly can be nearly equal, but ratios in some cases might favor males (Mech 1975). Annual mortality of pups commonly is about 50%, but varies widely, depending on densities of prey and competitors, disease, and weather (Rausch 1967, Mech 1977, Fuller 1989).

Home Ranges, Dispersal, and Territories

Reintroduced Mexican wolves can establish home ranges relatively quickly if they are held in a chain link enclosure for a few weeks to 2 months prior to release. Wolves also have shown high site fidelity if released with a litter of young pups. Sizes of home ranges of colonizing Mexican wolves based on the 95% minimum convex polygon method (Mohr 1947) vary from about 161 to more than 250 square miles. However, some studies of gray wolves recorded an average home range size of 123 square miles. Home range sizes in Minnesota (Stenlund 1955) and Alaska (Burkholder 1959) varied from 36 to more than 5,000 square miles, respectively, but no studies have evaluated size of home range in relation to selection of habitats by reintroduced wolves.

Most wolves released from captivity in the future probably will have had prior experience in the wild. Home range sizes and dispersal movements likely will be similar to those observed for Mexican wolves that have been released in Arizona. Wolves colonizing in Montana in the 1980s remained in a portion of their natal territory, and established territories adjacent to natal locations. Dispersing individuals moved more than

100 miles (Ream et al. 1991). Mexican wolves seem to exhibit similar behavior. About 15% of Mexican wolves released in Arizona eventually dispersed from release areas and did not return before either death or recapture. This percentage is less than in other studies of dispersal, where



Biologists weigh a Mexican wolf to record biological information before the animal is released.

up to 40% of wolves moved more than 20 miles (Boyd et al. 1996). Another 15% of Mexican wolves released in Arizona moved away from their release site after an alpha animal died, or following hard release with no acclimation period (not held in enclosures prior to release).

Biologists actively pursued and attempted to recapture wolves that left the reintroduction area, possibly affecting their movements. One Mexican wolf dispersed 300 miles from its release site before being struck and killed by a vehicle near Flagstaff, Arizona. In comparison, female wolves in Montana dispersed an average of 165 miles, and males dispersed an average of 95 miles (Boyd et al. 1996). Both sexes disperse primarily during winter, and females and males tended

to disperse at 2 and 3 years of age, respectively. The longest documented dispersal was at least 570 miles (Fritts 1983).

Active defense of a territory between different packs that involved physical encounters resulting in injuries has not been observed in the reintroduced population of Mexican wolves. However, preliminary analysis of location data shows relatively little temporal overlap of home ranges between most packs, indicating typical territorial behavior.

Survival

Reintroduced Mexican wolves experience levels of mortality similar to other populations of wolves that are exposed to high levels of human activity. During the first 5 years of the reintroduction project, humans caused most mortalities; 56% of deaths resulted from gunshots and collisions with vehicles. Natural causes accounted for 24% of deaths, and 20% of deaths were from unknown causes (Table 1). Wolves in the Greater Yellowstone Area experienced similar levels of mortality; 55% of deaths were caused by humans, 26% were from natural causes, and 19% were from unknown causes (Smith et al. 2003). Humans also caused 72% of deaths of wolves in Wisconsin during early years of recovery efforts, but caused only 22% of mortalities after 1985 (Wydeven et al. 1996). Human-related deaths and fragmentation of habitats are the primary threats to Mexican wolf survival.

Table 1. Known causes of mortalities of reintroduced Mexican wolves in Arizona, 1998 to 2003.

Cause of mortality	Number of mortalities	Percent of total mortalities
Brain tumor	1	4
Capture myopathy	1	4
Dehydration	1	4
Infection	1	4
Mountain lion	1	4
Parvovirus	1	4
Shot (firearms)	11	38
Snake bite	1	4
Unknown	6	20
Vehicle collision	4	14
Total mortalities	28	100

Detailed age-specific rates of mortality have not been calculated for Mexican wolves, partly because of artificial manipulation of the population through releases and removals for various reasons. About 24% of wolves released or born in the wild following releases died during the first 5 years of the project. Yearling Mexican wolves have a higher rate of mortality than other age classes, comprising about 36% of all mortalities. Ongoing radiotelemetry monitoring and analyses of data will provide estimations of rates for specific mortality causes. Colonizing wolves in Wisconsin experienced a 39% rate of mortality from 1979 to 1985, but mortality rate dropped to 18% between 1985 and 1992 (Wydeven et al. 1996). Some populations of wolves have persisted despite a sustained human-caused mortality rate of 30% (Mech 1970).

Wolves are vulnerable to predation control efforts because their populations can be reduced in localized areas, but if a reservoir population is nearby, recolonization can occur within a few years (Hayes et al. 2003). Outreach efforts intended to inform people about the presence of Mexican wolves and their similarities to coyotes will hopefully reduce accidental shootings of wolves.

Management and Conservation

Monitoring

Mexican wolves can be monitored using a variety of methods (Girmendonk and Groebner 1994), including tracking, scent posts, howling, photo traps, and other survey techniques. In addition, attaching a radiocollar to 1 member of a pack enables researchers to study communal behaviors and identify individual pack members, as well as pack size. Once typical size of home range of a pack is determined for an area, biologists can extrapolate to estimate size of a population over a larger area.

Effects on Prey Populations

Wolves in certain situations potentially limit abundance of prey (Mech and Karns 1977, Fuller 1990, Hayes et al. 2003). Wolves might affect populations of prey most when predation occurs within the same time span when other factors potentially influence abundance of prey, such as deep snow, disease, or a long winter. However, most research on predator-prey interactions indicates that wolves do not eliminate prey, but rather reduce magnitudes of fluctuations in prey populations.

Computer modeling using a reintroduction goal of 100 Mexican wolves for Arizona and New Mexico indicated effects on populations of prey varied, depending on factors other than predation (Green-Hammond 1994). The model was designed to evaluate impacts of predation on existing populations of prey using estimates of 0% and 25% alternate prey and levels of 17% and 50% com-

pensatory mortality (Green-Hammond 1994). If a population of ungulates was decreasing because of drought, predation by wolves would accelerate decline of the prey population. If a population of ungulates plausibly would increase without predation by wolves because of increased precipitation, that prey population would increase if wolves were present, but it would not increase as quickly. Annual changes in populations of prey as a result of predation by wolves never exceeded 3% for deer or elk, but effects of predation could accumulate over time.

When a reintroduction goal of about 100 wolves is reached, wolves might consume the equivalent of about 1/3 of an elk and 1/8 of a deer per square mile per year. Reintroduced Mexican wolves may not exert totally additive (adds to other sources of mortality) or totally compensatory (replaces other sources of mortality) levels of predation on populations of prey, but detailed studies will be necessary to confirm or refute this conclusion. Although unverified by research, at least a portion of predation by wolves is thought to be compensatory with other types of predation.

Mexican wolves might be expected to influence populations of prey differently than mountain lions (*Puma concolor*), coyotes, and bears. Cursorial hunting behavior by a pack of wolves (a hunting strategy practiced by wolves; wolves are slower than their prey over short distances, but have greater endurance over long distances) should select for less fit animals within the population of prey. In contrast, mountain lions and bears tend to stalk and ambush prey, and likely exert less selective pressure on unfit animals. However, a distinct difference in predation by wolves and mountain lions was not found in a study in and near Glacier National Park in Montana (Kunkel et al. 1999).

Effects on other Predators

Although no studies were conducted to investigate effects of Mexican wolves on other predators before wolves were extirpated in Arizona, anecdotal evidence suggests wolves might have suppressed populations of coyotes and mountain lions. Coyotes and mountain lions likely increased in abundance and distribution once wolves were extirpated from Arizona (Ligon 1927, Leopold 1959, Bednarz 1989, deVos and McKinney 2005). Moreover, wolves might affect populations of coyotes through direct killing or causing avoidance by coyotes of areas inhabited by wolves (Berg and Chesness 1978, Fuller and Keith 1981). Wolves have killed mountain lions (Hornocker Wildlife Research Institute 1993), but mountain lions also have killed Mexican wolves. It appears that packs of wolves have little vulnerability to predation by mountain lions and bears, but young wolves separated from packs might be vulnerable to predation.

However, direct interspecific interactions among predators (between predator species) probably are uncommon, and mortality from this source might not be significant for wolves or other large predators. In comparison, if packs of wolves displace mountain lions and bears from kills they have made, these preda-

tors suffer missed meals, must make other kills, and perhaps must move from areas of interspecific conflicts.

Research Needs

More specific information based on larger sample sizes is needed to better understand the dynamics of Mexican wolf populations. Recruitment of pups is poorly documented, but if other wolf reintroduction projects are indicative, the reintroduced population of Mexican wolves in Arizona will expand to fill available habitat. Risk of hybridization between Mexican wolves, domestic dogs, wolf/dog hybrids, and coyotes is considered low within the reintroduction area, but information is needed on consequences of wolves dispersing outside the area into habitats where dogs and wolf/dog hybrids are more abundant.

Specific information is needed on predation rates and effects of the reintroduced population of Mexican wolves on populations of prey to enhance management of predators and prey. Understanding effects of colonization by reintroduced wolves on other predators also is key to effective management of sympatric populations of predators.

Literature Cited

- Bailey, V. 1931. Mammals of New Mexico. U.S. Department of Agriculture, Bureau of Biological Survey. North American Fauna 53:303–313.
- Bednarz, J. C. 1989. An evaluation of the ecological potential of White Sands Missile Range to support a reintroduced population of Mexican wolves. U.S. Fish and Wildlife Service Endangered Species Report No. 19. U.S. Fish and Wildlife Service, Albuquerque, NM.
- Berg, W. E., and R. A. Chesness. 1978. Ecology of coyotes in northern Minnesota. Pages 229–247 in M. Bekoff, ed. Coyotes: biology, behavior, and management. Academic Press, New York, NY.
- Boyd, D. K., P. C. Paquet, S. Donelon, R. R. Ream, D. H. Pletscher, and C. C. White. 1996. Transboundary movements of a recolonizing wolf population in the Rocky Mountains. Pages 135–140 in L. N. Carbyn, S. H. Fritts, and D. R. Seip, editors. Ecology and Conservation of wolves in a changing world. Canadian Circumpolar Institute, University of Alberta, Edmonton.
- Brown, D. E. 1983. The wolf in the Southwest; the making of an endangered species. The University of Arizona Press, Tucson.

Biological Foundations for Management of Mammalian Predators in Arizona

- Burkholder, B. L. 1959. Movements and behavior of a wolf pack in Alaska. *Journal of Wildlife Management* 23:1–11.
- deVos, J. C., and T. McKinney. 2005. Recent trends in North American mountain lion populations. Pages 297–307 in C. van Riper and D. J. Mattson, editors. *The Colorado Plateau—Biophysical, socioeconomic, and cultural research*. University of Arizona Press, Tucson.
- Fuller, T. K. 1989. Population dynamics of wolves in north-central Minnesota. *Journal of Wildlife Management* 53:4–48.
- Fuller, T. K. 1990. Dynamics of a declining white-tailed deer population in north-central Minnesota. *Journal of Wildlife Management* 54:3–44.
- Fuller, T. K., and L. B. Keith. 1981. Non-overlapping ranges of coyotes and wolves in northeastern Alberta. *Journal of Mammalogy* 62:403–405.
- Fritts, S. H. 1983. Record dispersal by a wolf from Minnesota. *Journal of Mammalogy* 64:166–167.
- Girmendonk, A. L. 1994. Mexican wolf sighting reports in Arizona; 1983–1993. Unpublished. Report. Arizona Game and Fish Department, Phoenix.
- Girmendonk, A. L., and D. J. Groebner. 1994. Mexican wolf identification and survey techniques. Unpublished. Manual for Arizona Game and Fish Department, Phoenix.
- Green-Hammond, K. A. 1994. Assessment of impacts to populations and human harvests of deer and elk caused by reintroduction of Mexican wolves. Contractor report to the U.S. Fish and Wildlife Service, Albuquerque, NM.
- Hall, E. R., and K. R. Kelson. 1959. *The Mammals of North America*. Vol. II. The Ronald Press, New York, NY.
- Hayes, R. D., R. Farnell, R. M. P. Ward, J. Carey, M. Dehn, G. W. Kuzyk, A. M. Baer, C. L. Gardner, and M. O'Donoghue. 2003. Experimental reduction of wolves in the Yukon: Ungulate responses and management implications. *Wildlife Monographs* 152.
- Hoffmeister, D. F. 1986. *Mammals of Arizona*. The University of Arizona Press and Arizona Game and Fish Department, Tucson.

Mexican Wolf

- Hornocker Wildlife Research Institute. 1993. Wolf-mountain lion relationships in northwest Montana. Unpublished interim report.
- Johnson, M. R. 1992. The potential role of rabies in relation to possible Yellowstone wolf populations. in J.D. Varley and W.G. Brewster, eds. *Wolves for Yellowstone? Report to the U.S. Congress*. Vol. 4 Research and Analysis. National Park Service, Yellowstone National Park, WY.
- Kunkel, K., T. K. Ruth, D. H. Pletscher, and M. G. Hornocker. 1999. Winter prey selection by wolves and cougars in and near Glacier National Park, Montana. *Journal of Wildlife Management* 63:901–910.
- Leopold, A. S. 1959. *Wildlife of New Mexico*. University of California Press, Berkeley.
- Ligon, J. S. 1927. *Wildlife of New Mexico*. Report to the State Game Commission. Department of Game and Fish, Santa Fe, NM.
- Lindsay, E. H., and N. T. Tessman. 1974. Cenozoic vertebrate localities and faunas in Arizona. *Journal of Arizona Academy of Science* 9:3–24.
- McBride, R. T. 1980. The Mexican wolf (*Canis lupus baileyi*); a historical review and observations on its status and distribution. U.S. Fish and Wildlife Service Endangered Species Report No. 8. U.S. Fish and Wildlife Service, Albuquerque, NM.
- Mech, L. D. 1970. *The wolf; the ecology and behavior of an endangered species*. Natural History Press, Garden City, NY.
- Mech, L. D. 1975. Disproportionate sex ratios of wolf pups. *Journal of Wildlife Management* 39:737–740.
- Mech, L. D. 1977. Productivity, mortality and population trend in wolves from northeastern Minnesota. *Journal of Mammalogy* 58:559–74.
- Mech, L. D. 1990. Who's afraid of the big bad wolf? Audubon, March. (Reprinted in *International Wolf* 2(3):3-7.
- Mech, L. D., and P. D. Karns. 1977. Role of the wolf in a deer decline in the Superior National Forest. U.S. Department of Agriculture, Forest Service, Research Paper NC-148.

Biological Foundations for Management of Mammalian Predators in Arizona

- Mech, L. D., and M. E. Nelson. 1989. Polygyny in a wild wolf pack. *Journal of Mammalogy* 70:3 675–676.
- Mech, L. D., and U. S. Seal. 1987. Premature reproductive activity in wild wolves. *Journal of Mammalogy* 68:871–873.
- Merriam, H. R. 1964. The wolves of Coronation Island. *Proceedings of Alaska Science Conference* 15:27–32.
- Mladenoff, D. J., R. G. Haight, T. A. Sickley, and A. P. Wydeven. 1997. Causes and implications of species restoration in altered ecosystems: a spatial landscape projection of wolf population recovery. *Bioscience* 47:21–31.
- Mohr, C. O. 1947. Table of equivalent populations of North American small mammals. *American Midland Naturalist* 37:223–249.
- Paquet, P. C., J. Vucetich, M. L. Phillips, and L. Vucetich. 2001. Mexican wolf recovery; three-year program review and assessment. Conservation Breeding Specialist Group for the U.S. Fish and Wildlife Service, Albuquerque, NM.
- Parsons, D. R. 1994. Wild prey predation scenarios. Unpublished Report. U.S. Fish and Wildlife Service, Albuquerque, NM.
- Pimlott, D. H., J. A. Shannon, and G. B. Kolenosky. 1969. The ecology of the timber wolf in Algonquin Park. Ontario Department of Lands and Forests, Canada.
- Rausch, R. A. 1967. Some aspects of the population ecology of wolves, Alaska. *American Zoologist* 7:253–265.
- Ream, R. R., M. W. Fairchild, D. K. Boyd, and D. H. Pletscher. 1991. Population dynamics and home range changes in a colonizing wolf population. Pages 349–366 in R. B. Keiter and M. S. Boyce, editors. *The greater Yellowstone ecosystem; redefining America's wilderness heritage*. Yale University Press, New Haven, CT.
- Reed, J. E., R. J. Baker, W. B. Ballard, and B. T. Kelly. 2004. Differentiating Mexican gray wolf and coyote scats using DNA analysis. *Wildlife Society Bulletin* 32:685–692.
- Reed, J. E., W. B. Ballard, P. S. Gipson, B. T. Kelly, P. R. Krausman, M. C. Wal-

Mexican Wolf

- lace, and D. B. Wester. 2006. Diets of free-ranging Mexican gray wolves in Arizona and New Mexico. *Wildlife Society Bulletin* 34:1127–1133.
- Samuel, W. M., M. J. Pybus, and A. A. Kocan. 2001. *Parasitic diseases of wild mammals*. 2nd edition. Iowa State University Press, Ames.
- Singleton, P. H. 1995. Winter habitat selection by wolves in the North Fork of the Flathead River Basin, Montana and British Columbia. M. S. Thesis. University of Montana, Missoula.
- Smith, D. W., D. R. Stahler, and D. S. Guernsey. 2003. *Yellowstone Wolf Project: Annual Report, 2002*. National Park Service, Yellowstone Center for Resources, Yellowstone National Park, WY.
- Stenlund, M. H. 1955. A field study of the timber wolf (*Canis lupus*) on the Superior National Forest, Minnesota. Minnesota Department of Conservation Technical Bulletin 4.
- Wydeven, A. P., R. S. Schultz, and R. P. Thiel. 1996. Monitoring of a recovering gray wolf population in Wisconsin, 1979–1991. Pages 147–156 in L. N. Carbyn, S. H. Fritts, and D. R. Seip, editors. *Ecology and Conservation of wolves in a changing world*. Canadian Circumpolar Institute, University of Alberta, Edmonton.

Chapter 4

Mountain Lion

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Common Name - Mountain Lion, Cougar, Puma, Panther, Catamount
Scientific Name - *Puma concolor*

Life History

Description

The mountain lion (*Puma concolor*), except for the slightly larger jaguar (*Felis onca*), is North America's largest member of the cat family. Tawny, reddish brown or grayish brown adult (more than 24 months old) females and males range from about 75 to 105 pounds and 115 to 150 pounds, respectively (adult jaguars might weigh between about 125 and 250 pounds). Mountain lion kittens are born fully furred with black spots that disappear by adulthood, and weigh about 1.1 pounds at birth (Anderson 1983).

Mountain lions are elusive and shy, but tracks in dirt, dust, or snow can document their presence in an area. Well-defined tracks are larger than those made by bobcats (*Lynx rufus*), but have similar appearance in that they show 4 toes, rarely show claw marks, and heel pads are two-lobed on anterior borders, three-lobed on posterior borders. Mountain lion tracks generally range between about 3.0 to 4.3 inches long and 3.3 to 4.8 inches wide. Strides can range between about 13 and 32 inches while walking, but can increase to about 12 feet when running. Tracks of jaguars are unlikely to be seen in Arizona except perhaps occasionally near the border with Mexico, but are similar in appearance and size to tracks of mountain lions.

Life Span

Biological life span of free-ranging mountain lions potentially might be about 12 to 13 years, but few adults survive to this age. Estimated age distributions in 3 exploited and 3 unexploited mountain lion populations in the southwestern United States varied, but most animals were less than 6 years old (Cunningham et al. 1995). Mountain lions in the wild and in captivity reportedly have lived up to 15 and 18 years of age, respectively (Logan and Sweanor 2001).



Distribution and Abundance

Mountain lions survived massive extinctions of mammals at end of the last Pleistocene ice age and likely experienced a founder event in southwestern North America about 12,000 years ago (Culver et al. 2000). Mountain lions were extirpated more than a century ago throughout states and provinces in eastern and midwestern North America, but presently seem to be increasing in abundance and distribution in these regions (deVos and McKinney 2005). In general, mountain lions are distributed more broadly throughout North and South America than any mammal other than humans.

Mountain lions regularly occur throughout Arizona (Figure 1), except for low desert environs in southwestern areas of the state where they likely are rare or absent (Germaine et al. 2000).

Relative abundance of mountain lions in Arizona seems to be linked to abundance of mule deer (*Odocoileus hemionus*); thus, mountain lions might be more abundant in interior chaparral than in other habitat types (Shaw et al. 1988), but also are abundant in pinyon-juniper (*Pinus edulis-Juniperus* spp.) habitat. Distribution of mountain lions in California also corresponded with availability of mule deer, but few studies have examined simultaneous distribution of predators and their prey (Pierce et al. 2000a). In comparison, only a weak relationship existed between abundance of mule deer and mountain lions in Utah (Lindzey et al. 1994).

Landscape-level distribution of mountain lions might be monitored using track surveys or other sign (Smallwood and Fitzhugh 1995, Germaine et al. 2000). Monitoring microhabitat distributions, however, would be most effective using radiotelemetry. Habitat modeling and mortalities of mountain lions in hunting districts were used to analyze landscape-scale abundance and distribution in Montana (Riley and Malecki 2001). Biologists in Arizona likely would benefit from use of this method.

Comparatively few studies have attempted to quantify densities of mountain lions, and results vary widely. Using track survey data, average density of a heavily exploited population in Arizona was estimated to be 0.04/square mile (Cunningham et al. 1995). In comparison, average densities of mountain lions in central Arizona on areas with depredation control and sport harvest (versus those without these factors) were 0.07 and 0.04 per square mile, respectively, suggesting areas subject to high depredation and sport harvest represented a population sink, (i.e., mountain lions tended to move into these areas from adjacent source populations) (Cunningham et al. 2001). Other studies have used capture-recapture methods (radiotelemetry, physical marking of animals) and track surveys to estimate densities (Logan et al. 1986, Van Sickle and Lindzey 1991, Van Sickle and Lindzey 1992, Lindzey et al. 1994, Logan and Sweanor 2001). Estimates of mountain lion densities in several states and Canadian provinces using radiotelemetry or other techniques ranged from 0.02 to 0.16 per square mile (Cunning-

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ham et al. 2001, Logan and Sweanor 2001). Intensive capture-recapture studies based on radiotelemetry in localized study areas might provide reasonably accurate estimations of density. Analysis of deoxyribonucleic acid (DNA) in feces also might be used to estimate density of mountain lions in some areas (Ernest et al. 2000), but preliminary studies indicated this approach was not feasible in a desert environment in central Arizona (McKinney and McRae 2001).

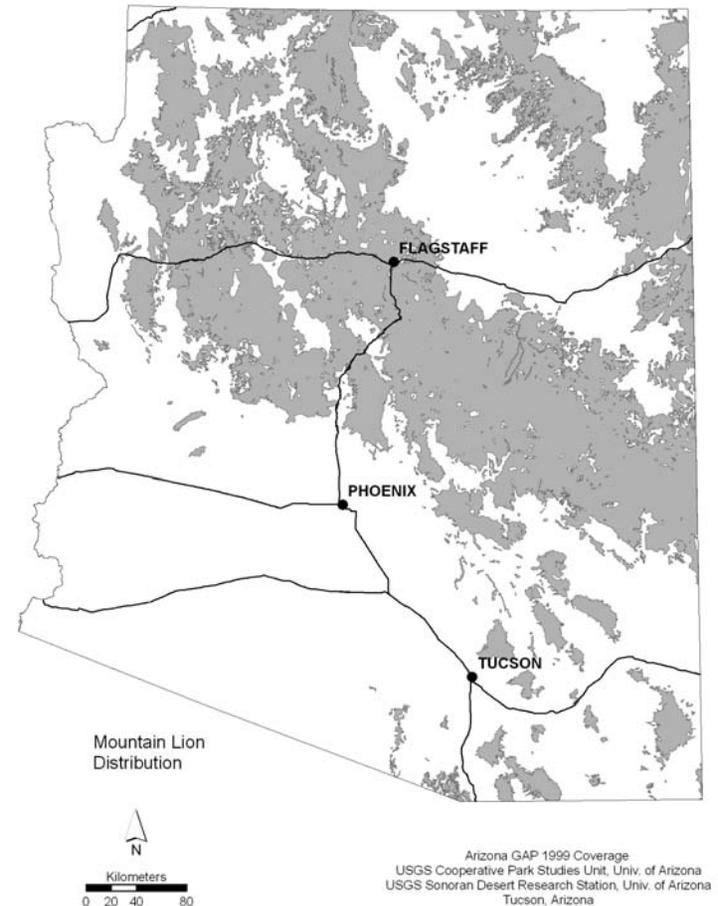


Figure 1. Distribution of mountain lions in Arizona.

Habitat

Mountain lions use habitat features selectively, are more abundant in steep, rugged topography, and might tend to avoid gentle terrain (Logan and Irwin 1985, Logan and Sweanor 2001, Riley and Malecki 2001). Loss of habitat as a result of human developments represents the greatest single threat to conservation of mountain lions (Logan and Sweanor 2001). Movement corridors between mountain ranges are a critical aspect of mountain lion habitat (Beier and Loe 1992; Beier 1993, 1995).



Rugged mountain lion habitat in the Mazatzal Mountains of central Arizona.

Diets and Prey

Mountain lions are obligate carnivores. Ungulates (hoofed mammals) likely represent nearly 70 % of their diets in North America (Iriarte et al. 1990). Mule and white-tailed (*O. virginianus*) deer, elk (*Cervus elaphus*), javelina (*Pecari tajacu*), and domestic cattle are principal prey of mountain lions in the West, but numerous studies indicate a variable diet. Bighorn sheep (*Ovis canadensis*) are considered to be an alternate prey to mule deer, the mountain lion's primary prey. Recent studies indicate mountain lions might select young (less than 1 year old) mule deer (Pierce et al. 2000a) and domestic calves (Cunningham et al. 1999) as prey, but among adult mule deer, they might select older animals (Pierce et al. 2000a) and females (Bleich and Taylor 1998), independent of body condition. In contrast, all age classes of white-tailed deer were equally vulnerable to predation by mountain lions (Kunkel and Pletscher 1999).

Based largely on analyses of feces contents, studies in Arizona and the Southwest indicate a wide range in mountain lion diets (Robinette et al. 1959, Leopold and Krausman 1986, Cashman et al. 1992, Cunningham et al. 1999, McKinney et al. 2006b). Diets of mountain lions (occurrence of remains identified primarily in scats) reflect a generalist predation strategy: bighorn sheep (2% to 26%), deer (39% to 85%), javelina (17% to more than 60%), cattle (trace to 34%), and rabbits and rodents (5% to 50%). Biologists believe mountain lions tend to hunt opportunistically; sex and individual behavior of the predator and relative availability and vulnerability of prey might influence predation (Leopold

and Krausman 1986, Ross et al. 1997, Logan and Sweanor 2001). Recent studies also indicate mountain lions kill and eat their own species (particularly kittens), and defend kills against other predators (Toweill and Meslow 1977, Boyd and O'Gara 1985, Koehler and Hornocker 1991, Pierce et al. 1998, Logan and Sweanor 2001). Remains of bobcats, coyotes (*Canis latrans*), and common gray foxes (*Urocyon cinereoargenteus*) occasionally occur in scats of mountain lions (Robinette et al. 1959, Cashman et al. 1992, McKinney et al. 2006b). Procedures based on studies with captive mountain lions allow researchers to convert frequency of occurrence of prey remains in scats to estimations of relative biomass and numbers of prey groups consumed (Ackerman et al. 1984).

Relationships between diets and spatial and temporal differences in abundance, distribution, and dynamics of mountain lions and their prey are poorly understood.

Results of hundreds of studies of predator diets provide few insights in this respect (Robbins 1983). Regardless, numerous investigations have provided valuable insights into partitioning of food resources, coexistence of predators, and structures of ecological communities (Koehler and Hornocker 1991, Arjo et al. 2002, McKinney and Smith 2007).



Mule deer are the primary prey of mountain lions in western North America.

Behavior

Hunting and Feeding.—Mountain lions are stalking predators, hunt primarily between dusk and dawn, and rely on cover and stealth to ambush prey or pursue them over short distances (Beier et al. 1995, Pierce et al. 2000a). They might kill large prey every 5 to 10 days, depending on age, sex, and reproductive status of the predator (Anderson and Lindzey 2003).

Mountain lions in 1 study stalked or sat in ambush an average of less than 1 hour, then traveled less than 1 mile in just more than an hour. They repeated this pattern about 6 times on nights when no prey was killed, and suspended the pattern when prey was killed. Mountain lions likely remain at kill sites until a carcass is consumed, requiring about 4 to 6 hours or less for small mammals and about 2 to 5 days for large prey (Beier et al. 1995). During feeding bouts after

sunset, females with kittens returned to kills earlier than when there were no kittens, possibly reflecting avoidance of cannibalism by conspecific males (Pierce et al. 1998). Evidence of predation or feeding by mountain lions at a kill site might include presence of scats, scrapes, tracks, canine teeth bite marks, carcass drag line, opening of the carcass first at abdomen or thorax, consumption of vital organs, expulsion of carcass paunch, hair plucking or shaving, and caching the carcass with a covering of ground debris (Shaw 1979, Logan and Sweanor 2001).

Dispersal.—Subadults tend to disperse from natal areas between about

10 to 33 months of age, and males have a greater tendency than females to disperse. Maximum dispersal distances between about 50 and more than 660 miles have been reported (Beier 1995, Sweanor et al. 2000, Logan and Sweanor 2001, Thompson and Jenks 2005).



Mountain lions are abundant in pinyon-juniper habitat in Arizona.

Dispersing mountain lions may move from and into subpopulations via habitat corridors in urban-fragmented landscapes. However, recent studies indicate connectivity of mountain lion populations among widely separated mountain ranges (Sweanor et al. 2000, Anderson et al. 2004). Encounters with humans and their environs during dispersal can increase mortalities (Beier 1995). Prescriptions for conservation and management of mountain lions should consider regional and landscape-level scales and connectivity among local populations (Beier 1993, 1995; Noss et al. 1996; Sweanor et al. 2000).

Home Ranges.—Annual home ranges of adult mountain lions reported in various studies range from about 58 to nearly 309 square miles for males, and from 15 to more than 116 square miles for females. Lifetime home ranges are somewhat larger than annual home ranges for both sexes. Habitat quality, sex, reproductive status, and intraspecific interactions influence home range size, but home range size was found to be independent of density and abundance of deer (Logan and Sweanor 2001). Adults tend to exhibit strong home range fidelity, but males are more likely than females to shift home ranges (Logan and Sweanor 2001), and adult mountain lions in some cases migrate seasonally, fol-

lowing movements of mule deer (Pierce et al. 1999). Home ranges often overlap spatially among and between sexes, although individuals tend to use overlapping areas at different times. Intraspecific aggression occurs, and males with unknown frequency kill or injure other males, females, subadults, and young (Logan and Sweanor 2001). In contrast, females apparently do not behave aggressively toward each other or conspecific young.

Regulation of Populations.—A system of land tenure and mutual avoidance with overlapping spatial distributions (Hornocker 1970, Seidensticker et al. 1973) generally has been an accepted paradigm for understanding regulation of populations of mountain lions. Confusion exists in published literature because limitation and regulation commonly are used as interchangeable terms. Any mortality factor that reduces rate of population growth is a limiting factor, whereas regulation infers a population level that is in approximate density-dependent equilibrium with long-term natality and mortality (Ballard et al. 2001). Density of mountain lions in Utah was regulated based on land tenure, but the population was limited by abundance of mule deer (Hemker et al. 1984). A population of mountain lions in California was limited by food supply (Pierce et al. 2000b), but social organization apparently did not limit a population of mountain lions in New Mexico below a level set by abundance of prey (Logan and Sweanor 2001). However, mutual interference among predators might provide an underlying mechanism producing non-uniform spatial distributions of both predators and prey (Alonso et al. 2002).

Interactions with Humans.—Preliminary findings indicated 48 fatal attacks, 177 nonfatal attacks, and 157 non-attack interactions between mountain lions and humans in North America since 1890 (Fitzhugh et al. 2003). Recorded encounters, injuries, and deaths involving mountain lions and humans have been increasing throughout the contiguous western United States, but causes of these trends are poorly understood (Mattson et al. 2003). Indications of potentially higher risk of attacks or encounters might include: 1) high abundance of mountain lions near urban areas, 2) intensive recreational use by humans in areas where mountain lions are present, 3) increased sightings of or close encounters with mountain lions, and 4) attacks on pets (Fitzhugh 1988). Children probably are more at risk than adults.

Reproduction

Estrus lasts from 1 to 10 days and the estrous cycle (time between estrus periods) can vary widely, ranging from about 14 to 35 days. In several studies, females bred first between 18 and 37 months old, whereas males bred first between 21 and 27 months of age (Maehr 1997, Ross and Jalkotzy 1992, Lindzey et al. 1994, Logan and Sweanor 2001). Mating pairs might remain together 1 to 4 days, and both sexes might be promiscuous. Older, resident males participate in all or most breeding (Seidensticker et al. 1973, Logan and Sweanor 2001).

Kittens are born throughout the year after a gestation period of about 91 to 92 days, but frequency of births generally peaks between about June and September (Anderson 1983, Logan and Sweanor 2001), corresponding with a peak breeding season of March to June. Intervals between births might be about 17 to 18 months or more, and reproductive success of females is highly variable. Kittens generally suckle until 6 to 8 weeks of age and are carnivorous and precocious after weaning. Females raise kittens alone and spend 14 to 19 months caring for them. About half of adult females in a population might have kittens each year, and about 75% might raise kittens during a given year (Logan and Sweanor 2001). Sex ratios of kittens, subadults, and adults statistically are 1:1, but there is a tendency for populations to have fewer adult and subadult males than females, likely reflecting greater dispersal and mortalities of males than females, and higher recruitment of females than males into populations (Seidensticker et al. 1973, Ross and Jalkotzy 1992, Logan and Sweanor 2001).

Survival

Estimates of survival of mountain lion kittens vary considerably among



Capturing, tranquilizing, attaching radiocollars to gather information on distribution and movements, and releasing captured animals unharmed are important aspects of mountain lion studies.

studies, primarily a result of differences in ages examined in relation to mortality, but survival rates might range from about 0.45 to 0.98 (Ross and Jalkotzy 1992, Maehr and Caddick 1995, Logan and Sweanor 2001). In one study, 36% of kittens died of natural causes, and most deaths occurred at equal to or less than 3 months of age. Infanticide by adult males was a major cause of death for kittens. Mountain lions produce an average litter of 3 kittens, but only 2 usually survive past weaning. In a New Mexico study (Logan and Sweanor 2001), survival of kittens appeared to be independent of decline in a mule deer population. Two female kittens in that study survived after being orphaned at 7.5 and 9.8 months of age.

Few studies have estimated survival of subadults, primarily because they often disperse from research areas. Estimated survival rate for males (56%) was lower than females (88%) in New Mexico (Logan and Sweanor 2001). Differences in rate of survival likely reflect philopatry (the tendency to return to a specific location to breed) of adult females and their female offspring, as well

as mortalities of subadults caused by adult males or other sources, patterns that are apparent in exploited and unexploited populations. Other research reported about 80% mortality of dispersing subadults (Beier 1995). Survival rate of adult mountain lions ranged from 69% to 92% in exploited populations and was 75% in unexploited populations (Lindzey et al. 1988, Anderson et al. 1992, Beier and Barrett 1993). Other studies reported survival rates of adults between 25% and 100% for females, compared to between 61% and 100% for males (Lindzey et al. 1988, Logan and Sweanor 2001). In comparison, survival rates of adult females and males in a heavily exploited population of mountain lions in Arizona were 67% and 58%, respectively (Cunningham et al. 2001). Survival rate for sexes combined in this study was lower than that reported in numerous states. Depredation control and sport hunting are primary sources of mortalities of adults in hunted populations (Cunningham et al. 2001), whereas natural mortality, largely from intraspecific strife, likely is a major cause of mountain lion deaths in both exploited and unhunted populations (Ruth et al. 1998, Logan and Sweanor 2001).

Management and Conservation

Mountain lions and other predators were exploited during the first half of the 20th century, primarily in attempts to benefit abundance of mule deer. This practice began to be questioned and generally was recognized as ineffective by the 1950s. Between 1906 and 1970, for example, 883 mountain lions were removed from about 1,200 square miles of Arizona's Kaibab Forest to benefit the mule deer herd. Abnormally high densities of deer developed during 2 predator control efforts between 1906 and 1923 (estimated removal of 0.03 mountain lions/square mile/year) and 1940 to 1963 (estimated removal of 0.002 mountain lions/square mile/year), but control efforts failed to sustain expected normal levels of production by mule deer (McCullough 1986). Most states and Canadian provinces enacted regulations between 1960 and 1970 to protect mountain lions, primarily by conferring game status and regulating harvests.

Legislative enactment in 1970 classified mountain lions as big game animals in Arizona (Cunningham et al. 1995), but harvest statewide differed little during years when bounties were paid (1952 to 1969) and the following two decades. Total harvest increased considerably from the 1990s through 2001 and reached 318 animals in 2002 (Arizona Game and Fish Department 2003a). Resource management agencies in western states and Canadian provinces rely on sustained yield harvesting through regulated sport hunting to manage mountain lions, lacking inexpensive, reliable methods at landscape scale to estimate sizes and monitor dynamics of populations relative to management prescriptions (Logan and Sweanor 2001).

However, effects of harvest differ depending on age and sex of animals killed, and proportion of adult females harvested likely is a key factor influencing

impacts of harvest on populations of mountain lions. Sex ratio of harvest alone likely is of limited value in identifying changes in abundance of populations. Numbers and age structure of harvested mountain lions should be combined to index effects of harvest on changes in populations. Increased levels of harvest in a study in Wyoming reduced abundance (about 50%) of mountain lions and corresponded with an increased proportion of adult females in the harvest from 14% to 41% (Anderson and Lindzey 2005).



Arizona Game and Fish Department personnel Tim Anderson (left) and Darren Tucker captured and released unharmed this adult mountain lion near Prescott, Arizona as part of an urban-related mountain lion study. The animal was restrained while anesthetized for the safety of the animal and the biologists.

Thus, although harvest of mountain lions in Arizona between 1990 and 2002 increased about 56% and females averaged about 49% of harvests (range of about 40% to 52%), data on age structure likely are necessary to evaluate impacts of harvest on the statewide population. Vulnerability of mountain lions to sport hunting might be highest for transient males, followed in descending order by resident males, transient females, and resident females (Anderson and Lindzey 2005). A population of mountain lions in a desert region of New Mexico consisted on average of 61% adults, 6% sub-adults, and 33% cubs (Logan and Sweanor 2001).

When 27% (6 males, 22 females) of mountain lions more than 1 year old were removed from a 733 square mile area during 1 sport-harvest season in Utah, abundance of adults recovered in about 9 months through replacement by transients and production of offspring by resident females, but the population likely would not have recovered as quickly from a second year's harvest of similar intensity. In contrast, harvest of about 19 mountain lions per year from 1988 to 1993 from an area less than 1,158 square miles in southeastern Arizona did not appear to reduce densities of mountain lions (Cunningham et al. 1995, 2001).

The hunting season for mountain lions in Arizona is yearlong, and unlimited tags are available over-the-counter (1 tag per hunter/year). Since 1970, the statewide bag limit in Arizona has been 1 mountain lion per hunter per year, but higher bag limits have been allowed or proposed and harvest objectives established in certain areas for purposes of management or research (Arizona Game and Fish Department 2003*b*). A hunter in areas with specific harvest objectives might be allowed to take 1 mountain lion per day until the harvest objective is reached. Once the harvest objective for an area is reached, the season can remain open and revert to a calendar year bag limit of 1 per hunter. Spatial controls can achieve high yields and avoid hazards of overharvest commonly associated with more broadly applied harvest quotas (McCullough 1996).

Most states, including Arizona, regulate harvest of females and kittens to some extent (Ruth et al. 2003). Most states and some Canadian wildlife resource agencies outlaw killing female mountain lions with kittens at their side. Seasons, bag limits, and harvest objectives in Arizona are in compliance with goals, mission, and authorities of the Arizona Game and Fish Department and Commission (Arizona Game and Fish Department 2001). The Arizona Game and Fish Department in 2003 initiated voluntary collection and submission of a mountain lion tooth by successful hunters to evaluate age structure of the harvest (Arizona Game and Fish Department 2003*b*). In July 2006, mandatory check-in was initiated for successful mountain lion hunters in Arizona. Hunters must report successfully taking a mountain lion within 48 hours of harvest. Mandatory physical check-in is required within 10 days of harvest, when hunters must present for inspection the head and the complete hide with evidence of sex attached. During inspections, biologists will assess physical and reproductive condition and evidence of disease, and collect a premolar tooth for age determination. Data collected from hunters will include harvest date and location, hunting method, and hunter effort (Arizona Game and Fish Department 2006).

Overharvest in some cases might be a potential threat to abundance of mountain lions (Logan and Sweanor 2001). Harvest and number of mountain lion hunting licenses sold might be correlated (Logan and Sweanor 2001), but this association was not apparent in Arizona between 1971 and 2001. Hunters using hounds in Arizona annually kill about 65% of harvested mountain lions, and about 15% of harvest is associated with depredation control (Arizona Game and Fish Department 2003*a*). Wildlife managers attempt to limit harvest of mountain lions to between 10% and 25% of estimated abundance, although reliable estimations of abundance of populations generally are lacking, and a sustainable annual harvest rate that allows for stability or growth of populations is uncertain (Lindzey et al. 1992, Ross and Jalkotzy 1992, Logan and Sweanor 2001). Although some believe sport hunters traditionally select big males for trophies (Logan and Sweanor 2001), ratio of males to females in Arizona's statewide harvest of mountain lions remained about 1:1 for the past 16 years. However, hunters using

hounds harvested 60% males, whereas those hunting without hounds harvested 60% females (Zornes et al. 2006).

Biologists in most states and provinces believe abundance of mountain lions increased in recent years, a conclusion perhaps based on upward trends in annual harvests by hunters (Logan and Sweanor 2001). However, recently perceived upward trends in abundance of mountain lions appear to be based more on subjective information than on empirical data (Hopkins 2003). Biologists presently lack reliable, cost-effective methods of monitoring trends of populations of mountain lions. In most western states, managers attempt to liberalize annual harvest until data suggest a population has declined (Logan and Sweanor 2001). Accelerating and expanding scope of conflicts between humans and large carnivores might have undermined resiliency mechanisms of these predators, and in some cases might have caused declines in abundance (Weaver et al. 1996).

Some states use track surveys to estimate trends of populations of mountain lions, but such surveys are problematic for 2 reasons—costs of sampling across large geographic areas are high, and indices derived from track surveys might be related only weakly to abundance (Smallwood and Fitzhugh 1995). Nonetheless, surveys of tracks and other sign might have potential for monitoring trends of mountain lion populations (Smallwood 1994, Smallwood and Fitzhugh 1995, Beier and Cunningham 1996, Harveson et al. 1999).

Little is known regarding habitat-specific spatial requirements of mountain lions in Arizona, but innate dispersal behavior of subadults suggests metapopulation structures are critical to persistence of viable populations. However, recent research indicates metapopulation structures may be questionable for mountain lions, and further research is needed. A metapopulation is a network of local subpopulations with some level of intermittent or regular movement of individuals (gene flow) among them. Identifying and delineating metapopulation components, including travel corridors and barriers to dispersal, are critical to conservation and management of mountain lions (Beier 1993, 1995; Sweanor et al. 2000). In the absence of immigration, habitat areas between 386 and 850 square miles likely are needed to support a persistent population of mountain lions consisting of 15 to 20 adults (Beier 1993). Limited immigration of even a few females and males per decade reduces this requirement to perhaps between 232 and 618 square miles. Fragmentation of connectivity corridors in metapopulations increases probability of extinction of small subpopulations.

Review of 4 studies evaluating relationships between predators and populations of deer in North America led to a conclusion that predators did not cause declines in abundance, but predation could limit populations of deer when they were below forage carrying capacity (Ballard et al. 2001). Seldom did predator reductions correspond with higher harvests of deer, but predation by mountain lions might cause significant mortality under certain conditions in some areas. Habitat carrying capacity, weather, human use patterns, number and type

of predator species, and habitat alterations potentially influence predator-prey relationships. Intensive radiotelemetry and manipulative research are crucial to advance our understanding of ecology of mountain lions, predation, and dynamics of predators and their prey (Ballard et al. 2001, Anderson and Lindzey 2003).

Predation by mountain lions possibly regulated herds of mule deer in California and Nevada (Bleich and Taylor 1998). In comparison, a study in New Mexico concluded predation by mountain lions represented compensatory mortality (i.e., it replaced other mortality factors) of mule deer (Logan and Sweanor 2001). Predation by mountain lions and drought limited the deer population; both variables influenced production and survival of fawns and growth of the population. Abundance of mountain lions in California followed but lagged decline of a population of mule deer under conditions of drought (Pierce et al. 2000b). A sustained low prey base might be required to trigger steep decline in survival of mountain lion kittens (Logan and Sweanor 2001).

Few studies have addressed population-level impacts of predation by mountain lions on bighorn sheep, although mountain lions are the only North American predator capable of population-level impacts on the species (Sawyer and Lindzey 2002). Predation by mountain lions might cause populations of bighorn sheep to decline and impede recovery, particularly if escape terrain is inadequate, and if abundance of preferred prey is low. Declines in abundance of mule deer might have contributed



Mountain lions recover quickly following capture and attachment of radiocollars, and within minutes resume activities.

to recent increases in predation of bighorn sheep in Arizona and other areas (Rominger and Weisenberger 1999, Kamler et al. 2002). Predation of bighorn sheep by mountain lions in Arizona represents a management concern (Kamler et al. 2002, McKinney et al. 2006a,b). Individual mountain lions usually are responsible for most predation within a given population of bighorn sheep, and predation varies among populations and between years. Mountain lions prey on adults and lambs, and population-level impacts might be compounded if mountain lions kill reproductive females and their offspring (Ross et al. 1997, Hayes et al. 2000, Sawyer and Lindzey 2002).

Experimental intervention or manipulation provides a useful approach to evaluate effects of environmental factors on populations. Experimental levels of harvest were used to evaluate relationships between harvest intensity and age structure of mountain lions (Anderson and Lindzey 2005). Another published

study used experimental reduction of mountain lions to evaluate relationships between the predator and dynamics of prey populations (Ballard et al. 2001). About half of a mountain lion population was translocated to determine effects on survival of mule deer in New Mexico (Logan and Sweanor 2001). More recently, Arizona Game and Fish Department studied a population of desert bighorn sheep (*O. c. mexicana*) that declined between 1994 and 1997, and experienced low lamb production through 1999 (McKinney et al. 2006b). Mountain lion reductions were used as an experimental element in evaluating various potential limiting factors affecting this population of bighorn sheep, including disease, nutritional status, predators, and levels of rainfall (McKinney et al. 2006b). Research suggested drought (ultimate factor), and nutritional status and predation by mountain lions (proximate factors) interacted to limit the population. Relative availability of ungulate prey per mountain lion likely affected diets of mountain lions, and fecal analyses demonstrated no predation or scavenging of bighorn sheep by bobcats and coyotes.

Density of mountain lions is related positively to abundance of cervids in habitats with stalking cover (Weaver et al. 1996). Predation might be unrelated to densities of mountain lions but related to relative abundance of alternate prey (Sawyer and Lindzey 2002). Density-dependent behavioral dynamics of predator-prey interactions influence predation, and mountain lions likely become relatively inefficient predators at higher population densities (Brown et al. 1999). Translocation of mountain lions likely has limited applicability for management (Ruth et al. 1998), but lethal removal of only a few mountain lions each year from an area might reduce extinction risk for a bighorn sheep population (Ernest et al. 2002). Importantly, studies relating to predation of bighorn sheep by mountain lions often have been merely incidental or observational, and typically have been conducted only when concerns about prey populations already existed. Thus, predation by mountain lions and population-level impacts of predation in general are poorly understood.

Research Needs

Inadequate knowledge and understanding of mountain lion ecology impairs effective conservation and management of the predator and its prey. Efforts to achieve persistence and prevent overkill of mountain lions are primary immediate concerns. Current needs of conservation and management of mountain lions in Arizona include:

- 1) Develop indices of sustainable harvest at landscape and local levels.
- 2) Evaluate relationships between intensity and age structure of harvest and utility of using proportional harvest of adult females to index changes in abundance of populations at habitat-specific and statewide levels. Current moun-

tain lion management guidelines address these issues through mandatory check-ins, collections of premolar teeth and age determinations, documentation of age, sex, and kill locations, and hunter questionnaire and hound hunter surveys in alternate years.

3) Determine habitat requirements, and obtain quantitative descriptions of existing habitat patches, habitat fragmentation, and habitat connectivity (travel corridors connecting local populations).

4) Develop methods to evaluate landscape-scale distribution and sizes of habitat patches and to assess and monitor habitat-specific abundance, distribution, and trends of populations relative to harvest prescriptions.

5) Evaluate predator-prey dynamics and environmental factors affecting these relationships.

6) Evaluate urban interface issues regarding interactions with humans.

Literature Cited

- Ackerman, B. B., F. G. Lindzey, and T. P. Hemker. 1984. Cougar food habits in southern Utah. *Journal of Wildlife Management* 48:147–155.
- Alonso, D., F. Bartumeus, and J. Catalan. 2002. Mutual interference between predators can give rise to Turing spatial patterns. *Ecology* 83:28–34.
- Anderson, A. E. 1983. A critical review of literature on puma (*Felis concolor*). Colorado Division of Wildlife Special Report No. 54, Denver.
- Anderson, A. E., D. C. Bowden, and D. M. Kattner. 1992. The puma on Uncompahgre Plateau, Colorado. Colorado Division of Wildlife Technical Publication No. 40, Denver.
- Anderson, C. R., and F. G. Lindzey, Jr. 2003. Estimating cougar predation rates from GPS location clusters. *Journal of Wildlife Management* 67:307–316.
- Anderson, C. R., Jr., and F. G. Lindzey. 2005. Experimental evaluation of population trend and harvest composition in a Wyoming cougar population. *Wildlife Society Bulletin* 33:179–188.
- Anderson, C. R., Jr., F. G. Lindzey, and D. B. McDonald. 2004. Genetic structure of cougar populations across the Wyoming Basin: metapopulation or megapopulation. *Journal of Mammalogy* 85:1207–1214.

Biological Foundation for Management of Mammalian Predators in Arizona

- Arizona Game and Fish Department. 2001. Wildlife 2006. Arizona Game and Fish Department, Phoenix.
- Arizona Game and Fish Department. 2003a. Hunt Arizona. Arizona Game and Fish Department, Information Branch, Phoenix.
- Arizona Game and Fish Department. 2003b. 2003–2004 Hunting Regulations. Arizona Game and Fish Department, Information Branch, Phoenix.
- Arizona Game and Fish Department. 2006. Arizona Hunting and Trapping Regulations. Arizona Game and Fish Department, Phoenix.
- Arjo, W. M., D. H. Pletscher, and R. R. Ream. 2002. Dietary overlap between wolves and coyotes in northwestern Montana. *Journal of Mammalogy* 83:754–766.
- Ballard, W. B., D. Lutz, T. W. Keegan, L. H. Carpenter, and J. C. deVos, Jr. 2001. Deer-predator relationships: a review of recent North American studies with emphasis on mule and black-tailed deer. *Wildlife Society Bulletin* 29:99–115.
- Beier, P. 1993. Determining minimum habitat areas and habitat corridors for cougars. *Conservation Biology* 7:94–108.
- Beier, P. 1995. Dispersal of juvenile cougars in fragmented habitat. *Journal of Wildlife Management* 59:228–237.
- Beier, P., and R. H. Barrett. 1993. The cougar in the Santa Ana Mountain Range, California. Orange County Cooperative Mountain Lion Study, Final Report.
- Beier, P., D. Choate, and R. H. Barrett. 1995. Movement patterns of mountain lions during different behaviors. *Journal of Mammalogy* 76:1056–1070.
- Beier, P., and S. C. Cunningham. 1996. Power of track surveys to detect changes in cougar populations. *Wildlife Society Bulletin* 24:540–546.
- Beier, P., and S. Loe. 1992. A checklist for evaluating impacts to wildlife movement corridors. *Wildlife Society Bulletin* 20:434–440.
- Bleich, V. C., and T. J. Taylor. 1998. Survivorship and cause-specific mortality in five populations of mule deer. *Great Basin Naturalist* 58:265–272.

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- Boyd, D., and B. O’Gara. 1985. Cougar predation on coyotes. *Murrelet* 66:17.
- Brown, J. S., J. W. Laundré, and M. Gurling. 1999. The ecology of fear: optimal foraging, game theory, and trophic interactions. *Journal of Mammalogy* 80:385–399.
- Cashman, J. L., M. Pierce, and P. R. Krausman. 1992. Diets of mountain lions in southwestern Arizona. *Southwestern Naturalist* 37:324–326.
- Culver, M., W. E. Johnson, J. Pecon-Slattery, and S. J. O’Brien. 2000. Genomic ancestry of the American puma (*Puma concolor*). *Journal of Heredity* 91:186–197.
- Cunningham, S. C., W. B. Ballard, and H. A. Whitlaw. 2001. Age structure, survival, and mortality of mountain lions in southeastern Arizona. *Southwestern Naturalist* 46:76–80.
- Cunningham, S. C., C. R. Gustavson, and W. B. Ballard. 1999. Diet selection of mountain lions in southeastern Arizona. *Journal of Range Management* 52:202–207.
- Cunningham, S., R. Ockenfels, S. Germaine, and J. Phelps. 1995. Predator/prey interactions. Unpublished Commission Briefing, Arizona Game and Fish Department, Phoenix.
- deVos, J. C., and T. McKinney. 2005. Recent trends in North American mountain lion populations. Pages 297–307 in C. van Riper and D. J. Mattson, editors. *The Colorado Plateau—Biophysical, socioeconomic, and cultural research*. University of Arizona Press, Tucson.
- Ernest, H. B., M. C. T. Penado, B. P. May, M. Syvanen, and W. M. Boyce. 2000. Molecular tracking of mountain lions in the Yosemite Valley region in California: genetic analysis using microsatellites and fecal DNA. *Molecular Ecology* 9:433–441.
- Ernest, H. B., E. S. Rubin, and W. M. Boyce. 2002. Fecal DNA analysis and risk assessment of mountain lion predation of bighorn sheep. *Journal of Wildlife Management* 66:75–85.
- Fitzhugh, E. L. 1988. Managing with potential for lion attacks against humans. *Proceedings of the Third Mountain Lion Workshop*, Prescott, Arizona.
- Fitzhugh, E. L., S. Schmid-Holmes, M. W. Kenyon, and K. Etling. 2003. Less-

Biological Foundation for Management of Mammalian Predators in Arizona

- ening the impact of a puma attack on a human. Seventh Mountain Lion Workshop, Jackson, Wyoming. (Abstract)
- Germaine, S. S., K. D. Bristow, and L. A. Haynes. 2000. Distribution and population status of mountain lions in southwestern Arizona. *Southwestern Naturalist* 45:333–338.
- Harverson, L. A., B. Route, F. Armstrong, N. J. Silvy, and M. E. Tewes. 1999. Trends in populations of mountain lions in Carlsbad Caverns and Guadalupe Mountains National Parks. *Southwestern Naturalist* 44:490–494.
- Hayes, C. L., E. S. Rubin, M. C. Jorgensen, R. A. Beta, and W. M. Boyce. 2000. Mountain lion predation of bighorn sheep in the Peninsular Ranges, California. *Journal of Wildlife Management* 64:954–959.
- Hemker, T. P., F. G. Lindzey, and B. B. Ackerman. 1984. Population characteristics and movement patterns of cougars in southern Utah. *Journal of Wildlife Management* 48:1275–1284.
- Hopkins, R. A. 2003. Mystery, myth and legend: the politics of cougar management in the new millennium. Seventh Mountain Lion Workshop, Jackson, Wyoming. (Abstract)
- Hornocker, M. G. 1970. An analysis of mountain lion predation upon mule deer and elk in the Idaho Primitive Area. *Wildlife Monographs* 21:1–39.
- Iriarte, J. A., W. L. Franklin, W. E. Johnson, and K. H. Redford. 1990. Biogeography variation of food habits and body size of the American puma. *Oecologia* 85:185–190.
- Kamler, J. F., R. M. Lee, J. C. deVos, Jr., W. B. Ballard, and H. A. Whitlaw. 2002. Survival and cougar predation of translocated bighorn sheep in Arizona. *Journal of Wildlife Management* 66:1267–1272.
- Koehler, G. M., and M. G. Hornocker. 1991. Seasonal resource use among mountain lions, bobcats, and coyotes. *Journal of Mammalogy* 72:391–396.
- Kunkel, K., and D. H. Pletscher. 1999. Species-specific population dynamics of cervids in a multi-predator ecosystem. *Journal of Wildlife Management* 63:1082–1093.
- Leopold, B. D., and P. R. Krausman. 1986. Diets of 3 predators in Big Bend National Park, Texas. *Journal of Wildlife Management* 50:290–295.

Mountain Lion

- Lindzey, F., B. B. Ackerman, D. Barnhart, and T. P. Hemmer. 1988. Survival rates of mountain lions in southern Utah. *Journal of Wildlife Management* 52:664–667.
- Lindzey, F. G., W. D. Van Sickle, B. B. Ackerman, D. Barnhart, T. P. Hemmer, and S. P. Laming. 1994. Cougar population dynamics in southern Utah. *Journal of Wildlife Management* 58:619–624.
- Logan, K. A., and L. L. Irwin. 1985. Mountain lion habitats in the Big Horn Mountains, Wyoming. *Wildlife Society Bulletin* 13:257–262.
- Logan, K. A., L. L. Irwin, and R. Skinner. 1986. Characteristics of a hunted mountain lion population in Wyoming. *Journal of Wildlife Management* 50:648–654.
- Logan, K. A., and L. L. Swenor. 2001. Puma. Pages 347–377 in S. Demerits and P. R. Krausman, editors. *Ecology and management of large mammals in North America*. Prentice Hall, Upper Saddle River, New Jersey.
- Maehr, D. S. 1997. *The Florida panther: life and death of a vanishing carnivore*. Island Press, Washington, D.C.
- Maehr, D. S., and G. B. Caddick. 1995. Demographics and genetic introgression in the Florida panther. *Conservation Biology* 9:1295–1298.
- Mattson, D. J., J. V. Hart, and P. Beier. 2003. A conceptual model and appraisal of existing research related to interactions between humans and pumas. Seventh Mountain Lion Workshop, Jackson, Wyoming. (Abstract)
- McCullough, C. Y. 1986. A history of predator control and deer productivity in northern Arizona. *Southwestern Naturalist* 31:215–220.
- McKinney, T., and B. McRae. 2001. Genetic analysis of feces for estimating size of a Sonoran Desert mountain lion population. *Desert Bighorn Council Transactions* 45:206–215.
- McKinney, T., and T. W. Smith. 2007. Diets of sympatric bobcats and coyotes during years of varying rainfall in central Arizona. *Western North American Naturalist* 67:8–15.
- McKinney, T., J. C. deVos, Jr., W. B. Ballard, and S. R. Boe. 2006a. Mountain lion predation of translocated desert bighorn sheep in Arizona. *Wildlife Society Bulletin* 34:1255–1263.

Biological Foundation for Management of Mammalian Predators in Arizona

- McKinney, T., T. W. Smith, and J. C. deVos, Jr. 2006*b*. Evaluation of factors potentially influencing a desert bighorn sheep population. *Wildlife Monographs* 164:1–36.
- Noss, R. F., H. B. Quigley, M. G. Doorknocker, T. Merrill, and P. C. Piquet. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. *Conservation Biology* 10:949–963.
- Pierce, B. M., V. C. Bleich, and R. T. Bowyer. 2000*a*. Selection of mule deer by mountain lions and coyotes: effects of hunting style, body size, and reproductive status. *Journal of Mammalogy* 81:462–472.
- Pierce, B. M., V. C. Bleich, and R. T. Bowyer. 2000*b*. Social organization of mountain lions: does a land-tenure system regulate population size? *Ecology* 81:1533–1543.
- Pierce, B. M., V. C. Bleich, C. B. Chetkiewicz, and J. D. Wehausen. 1998. Timing of feeding bouts of mountain lions. *Journal of Mammalogy* 79:222–226.
- Pierce, B. M., V. C. Bleich, J. D. Wehausen, and R. T. Bowyer. 1999. Migratory patterns of mountain lions: implications for social regulation and conservation. *Journal of Mammalogy* 80:986–992.
- Riley, S. J., and R. A. Malecki. 2001. A landscape analysis of cougar distribution and abundance in Montana, USA. *Environmental Management* 28:317–323.
- Robinette, W. L., J. S. Gashwiler, and O. W. Morris. 1959. Food habits of the cougar in Utah and Nevada. *Journal of Wildlife Management* 23:261–273.
- Robbins, C. T. 1983. *Wildlife feeding and nutrition*. Academic Press, New York, N.Y.
- Rominger, E. M., and M. E. Weisenberger. 1999. Biological extinction and a test of the “conspicuous individual hypothesis” in the San Andres Mountains, New Mexico. *North American Wild Sheep Conference* 2:293–309.
- Ross, P. I., and M. G. Jalkotzy. 1992. Characteristics of a hunted population of cougars in southwestern Alberta. *Journal of Wildlife Management* 56:417–426.
- Ross, P. I., M. G. Jalkotzy, and M. Festa-Bianchet. 1997. Cougar predation on

Mountain Lion

- bighorn sheep in southwestern Alberta during winter. *Canadian Journal of Zoology* 74:771–775.
- Ruth, T. K., K. A. Logan, L. L. Sweanor, M. G. Hornocker, and L. J. Temple. 1998. Evaluating cougar translocation in New Mexico. *Journal of Wildlife Management* 62:1264–1275.
- Sawyer, H., and F. Lindzey. 2002. A review of predation on bighorn sheep (*Ovis canadensis*). Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, Wyoming.
- Seidensticker, J. C., M. G. Hornocker, W. V. Wiles, and J. P. Messick. 1973. Mountain lion social organization in the Idaho Primitive Area. *Wildlife Monograph* No. 35.
- Shaw, H. G. 1979. A mountain lion field guide. Arizona Game and Fish Department Special Report No. 9, Phoenix.
- Shaw, H. G., N. G. Woolsey, J. R. Wegge, and R. L. Day, Jr. 1988. Factors affecting mountain lion densities and cattle depredation in Arizona. Arizona Game and Fish Department Research Branch Final Report, Phoenix.
- Smallwood, K. S. 1994. Trends in California mountain lion populations. *Southwestern Naturalist* 39:67–72.
- Smallwood, K. S., and E. L. Fitzhugh. 1995. A track count for estimating mountain lion *Felis concolor californica* population trend. *Biological Conservation* 71:251–259.
- Sweanor, L. L., K. A. Logan, and M. G. Hornocker. 2000. Cougar dispersal patterns, metapopulation dynamics, and conservation. *Conservation Biology* 14:798–808.
- Thompson, D. J., and J. A. Jenks. 2005. Long-distance dispersal by a subadult male cougar from the Black Hills, South Dakota. *Journal of Wildlife Management* 69:818–820.
- Toweill, D. E., and E. C. Meslow. 1977. Food habits of cougars in Oregon. *Journal of Wildlife Management* 41:576–578.
- Van Sickle, W. D., and F. G. Lindzey. 1991. Evaluation of a cougar population estimator based on probability sampling. *Journal of Wildlife Management* 55:738–743.

Biological Foundation for Management of Mammalian Predators in Arizona

Van Sickle, W. D., and F. G. Lindzey. 1992. Evaluation of road track surveys for cougars (*Felis concolor*). *Great Basin Naturalist* 52:232–236.

Weaver, J. L., P. C. Paquet, and L. F. Ruggiero. 1996. Resilience and conservation of large carnivores in the Rocky Mountains. *Conservation Biology* 10:964–976.

Zornes, M. L., S. P. Barber, and B. F. Wakeling. 2006. Harvest methods and hunter selectivity of mountain lions in Arizona. Pages 85–89 in *Proceedings of the Southwest Section of the Wildlife Society, Managing Wildlife in the Southwest*.

Chapter 5

Coyote

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Common name - Coyote
Scientific name - *Canis latrans*

Life History

Description

Male coyotes (*Canis latrans*) weigh between 17 and 44 pounds, and females weigh between 15 and 40 pounds. Males are taller and longer than females, although there is significant geographic variation among subspecies. Length of

head and body ranges from about 3.3 to 5.2 feet, and the tail is about 1.3 feet long; coyotes stand about 22 to 26 inches high at the shoulder. Coyotes have wide, pointed, erect ears, a tapering muzzle, and a black nose. The top of the muzzle forms an almost continuous line with the forehead. Paws have 4 toes with nonretractable claws, and are more elongated than those of similar-sized domestic dogs

(*C. familiaris*). Forepaws have a rudimentary “thumb” that is reduced to a claw and is located high on the inner side of the leg. Claws are not used in attack or defense; they are typically blunted from constant contact with the ground and often do not leave deep marks (Bekoff 1995).

Coyotes usually are smaller than Mexican wolves (*C. lupus baileyi*), but overlap in size with domestic dogs and red wolves (*C. rufus*). Coyotes can be differentiated from Mexican wolves by weight, length, and height at shoulder.



A coyote track on a dirt road in Sonoran Desert scrub habitat. Front tracks are generally 2 1/4– 2 3/4 inches long by 1 3/4–2 3/8 inches wide. Hind tracks measure 2–2 3/8 inches long by 1 1/2–1 7/8 inches wide.

Mexican wolves weigh 50 to 90 pounds, are 54 to 66 inches long, and stand 26 to 32 inches tall at the shoulder (D. Groebner, Arizona Game and Fish Department, unpublished data).

Distribution

Coyotes have been present in Arizona for about 2 million years, and currently are widely distributed throughout the state (Figure 1). Millions of years ago, early canid ancestors known as “miacids” evolved, splitting into 2 evolution-

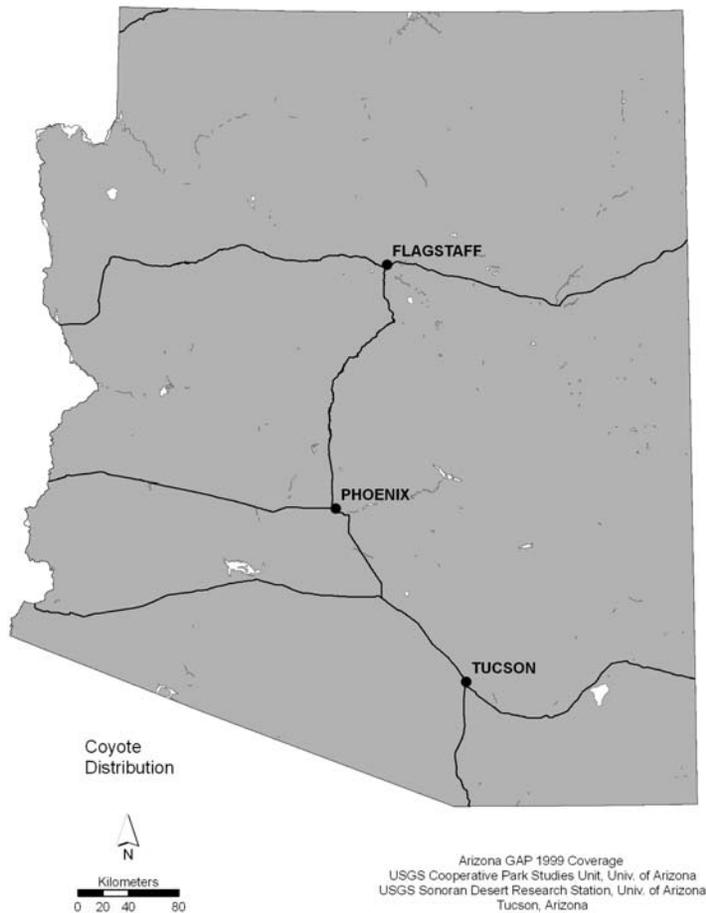


Figure 1. Distribution of coyotes in Arizona.

Coyote

ary lines, felines (cat family) and canids (dog family). These early canids were capable of adjusting to a variety of environments, and thus prospered. Despite climatic changes and geographical upheavals, many dog-like animals began to appear as the centuries passed. One that survived the Pleistocene Ice Age is now considered to be the immediate ancestor of modern day coyotes. Coyotes currently are found between northern Alaska and Costa Rica, and throughout mainland United States and Canada (Bekoff 1995). They are found throughout Arizona in all habitat types, and are the most widely distributed carnivore in Arizona (Arizona Game and Fish Department 2003).

Habitat

Coyotes are highly mobile and opportunistic predators that thrive in diverse habitat types, including arid, hot deserts, grasslands and plains, colder high elevation forests, and large urban cities (Kleiman and Eisenberg 1973, Bekoff 1982). Distribution of coyotes appears to be limited only by available water; coyotes in desert regions of Arizona must have access to free water (Golightly and Ohmart 1983). Decline of competitors such as gray wolves (*C. lupus*) and possibly mountain lions (*Puma concolor*) probably was an important factor contributing to range expansion by coyotes in the United States. Overgrazing by livestock and modified habitat conditions that resulted in more suitable habitat for small rodents on which coyotes prey also might have influenced changes in their abundance and distribution (Bekoff 1995).



Coyotes live in a variety of habitats throughout Arizona, including semi-desert grasslands grazed by livestock.

Diets and Prey

Coyotes are classified as carnivores, but they often are omnivorous and opportunistic with respect to food sources. Diets vary greatly seasonally and geographically, as do methods they use to capture prey. Coyotes are active predators, relying primarily on vision while hunting. They can fish, climb trees, and scavenge in pursuit of food. Coyotes are active throughout the day, but tend to be most active in early morning and near sunset. Although coyotes are very successful scavengers, particularly on large ungulates, they might not prey on large

animals unless alternative food sources are scarce or they encounter young, old, sick, or otherwise defenseless individuals (Bekoff 1982, 1995).

More than 1,800 fecal samples (scats) from coyotes were analyzed from ungrazed Three Bar Wildlife Area and grazed areas of Tonto National Forest in central Arizona (Horejsi 1982). Remains of prey (frequency of occurrence) found in scats from Three Bar Wildlife Area were: deer (*Odocoileus* spp.; 8.5%), javelina (*Pecari tajacu*; 1.0%), rabbits (23.1%), rodents (11.5%), livestock (0.6%), birds (5.6%), eggs (2.0%), reptiles (1.7%), insects (7.1%), and vegetation (37.2%). Occurrence of remains in scats from a grazed portion of Tonto National Forest were: deer (7.1%), javelina (0.2%), rabbits (16.5%), rodents (11.8%), livestock (4.1%), birds (5.1%), eggs (1.9%), reptiles (1.8%), insects (7.0%), and vegetation (42.2%). Densities of rabbits influenced rate of predation on deer fawns (Horejsi 1982).

On Anderson Mesa in Arizona, 11%–63% of coyote scats collected in spring during pronghorn (*Antilocapra americana*) fawning season contained remains of pronghorn, 21%–46% contained remains of elk (*Cervus elaphus*), and 14%–37% contained remains of rabbits and rodents (T. McKinney and M. Zornes, Arizona Game and Fish Department, unpublished data; Neff and Woolsey 1980). Insects and vegetation also were important foods during spring; birds, cattle, reptiles, and mule deer (*O. hemionus*) comprised minor parts of diets. On pronghorn winter range, cottontails (*Sylvilagus* spp.) and jackrabbits (*Lepus* spp.) were the most commonly occurring food items, followed by cattle, then rodents. Remains of pronghorn and mule deer occurred in less than 10% of scats, possibly reflecting scavenging (Neff and Woolsey 1980).

Throughout North America, coyotes consume a wide variety of foods year-round, but small mammals, deer fawns, plants, birds, and invertebrates comprise much of their diets during summer months. Larger wildlife such as deer (either prey or carrion), livestock carrion, or rabbits are more predominant in diets of coyotes during winter. In areas where diets of coyotes have high occurrence of deer, snow depth is an important factor—deeper snow corresponds with nutritionally stressed deer and reduced availability of cover. Coyotes hunt in pairs or larger groups during the winter mating cycle, increasing success in killing larger prey (Voigt and Berg 1987). Diets of coyotes and bobcats overlap in Arizona's deserts,



Counting coyote scats may provide useful insights into relative abundance and distribution of coyotes in different Arizona habitats. This scat was found on a gravel road in Sonoran Desert scrub habitat.

but the species partition food resources, and coyotes rely on large prey more than bobcats (McKinney and Smith 2007).

Coyotes hunt and locate prey using sight, but they also use hearing, and smell (Wells and Lehner 1978). Each sense is well developed, but movements of prey, particularly attempts to flee, trigger attack (Lehner 1976). Coyotes usually trot slowly and quietly while hunting. When prey are spotted, they often freeze, then pounce to attack, particularly when prey are small. Coyote pups as young as 32 days of age often exhibit this “stalk and pounce” behavior (Bekoff 1977). Coyotes quietly stalk larger or more distant prey until the “right time” for attack. When prey are located, coyotes appear to “lock onto the target,” switching from foraging or ranging (travel) mode to kill mode. During kill mode, it seems to be difficult to divert the attention of a coyote or dissuade it from attack. Researchers observed coyotes preying on domestic animals and noted this singular focus on selected prey, almost to the exclusion of awareness of extraneous stimuli (Lehner 1976; Bekoff 1977, 1982).

Behavior

Coyotes are one of the most adaptable animals in North America, and can modify their habit patterns, diets, and social dynamics to survive in a wide variety of habitats. In addition to innate behavioral traits, learned behaviors assist in adaptations to specific circumstances (Lehner 1976). Recent adaptations to urban and suburban habitats have spanned several generations, and might involve learned behaviors passed from parent to offspring (Lehner 1976). If such adaptations occur at different rates in various family groups, this could explain why coyotes seem to exhibit several different patterns of behavior in urban areas. Coyotes most closely adapted to contact with humans might dwell entirely within an urban area. Other coyotes might rest and den in wildland fringe areas, but enter urban areas for food and water. Coyotes enter urban areas more frequently as caution of humans lessens, and some apparently enter urban areas seasonally as transient, non-territorial animals (Shargo 1988).



Coyotes often inhabit urban environments where they seem to have little fear of humans.

Well-meaning citizens that seek to enhance wildlife viewing opportunities might be doing serious harm if they feed or provide abundant resources for

wildlife in urban areas. Such actions potentially contribute to wildlife populations that far exceed natural carrying capacity of an area. Supplemental feeding also can change an animal's natural instincts for finding food, and can change behavior towards humans that potentially leads to human-wildlife conflicts (Baker and Timm 1998). Feeding coyotes contributes to subsequent attacks on humans and their pets (Parker 1995). Coyotes in some areas have ceased to regard humans as threats, perceiving them instead merely as sources of food. Curtailment of sport hunting and target shooting around urban areas also has reduced opportunities for coyotes to learn wariness of humans (Jurek 1997). Animals must avoid destruction by natural enemies; thus it might benefit coyotes to be cautious of humans. Mountain lions and wolves are their only other natural enemies. In many urban areas, human behavior has contributed to less caution of humans by coyotes (Young and Jackson 1951).



The word coyote means "barking dog" and comes from the Nahuatl (Aztec) word cōyotl.

Lifespan

Life expectancy is between 8 and 12 years in the wild, but coyotes can live as long as 18 years in captivity. Mortality in the wild depends greatly on the level of exploitation to which a population is exposed (Bekoff 1982, 1995). Mortality among age classes likely is high for coyotes 1 year old or younger. Annual mortality of coyotes more than 1 year old in lightly exploited and unexploited populations in Texas can be 40%, but survival increases between about 4 and 8 years of age (Knowlton 1972). The mortality rate of coyotes from 0 to 1 year old in Alberta, Canada was 71%; coyotes older than 1 year experienced mortality rates ranging from 36% to 42% (Nellis and Keith 1976).

In general, mortality of adult coyotes tends to be greater among younger age classes (1 to 2 years of age) and relatively older animals (more than 8 years old). Conversely, survival appears to be high among coyotes between 3 and 7 years of age, especially among individuals that maintain associations with territorial groups. Mortalities of coyotes are linked closely to human activities—directly from exploitation (e.g., hunting, trapping), and indirectly through collisions with

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automobiles and encounters with domestic dogs (Knowlton and Stoddart 1983). Coyotes have relatively few predators in the wild, and humans cause up to 90% of deaths in coyotes more than 5 months old, either purposefully with firearms and traps, or accidentally with vehicles. Wolves, black bears (*Ursus americanus*), mountain lions, and golden eagles (*Aquila chrysaetos*) can prey on coyotes (Voigt and Berg 1987).

Reproduction

Coyotes are almost always monogamous, and pair bonds can last more than 4 years. When coyotes are in packs, usually only 1 pair mates per season. Males and females are able to breed during their first year of life, usually when about 9 to 10 months old. Females typically have 1 estrous cycle (periods of sexual receptivity) per year, and males seem to go through an annual cycle of spermatogenesis (sperm production). Estrus (sexual receptivity) lasts about 2 to 5 days, and ovulation occurs about 2 to 3 days before the end of female receptivity. Courtship can last for as long as 2 to 3 months before mating takes place. Coyotes typically breed in February and March, gestation averages 63 days, mean litter size is 6 pups, and sex ratio at birth is 1 male:1 female. Size of litters can vary with intensity of exploitation, densities of coyotes, and abundance of prey. Pups are born blind and helpless, usually in an excavated den. Pups emerge from dens at about 2 to 3 weeks of age, and are weaned at about 5 to 7 weeks of age. Pups can eat some solid foods by 10 to 12 days old, and eat large pieces of meat by 1 month of age. By 6 months of age, pups have permanent teeth, are nearly adult size, and are no longer dependent on adults for food. At this point, most young disperse, but some might remain with the family pack throughout winter (Bekoff 1982, 1995).

Movements

Dispersal of young-of-the-year coyotes from natal areas usually occurs between 4 and 10 months of age, although some individuals disperse after they are 1 year old. There appear to be no sex-related differences in proportions of individuals that disperse or in time periods or distances individuals travel before establishing resident home ranges. Dispersing individuals generally suffer much greater mortality than sedentary peers that remain in natal groups in which they were born (Bekoff and Wells 1986). Causes of dispersal are unknown, although there is an apparent association between dispersal and availability of food. If adequate food is available to a mated pair, competition among individuals and likelihood of dispersal are reduced, and social bonds are more likely to form (Bekoff 1977, White and Harris 1994).

Availability of food is a major factor influencing social organization and territoriality. When packs form, members share defense of food and territory (Bekoff and Wells 1986). Typically, only pack members defend territories; solitary

and paired individuals do not (Bekoff 1992). Generally, it requires an average of more than 2 individuals to deter intruders successfully (Bekoff and Wells 1986). Coyotes have a network of frequently used trails or “runways” within their home ranges. Trails might lead to and from den sites, watering points, and landmark features. Along runways, coyotes mark scent posts, visit them regularly, and use them to establish territorial boundaries, ostensibly providing information about sex of territory occupants. Coyotes urinate at scent posts and often defecate and scratch nearby ground with their hind feet. Scent posts can be used as indicators of coyote abundance (Woolsey 1985).



Coyotes are found throughout the state, including the Chiricahua Mountains in southeastern Arizona.

It is difficult to generalize sizes of coyote home ranges because of inconsistencies among studies in sampling design and analysis of data (Laundré and Keller 1984). In general, sizes of home ranges of females and males seem similar, but sizes of home ranges differ regionally and seasonally with respect to temperature, food supply, reproductive status females, and presence of pups. Size of home range also varies according to age and whether or not an animal is transient or a more sedentary member of a pack (Bekoff 1995). In various studies, female and male home ranges have ranged seasonally from 3.8 to 15.7 square miles and 4.4 to 30.0 square miles, respectively. Estimates of sizes of annual home ranges of females and males have ranged from 2.6 to 35.3 square miles and 3.7 to 28.3 square miles, respectively (Laundré and Keller 1984).

Disease

Parasites and diseases sometimes cause deaths in populations of coyotes. Outbreaks of sarcoptic mange, an infestation by microscopic mites that causes thickening of the skin, loss of hair, and itching, commonly occur in Arizona. Heartworms and hookworms also are common parasites of coyotes. Coyotes also can be susceptible to diseases such as distemper, canine hepatitis, rabies, and parvovirus (Bekoff 1982, Voigt and Berg 1987).

Management and Conservation

History

Coyotes are classified as predators in Arizona and can be harvested throughout the year by anyone possessing a hunting license. Coyotes can be hunted statewide during daylight hours with an unlimited bag and possession limit. During the 1980s, when fur prices were at an all time high, predator callers and trappers added significantly to harvest of coyotes (Arizona Game and Fish Department 2003). In 1993, an initiative banned trapping on public lands within Arizona; numbers of licensed trappers declined but already had decreased because of low fur prices. Since 1981, annual harvest of coyotes in Arizona has exceeded harvest of other nongame predators and furbearers. During recent years (1990 to 2002), harvest in Arizona varied considerably, with trappers taking between 178 and 2,683 coyotes per year, and other sportsmen harvesting between 17,075 and 45,781 coyotes annually. Numbers of trappers and hunters during this period ranged from about 30 to 200 and 9,000 to 15,000, respectively, and harvest data suggest no clear trends in or effects of levels of sport harvest on abundance of coyotes during recent decades.



Wildlife managers need to estimate presence and abundance of wildlife, but many carnivores are secretive, more active at night, or occur at low densities. Therefore, biologists use scent stations along transects to determine long-term changes in abundance and distribution of coyotes in various habitats.

Population Trends

Currently, there are no protocols followed for collecting data statewide

on coyotes in Arizona; therefore, there are no indicators of population trends (Woolsey 1985), other than estimates of annual harvests and hunter effort. However, age structure of coyote populations might be a useful management tool for evaluating trends of populations. An influx of young coyotes in a population might mean an increasing coyote population. Monitoring changes in age structure can be used in predicting future population trends in coyote populations (Woolsey 1985). Models of population dynamics are useful for estimating responses of populations of coyotes to exploitation and hypothetical birth intervention techniques (Connolly and Longhurst 1975). Scent-station surveys likely could be used effectively to index long-term trends in relative abundance of coyotes (Connolly and Longhurst 1975).

Monitoring Techniques

Scat counts were an effective method for indexing relative abundance of common gray foxes (*Urocyon cinereoargenteus*), coyotes, and bobcats (*Lynx rufus*) in burned and unburned chaparral forests (Cunningham et al. 2001). Scats were removed from permanent transects 0.6 miles long on unmaintained roads or hiking trails, then counted

10 to 21 days later. However, scent-station surveys might be the most widely used method to index relative abundance of coyote populations (Linhart and Knowlton 1975). Several hundred scent station transects were developed in 17 western states. Each transect consisted of 50 scent stations located at 0.3-mile intervals along a continuous 14.7-mile route. Each scent station consisted of a



Biologists capture, tag, and release coyotes to monitor movements and activities at different times of year.

perforated plastic capsule containing a fermented-egg attractant that was placed centrally in a 1-yard circle of sifted dirt. Visitations by coyotes based on tracks were recorded daily for 5 consecutive days during September to index relative abundance of coyotes, and indices were compared between states, regions, and years. More recently, scent-station surveys have been modified to use commercial scent attractants, shorter transects, and single nights for measuring visitation by coyotes (Sargeant et al. 1998).

Population Ecology

Density dependent variations in natality and survival likely are strong compensatory responses of coyote populations in response to exploitation. Levels of harvest inflicting less than 50% mortality thus likely would not lead to decline of populations (Sterling et al. 1983). Populations of coyotes likely are regulated by interactions between social organization and availability of food (Gese et al. 1989).

Coyotes seem to rely primarily on rabbits and rodents for food, which permits maintenance of higher ratios of coyotes to ungulates than could exist if ungulates were the sole or primary food source. When coyotes are numerous, they might inflict serious losses in populations of ungulates for a brief period of time, if weather conditions such as drought temporarily increase vulnerability of prey. Abundance of alternative prey species ostensibly reduces predation by coyotes on ungulates (Cunningham et al. 1995).

Predation might enhance adjustment of ungulate numbers to carrying capacity of a habitat, rather than being merely a proximate limiting factor (Connolly 1978). Low survival of young is the most common indication of predation by coyotes as a limiting factor. Predation by coyotes at times can be a primary cause of mortality of young ungulates; coyote predation was the predominant cause of mortality of pronghorn fawns on Anderson Mesa in Arizona (Neff and Woolsey 1979). Similarly, coyotes potentially prey on deer fawns and elk calves (Hamlin et al. 1984, Gese and Grothe 1995).

Coyotes also potentially influence structure of communities of sympatric fauna. Experimental removal of coyotes in Texas resulted in higher abundance of rodents, black-tailed jack rabbits (*Lepus californicus*), badgers (*Taxidea taxus*), bobcats, and common gray foxes on treatment sites (Henke and Bryant 1999).

Research Needs

Studies are needed in Arizona to develop survey methods to index trends in coyote abundance, and to evaluate reproduction, and dispersal and survival of juveniles, particularly in relation to intensities of harvest and specific habitats. Little is known about effects of harvest on abundance and productivity of coyote populations in Arizona, and about relationships between populations of bobcats and coyotes relative to habitat and unpredictable changes in climate and weather. Wildlife managers in Arizona need effective survey methods to develop long-term, habitat-specific monitoring of relative abundance of coyotes, and to enhance understanding of impacts of different levels of harvest. Current coyote management guidelines in Arizona address research needs, and include:

- 1) Develop and implement standardized statewide survey methods to index abundance and trends;

2) Develop an annual harvest survey; and

3) Evaluate age structure of harvested coyotes using jaws submitted by hunters and trappers.

Literature Cited

- Arizona Game and Fish Department. 2003. Hunt Arizona. Arizona Game and Fish Department, Information Branch, Phoenix.
- Baker, R. O., and R. M. Timm. 1998. Management of conflicts between urban coyotes and humans in southern California. *Proceedings of the 18th Vertebrate Pest Conference* 18:299–312.
- Bekoff, M. 1977. Mammalian dispersal and the ontogeny of individual behavioural phenotypes. *American Naturalist* 111:715–732.
- Bekoff, M. 1982. Coyote. Pages 447–459 *in* J. A. Chapman and G. A. Feldhamer, editors. *Wild Mammals of North America*. Johns Hopkins University Press, Baltimore, MD.
- Bekoff, M. 1995. Coyotes: victims of their own success. *Canid News* 3:1–6.
- Bekoff, M., and M. C. Wells. 1986. Social ecology and behaviour of coyotes. *Advances in the Study of Behaviour* 16:251–338.
- Connolly, G. E. 1978. Predators and predator control. Pages 369–394 *in* J. L. Schmidt and D. L. Gilbert, editors. *Big Game of North America*. Stackpole Books, Harrisburg, PA.
- Connolly, G. E., and W. M. Longhurst. 1975. The effects of control on coyote populations. University of California, Davis, Division of Agricultural Sciences Bulletin 1872. 37 pp.
- Cunningham, S. C., L. A. Haynes, C. Gustavson, and D. D. Haywood. 1995. Evaluation of the interaction between mountain lions and cattle in the Aravaipa-Klondyke area of southeast Arizona. Arizona Game and Fish Department Technical Report No. 17, Phoenix.
- Cunningham, S. C., L. M. Monroe, L. Kirkendall, and C. L. Ticer. 2001. Effects of the catastrophic Lone Fire on low, medium, and high mobility wildlife species. Arizona Game and Fish Department Research Branch

Technical Guidance Bulletin No. 5, Phoenix. 51 pp.

- Gese, E. M., and S. Grothe. 1995. Analysis of coyote predation on deer and elk during winter in Yellowstone National Park, Wyoming. *American Naturalist* 133:36–43.
- Gese, E. M., O. J. Rongstad, and W. R. Mytton. 1989. Population dynamics of coyotes in southeastern Colorado. *Journal of Wildlife Management* 53:174–181.
- Golightly, R. T., Jr., and R. D. Ohmart. 1983. Metabolism and body temperature of two desert canids: coyotes and kit fox. *Journal of Mammalogy* 64:624–635.
- Hamlin, K. L., S. J. Riley, D. Pyrah, A. R. Dood, and R. J. Mackie. 1984. Relationships among mule deer fawn mortality, coyotes, and alternate prey species during summer. *Journal of Wildlife Management* 48:489–499.
- Henke, S. E., and F. C. Bryant. 1999. Effects of coyote removal on the faunal community in western Texas. *Journal of Wildlife Management* 63:1066–1081.
- Horejsi, R. G. 1982. Mule deer fawn survival on cattle-grazed and ungrazed desert ranges. Arizona Game and Fish Department Federal Aid Project Report, Phoenix.
- Jurek, R. 1997. If you love animals don't feed them. *Outdoor California* 58(3):4–6.
- Kleiman, D. G., and J. F. Eisenberg. 1973. Comparisons of canid and felid social systems from an evolutionary perspective. *Animal Behaviour* 21:637–659.
- Knowlton, F. F. 1972. Preliminary interpretations of coyote population mechanics with some management implications. *Journal of Wildlife Management* 36:369–382.
- Knowlton, F. F., and L. C. Stoddart. 1983. Coyote population mechanics: another look. Pages 93–117 *in* F. L. Bunnell, D. S. Eastman, and J. M. Peek, editors. *Symposium on Natural Regulation of Wildlife Populations*. Proceedings of the Northwest Section of the Wildlife Society. Vancouver, B.C., Canada.
- Laundré, J. W., and B. L. Keller. 1984. Home-range size of coyotes: a critical

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review. *Journal of Wildlife Management* 48:127–139.

Lehner, P. N. 1976. Coyote behavior: implications for management. *Wildlife Society Bulletin* 4(3):120–126.

Linhart, S. B., and F. F. Knowlton. 1975. Determining the relative abundance of coyotes by scent station lines. *Wildlife Society Bulletin*, Vol. 3, No 3, Fall 1975.

McKinney, T., and T. W. Smith. 2007. Diets of sympatric bobcats and coyotes during years of varying rainfall in central Arizona. *Western North American Naturalist* 67:8–15.

Neff, D. J., and N. Woolsey. 1980. Coyote predation on neonatal fawns on Anderson Mesa, Arizona. *Proceedings of the Pronghorn Antelope Workshop* 9:80–93.

Nellis, C. H., and L. B. Keith. 1976. Population dynamics of coyotes in central Alberta, 1964–68. *Journal of Wildlife Management* 40:389–399.

Parker, G. 1995. *Eastern coyote: the story of its success*. Nimbus Publ., Halifax, Nova Scotia.

Sargeant, G. A., D. H. Johnson, and W. T. Berg. 1998. Interpreting carnivore scent-station surveys. *Journal of Wildlife Management* 62:1235–1245.

Shargo, E. S. 1988. Home range, movements, and activity patterns of coyotes (*Canis latrans*) in Los Angeles suburbs. Ph. D. dissertation, Univ. of California-Los Angeles.

Sterling, B., W. Conley, and M. R. Conley. 1983. Simulations of demographic compensation in coyote populations. *Journal of Wildlife Management* 47:1177–1181.

Voigt, D. R., and W. E. Berg. 1987. Coyote. Pages 344–357 in *Wild Furbearer Management and Conservation in North America*. Ontario Ministry of Nat. Res. Ontario.

Wells, M. C., and P. N. Lehner. 1978. The relative importance of the distance senses in coyote predatory behavior. *Animal Behaviour* 26:251–258.

White, P. L. C., and S. Harris. 1994. Encounters between red foxes (*Vulpes vulpes*): implications for territory maintenance, social cohesion, and dis-

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persal. *Journal of Animal Ecology* 63: 315–327.

Woolsey, N. G. 1985. *Coyote field guide*. Arizona Game and Fish Department Special Report 15, Phoenix.

Young, S. P., and H. H. T. Jackson. 1951. *The clever coyote*. Wildlife Management Institute. University of Nebraska Press, Lincoln, NE.

Chapter 6

Kit Fox

Jim deVos, Research Branch
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Common name - Kit fox

Scientific name - *Vulpes velox macrotis*

Life History

Description

Kit foxes (*Vulpes velox macrotis*) are the smallest canids in North America. Arid-land foxes initially comprised 2 similar but separate species, kit fox (*Vulpes macrotis*) and swift fox (*Vulpes velox*). Recent studies (Hall 1981) grouped the 2 foxes within a single species (*Vulpes velox*), comprised of 2 subspecies, *V. v. macrotis* (kit fox) and *V. v. velox* (swift fox).

Adult kit foxes usually weigh between about 3 and 5 pounds. Color of the coat varies seasonally, but adults have abundant white-tipped guard hairs that give them a “grizzled” appearance. Overall coloration is pale and ranges from buff to gray, and the tip of the bushy tail is black (Hoffmeister 1986). Fur on shoulders and front legs of adults is distinctly reddish, the tail is grayish black and darkest on the tip, and underparts are white. Length of head and body is about 1.2 to 1.7 feet, and length of the tail is about 0.7 to 1.1 feet. Sex-related differences in size and weight are not marked, but males are heavier than females. In Arizona, 7 kit foxes (sexes combined) weighed an average of 3.8 pounds; males averaged 4.0 pounds and females averaged 3.7 pounds (Zoellick and Smith 1992). Ears appear large relative to size of body and head, and are larger than those of other small North American canids (McGrew 1979). At about 1 month of age, kit foxes are covered with short, wooly fur, and have long black guard



Kit foxes are small canids that rely on underground dens more than coyotes or larger species of foxes.



hairs showing through brown fur on the back and hips (Egoscue 1956). Well-defined tracks of kit foxes in dust are similar in appearance and size to tracks of common gray foxes (*Urocyon cinereoargenteus*), show 4 toes with claw marks, and are about 1.1 to 1.8 inches long and about 1.1 to 1.5 inches wide.

Distribution

Kit foxes inhabit arid regions and are most common in low elevation, desert-like habitats in the western United States (Dragoo et al. 1990). Kit foxes in Arizona are distributed widely in southern areas of the state, and in northeastern and western regions (Figure 1). Kit foxes primarily inhabit desert scrub and grasslands in much of southern Arizona, and north along desert regions associated with the Colorado River (Hoffmeister 1986). Taxonomic nomenclature is controversial regarding subspecies of kit fox; a possible second subspecies (*V. m. neomexicana*) inhabits Great Basin deserts and grasslands in northeastern Arizona and grasslands in southeastern Arizona, and *V. velox arsipus* (Cockrum 1960, Dragoo et al. 1990) and *V. v. macrotis* might be used as appropriate subspecific designations (Hoffmeister 1986).



These drawings of kit fox tracks emphasize the presence of claw marks.



Kit foxes inhabit low elevation deserts with flat terrain and sparse vegetation in the Southwest.

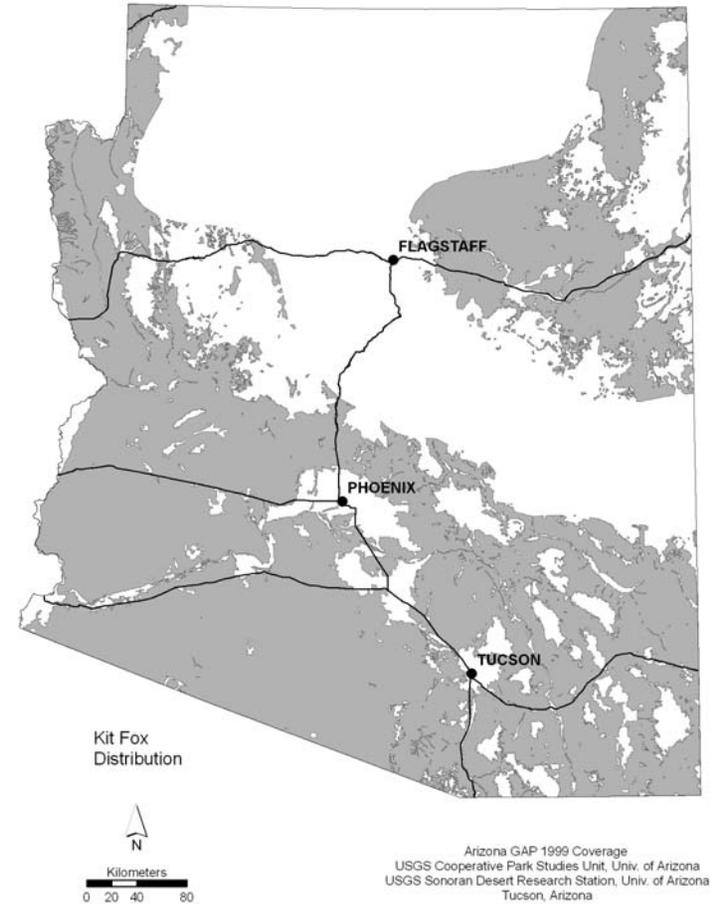


Figure 1. Distribution of kit foxes in Arizona.

Habitat

Throughout their range, kit foxes are primarily associated with desert shrub or shrub-grass habitats (McGrew 1979). Kit foxes tend to den and rest in areas where predators such as bobcats (*Lynx rufus*) or coyotes (*Canis latrans*) lack concealment cover; for example, areas with flat terrain and sparse vegetation (Egoscue 1962, Zoellick et al. 1989). Desert riparian areas might provide important foraging areas for kit foxes, because these areas have greater abundance of prey (Zoellick et al. 1989). Although riparian corridors were important in 1 study

area, kit foxes used flat areas dominated by creosotebush (*Larrea tridentata*) more than expected and used riparian corridors less than expected, based on availability of habitat types, time kit foxes spent in each habitat, and relocation frequencies of marked kit foxes (Zoellick et al. 1989).

Availability of water sources does not seem to be necessary within home ranges of kit foxes; many dens in 1 study were long distances from permanent water sources. Kit foxes likely meet requirements for water through metabolic processes (Egoscue 1956). In Death Valley, California, dens were located in flat areas dominated by low growing vegetation, whereas kit foxes hunted primarily in areas of sand dunes less than 1 mile from den sites (Grinnell et al. 1937). Availability

of dens and den sites are important features of kit fox habitats, and suitable den sites often are occupied for consecutive years (Egoscue 1956). Entrances to dens frequently are located near small shrubs or bushes, where kit foxes can rest in shade and be close to the den to avoid predators. Veg-



Kit fox habitat in sparse shrub vegetation in northwestern Arizona.

etation density at 22 den sites and at 22 paired, randomly selected sites indicated creosotebush was more dense at den sites than at random sites (Golightly 1981). Total cover of vegetation was greater at den sites, and soil characteristics differed between den sites and other locations (Golightly 1981). More gravel (particles more than 0.08 inches diameter) and clay (particles less than 0.00008 inches diameter) were present at surfaces of den sites and at a depth of 5.9 inches.

Individual dens in western Arizona were non-randomly located. Fewer dens were located less than 328 feet and more than 656 feet from riparian areas; most were established between 328 and 656 feet from riparian areas, differing from expected random distributions (Zoellick et al. 1989). This distribution pattern of dens likely allowed kit foxes optimal access to a greater abundance of prey species found in riparian areas. However, den sites of kit foxes consistently were located in areas with low density and height of vegetation, ostensibly to enhance avoidance of other predators. Kit foxes occupied habitats used only infrequently

by larger predators, presumably to avoid adverse interspecific interactions, i.e., interactions between kit foxes and other predators (Golightly 1981, Zoellick et al. 1989).

Dens of kit foxes likely are not much deeper than 5 feet. Tunnels within dens are likely small, seldom exceeding 8 inches high and 7 inches wide, and

dens might have from 1 to 5 or more entrances. Tunnels within a den complex might be up to about 61 feet long (Golightly 1981). Thermal environment in a den likely is quite stable and has been measured at about 71.6 °F (Golightly 1981). Seven adult kit foxes monitored with telemetry equipment used a total of 51 subterranean dens; individuals used several separate dens.

Body temperatures of kit foxes varied throughout the day, but were consistent during summer and winter. Resting body temperatures were 100.0 °F in summer and 100.2 °F in winter. Body temperatures increased when kit foxes were active and decreased as activity levels declined. Body temperature was lowest at mid-day (1 p.m.), and was highest during evening hours between 8 and 10 p.m. (Golightly 1981).

Dens are particularly important to kit foxes for conservation of water in summer and energy in winter (Golightly 1981). Reliance on dens as refugia and centers of activity makes kit foxes behaviorally inflexible and predictable with respect to requirements for suitable denning areas. Suitable den sites are important components of their home ranges. They selectively locate and dig dens in comparatively limited sites with respect to locations and soil characteristics. Thus, suitability of den sites is an important consideration for developing management plans. Because habitats occupied by kit foxes differ throughout their range of potential distribution, further research is needed to identify and quantify habitat requirements and abundance of kit foxes.

Diets and Prey

Diets of kit foxes consist primarily of small mammals (particularly rodents), but kit foxes are opportunistic predators and potentially prey on numerous



Kit fox dens are usually located near clumped shrubs in the Lower Sonoran Desert in Arizona.

species of mammals, birds, reptiles, and insects. Diets in Utah included black-tailed jackrabbits (*Lepus californicus*), Ord's kangaroo rats (*Dipodomys ordii*), deer mice (*Peromyscus maniculatus*), burrowing owls (*Speotyto cunicularia*), horned larks (*Eremophila alpestris*), meadowlarks (*Sturnella neglecta*), brown-sided lizards (*Uta stansburiana*), and sand crickets (*Stenopelmatus* spp.) (Egoscue 1956). Merriam's kangaroo rats (*D. merriami*), pocket mice (*Perognathus* spp.), and white-throated wood rats (*Neotoma albigula*) are common nocturnal species in areas inhabited by kit foxes in western Arizona (Zoellick et al. 1989), and are the most common items in diets of kit foxes inhabiting creosotebush flats in central Arizona (Fisher 1981). Diets of kit foxes also included 2 species that are active during daylight hours and commonly occur at high densities during spring and summer—round-tailed ground squirrels (*Spermophilus tereticaudus*) and rabbits (43% and 39% of diets, respectively). Remains of cottontail rabbits (*Sylvilagus auduboni*) also have been found at sites of kit fox dens (Zoellick et al. 1989).



Kit foxes are almost entirely nocturnal and spend most daylight hours resting in underground dens.

Behavior

Kit foxes are almost entirely nocturnal (Morrell 1972, Zoellick 1990). Near Tonopah, Arizona, kit foxes were active in only 4% (15 of 385) of locations during daylight hours; remaining locations indicated animals were inactive and within or loafing at the mouths of dens (Zoellick 1990). Kit foxes were more active during the first 3-hour period after sunset than during other times (Zoellick 1990). In another study during winter, kit foxes were most active during the first 3 hours after sunset, and again were active in the last 3 hours prior to sunrise (Golightly 1981). Activity during summer months was more evenly distributed throughout the night.

Male and female kit foxes both moved long distances at night (Zoellick

Kit Fox

et al. 1989), but average movement of males (8.89 miles) was greater than that of females (7.33 miles). Males and females moved greater distances during the breeding season than during other biological seasons (Zoellick et al. 1989). Similar to other canids, kit foxes produce a series of alarm, fear, or distress sounds, including barking, to alert other kit foxes of danger, and growling to intimidate other kit foxes or other canid species.

Reproduction

Based on radiotelemetry studies, kit foxes often breed during December and January, and pups are born during February and March after a 2-month gestation period (Morrell 1972). In Utah, young usually were born during March or April (Egoscue 1956). Pups are independent of their parents at 4 to 5 months of age (Hoffmeister 1986). Pair formation begins during October and November, and pairs remain together at least until young are weaned (Morrell 1972), when adults usually separate until the following season of pair formation. However, kit foxes might be monogamous (pair members remain together until 1 is lost) (Egoscue 1956).

Males seem to kill and provide most food for females and litters during pup rearing (Egoscue 1956). Females are very attentive to young pups. In general, litters range from 3 to 6 pups, but very little information is available regarding litter sizes of kit foxes in the Southwest. A family of kit foxes that included 6 young reportedly were moved nightly in response to rising waters of newly created Lake Mead (Grater 1939).

Home Range, Density and Dispersal

Three female and 4 male kit foxes in Arizona were equipped with radio collars from December 1982 to March 1984 (Zoellick and Smith 1992). Home range size was calculated using a grid-cell method. Mean size of home range for males and females combined was 4.32 square miles. Average sizes of home range for females and males in Arizona were 3.78 and 4.75 square miles, respectively, greater than sizes of home ranges reported in Utah (Daneke et al. 1984) and California (Morrell 1972). Differences may have reflected lower densities of prey on the study area in Arizona than on study areas in the other states (Zoellick and Smith 1992). However, sizes of individual home ranges in California in another study averaged 4.5 square miles, and home range size was not affected by differences in abundance of prey. This suggests that maintenance of large and relatively non-overlapping home ranges might represent adaptation to drought-induced periods of prey scarcity (White and Ralls 1993).

Home ranges of males in western Arizona during the breeding season overlapped considerably (Zoellick and Smith 1992) because of movements by males to den sites of nearby pairs. However, female home ranges did not overlap, suggesting restricted movements. In contrast, home ranges of neither females or

males overlapped in another study (Daneke et al. 1984). Den sites were abandoned in Utah (Egoscue 1956), likely caused by either depletion of food resources in the immediate vicinity of the den site, or by significant parasite loads developed while pups were dependent on the den site.

Density of kit foxes is poorly documented, although an 18-square mile study area contained 5 adult pairs (0.28 pairs per square mile) (Egoscue 1956). Density of kit foxes on a study area in western Arizona was 0.56 to 0.71 per square mile, compared to densities of 0.95 to 5 per square mile in California (Grinnell et al. 1937). Mean density of adult kit foxes reportedly is about 0.14 per square mile (White and Garrott 1999).

Dispersal by kit foxes is relatively poorly documented or understood (White and Garrott 1999). One study documented dispersal by a young male, a young female, and 1 animal of unknown age (Daneke et al. 1984). Dispersal was greatest for the individual of unknown age that was recaptured about 75 miles from the original capture site. This kit fox would have had to cross a mountain range of about 11,150 feet maximum elevation. Given the vegetation community at this elevation, and the likelihood that a kit fox would not use this vegetation type, the animal likely moved around the mountain, dispersing about 105 miles. In comparison, the young female moved about 1 mile from the original capture site, remained there about 6 weeks, then returned to an area closer to her natal home range. The young male moved about 0.25 miles from the capture site and established a home range that included its natal home range.



Kit foxes are curious, have little fear of humans, and occasionally might enter human residences.

Survival

Kit foxes can live to about 8 to 9 years of age, but foxes older than 5 years typically represent less than 5% of a population. Median age in a population usually is between 1.8 and 2.2 years. Juveniles typically have lower rates of survival than adults. Overall sex ratios likely are about 1:1 for adults (2 or more years old), yearlings (1 year old), and juveniles (less than 1 year old). Populations of kit foxes usually consist of about 20%–45% adults, 5%–20% yearlings, and 40%–70% pups (White and Garrott 1999).

Many kit foxes in Utah were killed by vehicles (Egoscue 1956). Great-

Kit Fox

horned owls (*Bubo virginianus*) might be another source of mortality. Predation by coyotes might be a variable but potentially important source of mortality in populations of kit foxes, but effects of coyote-related mortalities on dynamics of populations remains uncertain (White and Garrott 1997). Captive coyotes promptly killed and ate a kit fox that was introduced into the same holding pen (Egoscue 1956). Although use of traps and poisons to control predators is currently limited in Arizona, kit foxes seem particularly susceptible to mortality from these sources (Grinnell et al. 1937, Egoscue 1956).

Management and Conservation

Population Trends

Kit foxes fill a unique niche in the American West, inhabiting arid regions that include desert scrublands with sparse vegetation and desert grasslands. These habitats have undergone extensive modification over the past century, and abundance of kit foxes has declined substantially in many areas (Zoellick et al. 1989). Several extant subspecies of kit foxes currently are listed as endangered in the United States (Olterman and Verts 1972, U.S. Fish and Wildlife Service 1987). The only subspecies occurring in Mexico also has been classified as endangered, largely caused by agricultural conversions of native habitats (Maldonado et al. 1997). Conversions of historic habitats of kit foxes to agricultural lands caused extinction of 1 subspecies, the long-eared kit fox (*V. m. macrotis*) (Grinnell et al. 1937), and widespread declines in populations of other subspecies of kit foxes (Laughrin 1970, Knapp 1978).

Unpredictable fluctuations in precipitation in desert systems contribute to high-amplitude, high-frequency fluctuations of populations of kit foxes and their prey, particularly rabbits and rodents. Levels of precipitation contribute to density-independent variations in reproductive rates of kit foxes, which are influenced by food supply. However, both density dependence and levels of rainfall might influence abundance of kit foxes (Dennis and Otten 2000). Thus, population sizes of kit foxes typically exhibit marked instability, and densities often vary 5-fold or more from year to year (White and Garrott 1999). Abundance of prey and behavioral spacing mechanisms might be major factors regulating densities of kit foxes (White and Garrott 1997).

No empirical data are available on the status of kit foxes in Arizona, but desert grasslands in this state have been altered by forage overutilization by large herbivores and absence of historic fire regimes. These changes undoubtedly have reduced quality and quantity of habitats suitable for kit foxes. Conversions of native habitats to both agricultural and urban lands (Knapp 1978, Maldonado et al. 1997) likely has adversely affected abundance and distribution of kit foxes in Arizona. Kit foxes need large areas of intact habitat to meet dietary and social requirements (Zoellick et al. 1989). Populations of kit foxes in Arizona might

be particularly sensitive to and affected by large-scale land conversions that have occurred and likely will continue to occur. Recent preliminary research in Arizona suggested abundance of kit foxes in semi-desert and Great Basin grasslands in southeastern and northeastern regions, respectively, were much lower than in northwestern and southwestern areas of the state (T. McKinney, Arizona Game and Fish Department, unpublished data).

Harvest Strategies

The Arizona Game and Fish Department currently classifies kit foxes as predatory mammals, and harvest is regulated via Commission Order 13, which includes all predatory mammals that are not classified as big game species. Currently, harvest season for kit foxes opens statewide August 1 and closes March 31, except for some wildlife refuges and metropolitan areas. Legal harvest on Kofa and Imperial national wildlife refuges opens in October and closes in February. A limited weapons season for kit foxes in urban interfaces in Phoenix and Flagstaff has the same opening and closing dates as the statewide season. Legal methods of harvest of kit foxes include trapping, as prescribed in Arizona Game and Fish Commission rule R-12-4-307, and shooting, as prescribed in Commission rules R-12-4-301 and R-12-4-304. There is no limit on the number of kit foxes that can be harvested using either method.

Management of kit foxes in Arizona is guided by Arizona Game and Fish Department's Wildlife 2006 (Arizona Game and Fish Department 2001). The goal for all predatory mammals is to maintain historic range and distribution of species and to allow for maximum recreational, economic, and aesthetic uses commensurate with existing populations. No specific objective is outlined for kit foxes, but the desired annual harvest level for all 3 fox species found in Arizona (common gray fox, kit fox, red fox [*Vulpes vulpes*]) is 3,000 to 5,000 animals. Annual harvest by hunters and trappers in Arizona between 1990 and 2002 averaged about 4,800 foxes. The Arizona Game and Fish Commission adopted a predator management policy in 2000, but there is no mention of kit foxes in this policy.

Population Ecology

Kit foxes are a small-sized predator. Some research suggests predation by coyotes can be an additive mortality factor (increases total mortalities in a population), but abundance of prey likely regulates populations of kit foxes (Ralls and White 1995, White and Garrott 1997). Moreover, levels and patterns of rainfall, as well as prey abundance, might influence kit fox populations (White et al. 1996, Dennis and Otten 2000, Ernest et al. 2000). Wide fluctuations in abundance of kit foxes might be intrinsic in desert systems they inhabit and might not reflect persistent or special causes such as disease or predation (White and Garrott 1999).

Abundance of kit foxes has declined throughout their range of distribu-

tion. Human-caused factors such as trapping and poisoning have been implicated as part of the reason for declines. Loss of habitat caused by agricultural and urban expansion has adversely affected kit foxes and will continue to do so in the future. Selection of den sites by kit foxes is quite specific; sites are selected based on soil type, predator avoidance strategies, and vegetation structure. Lack of fire and overutilization of forage resources by exotic grazers in the Southwest have led to degradation of desert grasslands through increased shrub invasion. As a result, less suitable habitat is available for kit foxes, including fewer suitable den sites.

Research Needs

Throughout their range in the southwestern U. S. and northwestern Mexico, kit foxes seldom have been studied, and only a small portion of studies have provided results of broad-based ecological research. As a result, there are many gaps in our knowledge and understanding of this species. The following is an assessment of information, listed in order of priority, needed to develop management prescriptions for kit foxes.

1) Throughout their range, populations of kit foxes are in decline because of various human-related factors, such as direct destruction of habitat from urban development, changes in vegetative communities from more open grasslands to shrublands, and conversion of native lands to agricultural lands. Similar conversions and losses of habitats undoubtedly have occurred in Arizona. Documenting distribution, abundance, and long-term trends of kit fox populations and evaluating factors associated with these variables are important and needed management actions in Arizona.

2) There are no well developed and tested methods to document changes and trends of kit fox populations in Arizona. Suitable approaches to establish and monitor populations likely would involve indirect methods such as surveys of scent-station transects or abundance of scats, as well as estimations of sex ratios and age structures using dental annuli.

3) Little information is available regarding direct sources of mortality of kit foxes. Broad-based ecological studies are needed to document several aspects of biology and ecology of kit foxes in Arizona, including sources and rates of mortality, importance of large blocks of habitat (Zoellick et al. 1989), and interactions between populations of kit foxes and abiotic and biotic variables, including other predators.

4) The Arizona Game and Fish Department combines all fox species for purposes of management. Uncertainties exist regarding kit foxes in Arizona, including ecological requirements, abundance and distribution of populations,

availability and use of habitats, and levels and sources of threats to which they might be exposed. It might be more appropriate to develop approaches to management of foxes in Arizona that distinguish kit foxes from other species of foxes.

5) Current fox species management guidelines in Arizona address research needs for kit foxes, and include:

- a) Develop and implement standardized statewide survey methods to index abundance;
- b) Assess status and impacts of harvest;
- c) Develop annual harvest surveys; and
- e) Evaluate age structure of harvested kit foxes using jaws submitted by hunters and trappers.

Literature Cited

- Arizona Game and Fish Department. 2001. Wildlife 2006. Arizona Game and Fish Department, Phoenix.
- Cockrum, E. L. 1960. The recent mammals of Arizona: their taxonomy and distribution. University of Arizona Press, Tucson.
- Daneke, D., H. Sunquist, and S. Berwick. 1984. Notes on kit fox biology in Utah. *Southwestern Naturalist* 29:361–362.
- Dennis, B., and M. R. M. Otten. 2000. Joint effects of density dependence and rainfall on abundance of San Joaquin kit fox. *Journal of Wildlife Management* 64:388–400.
- Dragoo, J. W., J. R. Choate, T. L. Yates, and T. P. O'Farrell. 1990. Evolutionary and taxonomic relationships among North American arid-land foxes. *Journal of Mammalogy* 71:318–332.
- Egoscue, H. J. 1956. Preliminary studies of the kit fox in Utah. *Journal of Mammalogy* 37:351–357.
- Egoscue, H. J. 1962. Ecology and life history of the kit fox in Tooele County, Utah. *Ecology* 43:481–497.
- Ernest, S. K., M., J. H. Brown, and R. R. Parmenter. 2000. Rodents, plants, and precipitation: spatial and temporal dynamics of consumers and resources. *Oikos* 88:470–48.

- Fisher, J. L. 1981. Kit fox diet in south-central Arizona. M. S. Thesis. University of Arizona, Tucson.
- Golightly, R. T. 1981. Comparative energetics of two desert canids: the coyote (*Canis latrans*) and the kit fox (*Vulpes macrotis*). Ph.D. Thesis. Arizona State University, Tempe.
- Grater, R. K. 1939. The desert kit fox moves his family. *Nature Notes*, J. H. Sedgwick, Peoria, Ill. 6:77–81 (Cited in Hoffmeister 1986).
- Grinnell, J., J. Dixon, and J. M. Linsdale. 1937. Fur-bearing mammals of California. University of California Press, Berkeley.
- Hall, E. R. 1981. The mammals of North America, Second Edition. John Wiley and Sons, New York, NY.
- Hoffmeister, D. F. 1986. The Mammals of Arizona. University of Arizona Press and the Arizona Game and Fish Department, Tucson.
- Knapp, D. K. 1978. Effects of agricultural development in Kern County, California, on the San Joaquin kit fox in 1977. California Department of Fish and Game. Nongame Wildlife Investigation. Final Report. Project E-1-1, Job V-1.21.
- Laughrin, L. 1970. San Joaquin kit fox; its distribution and abundance. California Department of Fish and Game. Wildlife Management Branch Administrative Report 70-2.
- Maldonado, J. E., M. Cotera, E. Geffen, and R. K. Wayne. 1997. Relationships of the endangered Mexican kit fox (*Vulpes macrotis zinseri*) to North American arid-land foxes based on mitochondrial DNA sequence data. *Southwestern Naturalist* 42:460–470.
- McGrew, J. C. 1979. Mammalian species: *Vulpes macrotis*. *American Society of Mammalogy* 123:1–6.
- Morrell, S. H. 1972. Life history of the San Joaquin kit fox. *California Fish and Game* 58:162–174.
- Olterman, J. H., and B. J. Verts. 1972. Endangered plants and animals of Oregon. IV. Mammals. Agriculture Experimental Station Special Report, Oregon State University, Corvallis.

Biological Foundations for Management of Mammalian Predators in Arizona

- Ralls, K., and P. J. White. 1995. Predation of San Joaquin kit foxes by larger canids. *Journal of Mammalogy* 76:723–729.
- U. S. Fish and Wildlife Service. 1987. Endangered and threatened wildlife and plants. Washington, D. C. Special Report, 50CFR 17.11 and 17.12.
- White, P. J., and K. Ralls. 1993. Reproduction and spacing patterns of kit foxes relative to changing prey availability. *Journal of Wildlife Management* 57:861–867.
- White, P. J., C. A. Vanderbilt White, and K. Ralls. 1996. Functional and numeric responses of kit foxes to a short-term decline in mammalian prey. *Journal of Mammalogy* 77:370–376.
- White, P. J., and R. A. Garrott. 1997. Factors regulating kit fox populations. *Canadian Journal of Zoology* 75:1982–1988.
- White, P. J., and R. A. Garrott. 1999. Population dynamics of kit foxes. *Canadian Journal of Zoology* 77:486–493.
- Zoellick, B. W., N. S. Smith, and R. S. Henry. 1989. Habitat use and movements of desert kit foxes in western Arizona. *Journal of Wildlife Management* 53:955–961.
- Zoellick, B. W. 1990. Kit fox activity. Pages 151-155 *in* P. R. Krausman and N. S. Smith, editors. *Managing Wildlife in the Southwest*. Arizona Chapter of the Wildlife Society, Phoenix.
- Zoellick, B. W., and N. S. Smith. 1992. Size and spatial organization of home ranges of kit foxes in Arizona. *Journal of Mammalogy* 73:83–88.



Chapter 7

Gray Fox

Darren G. Tucker, Region III
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Common name - Gray fox

Scientific name - *Urocyon cinereoargenteus*

Life History

Description

Common gray foxes (*Urocyon cinereoargenteus*) are relatively small canids; adults throughout their range in North America generally weigh from about 6.6 to 15.4 pounds, and males are slightly larger than females (Grinnell et al. 1937, Sullivan 1956, Jackson 1961). Total length ranges from about 32.5 to 44.5 inches, length of the tail ranges from about 11 to 17 inches, and length of the hind foot is between about 4 and 6 inches (Hall 1981). Gray foxes in Arizona weighed between about 5.5 and 8.8 pounds (Hoffmeister 1986). Gray foxes have shorter extremities than red foxes (*Vulpes vulpes*), although both have similar weights.



Gray foxes are relatively small mammalian predators found in diverse habitats in Arizona.

Juvenile gray foxes weigh less than adults until about mid-August following birth, when some individuals can be nearly as heavy as minimum weights of adults.

Coloration of adults varies little, and has been described as “pepper and salt” in appearance (Samuel and Nelson 1982). Pelage has grizzled upper parts due to guard hairs banded with black, gray, and white. Black-tipped hairs in the middle of the back form a dark, longitudinal stripe extending to a conspicuous mane of coarse, black hair along top of the black-tipped tail. Portions of limbs,

neck, and sides are cinnamon-colored. Underparts are buff colored, and white shows on ears, throat, belly, chest, and hind legs. Black, rufous, and white facial markings provide distinctive accents (Fritzell 1987). A musk gland on top of the tail extends one third to one half the length of the tail. This is the largest tail gland of North American canids and might function to identify individuals (Seton 1923, Hildebrand 1952, Rue 1968).

Pups are nearly hairless and have dark skin at birth, and their eyes open at 10 to 12 days of age (Linhart 1968). Juveniles have characteristic puppy fur until about mid-August, when they can be distinguished from adults by brightly-colored tails. In contrast, tails of adults at this time are faded and badly worn due to shedding prior to new growth of hair in autumn. A single annual molt extends from summer to autumn (Grinnell et al. 1937), and fur is prime between late November and mid-February, with peak primeness during December (Stains 1979). Gray foxes have 6 mammae; compared to adult females, juvenile females generally have smaller mammary nipples (Lord 1961).

Skulls of gray foxes can be distinguished from those of other canids in North America by the presence of conspicuous, widely separated temporal ridges that approach each other posteriorly to produce a lyrate or U-shaped form on top of the skull. The supraorbital crest is slightly curved downward and has a distinct concave depression on the upper surface, and temporal bones are roughened below temporal ridges (Fritzell 1987). The rostrum is relatively heavy and short. A distinct “step” occurs on the ventral border of the dentary, midway between the interior border and the coronoid process and tip of the angular process. This step results in 3 “notches” at rear of the jaw, in contrast to jaws of kit foxes (*Vulpes velox macrotis*) and red foxes (Hall and Kelson 1959).

Well-defined tracks of gray foxes are similar in appearance and size to tracks of kit foxes, slightly smaller than those of red foxes, show 4 toes with claw marks, triangular shaped heel pads, and are about 1.5 to 1.6 inches long and slightly narrower in width (Murie 1974, Fritzell 1987). Tracks of coyotes (*Canis latrans*) also are similar in appearance to those of gray foxes, but generally are larger and similar in size to tracks of red foxes.



The gray fox (above) has a distinct U-shaped depression on the top of its skull, and a shorter nose than the red fox (below).

Distribution

Gray foxes occur throughout much of the United States, except for portions of the Northwest, northern Rocky Mountains, and Great Plains. They inhabit brushy, wooded, and rocky regions from extreme southern Canada to northern Columbia and Venezuela (Fritzell 1987). Gray foxes occur throughout Arizona (Figure 1), are particularly abundant at elevations within or below oak (*Quercus* spp.) and pinyon-juniper (*Pinus* spp.–*Juniperus* spp.) woodlands, and occupy habitats between about 10,000 feet and lowest desert regions (Hoffmeister 1986).

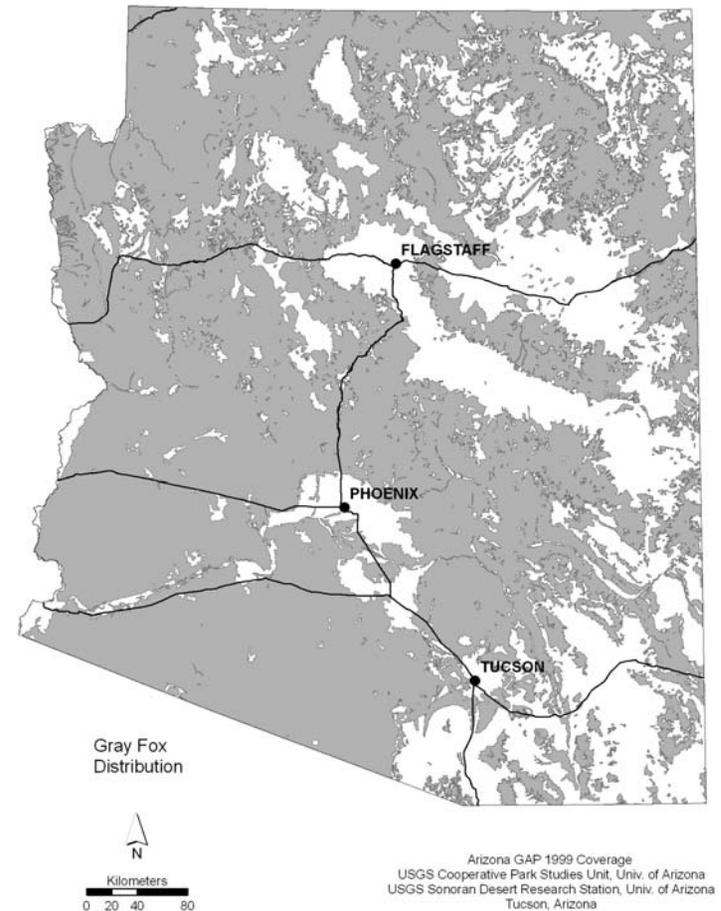


Figure 1. Distribution of gray foxes in Arizona.

The genus *Urocyon* has been present in North America for about 5 million years, and oldest fossils found are from the Pliocene epoch. Oldest fossils of *U. cinereoargenteus* are associated with Pleistocene deposits about 1 million years old (Fritzell 1987). Fossils of gray foxes have been found in eastern North America (Guilday and Bender 1958, Waters 1964). Gray foxes historically inhabited much of eastern North America (Churcher 1959), but disappeared from New England until the 19th century (Grayce 1957). *Urocyon* moved north and east during a warm Hypsithermal period (the warmest postglacial period from 5000 to 2000 B.C. when temperatures rose to as much as 36 °F above present temperatures, precipitating a dramatic rise in sea level), and its range contracted southward as weather cooled. Other shifts in distribution might have occurred during a second warm period (1000 to 1300 A.D.). Re-distribution into New England since 1850 might be related to a climatic warming trend (Dorf 1959).

Habitat

Vegetation types occupied by gray foxes can include ponderosa pine (*P. ponderosa*) and Douglas fir (*Pseudotsuga* spp.), but are primarily open desert scrub, chaparral, and pinyon-juniper (Hoffmeister 1986). In general, dense, protective cover is an essential component of habitats of gray foxes (Fuller 1978, Haroldson and Fritzell 1984, Hallberg and Trapp 1984, Fritzell 1987, Yearsley and Samuel 1980). Brushlands, riparian corridors, rocky, broken terrain, and diversity of cover and vegetation can be important components of habitats



Gray fox habitat in chaparral vegetation near Pinal Mountains in southeastern Arizona. Fire previously reduced vegetation density in much of the foreground area of the photo.

(Wood 1952, 1958; Fuller 1978; Trapp 1978; Hallberg and Trapp 1984; Fritzell 1987; Fedriani et al. 2000). Wildfire in central Arizona reduced vegetation more than 90% and likely contributed to decline in abundance of gray foxes (Cunningham et al. 2006). Dens of gray foxes can be located in crevices in cliffs, holes in the ground, piles of brush, rocks, or wood, hollows of trees, and mine shafts (Trapp and Hallberg 1975, Hoffmeister 1986).

Diets and Prey

Gray foxes eat a wide variety of animal and plant matter, and might be more omnivorous (eat both plants and animals) than other canids in North America. Food resources eaten by gray foxes vary by region and season, and in Arizona include small rodents, rabbits, insects, reptiles, birds, carrion, and fruits of cactus, hackberry (*Celtis* spp.), manzanita (*Arctostaphylos* spp.), and juniper; scats (feces) often consist entirely of berries or seeds (Hoffmeister 1986, Fritzell 1987, Novaro et al. 1995, Cunningham et al. 2006). Fruits comprised nearly 91% of diets of gray foxes in Zion National Park in Utah between September and February, and average occurrence of fruits in diets declined to about 43% during other months (Trapp and Hallberg 1975). Insects and grasses in diets of gray foxes in Arizona increased during March through August (Hoffmeister 1986). In Belize, diets of gray foxes consisted primarily of fruits during the transition from late dry season to early wet season (Novaro et al. 1995).

During autumn and winter in Maryland, plants, mammals, birds, and insects, in descending order, were major foods in diets of gray foxes (Hockman and Chapman 1983). In another study in Maryland, animals exceeded plant material in diets (Llewellyn and Uhler 1952). Similarly, diets of gray foxes in Texas consisted mainly of small mammals, followed by birds, plants, and insects (Wood 1954). Plants were eaten more by gray foxes than by sympatric red foxes, and gray foxes scavenge carrion when it is available (Llewellyn and Uhler 1952, Scott 1955, Hoffmeister 1986).

In the Sonoran Desert of the Southwest, gray foxes were opportunistic with respect to foods eaten, and mammals comprised (frequency of occurrence) 57% to 65% of diets, whereas insects comprised 53% to 57%, and plants comprised 39% to 40% (Turkowski 1969). Gray foxes in Mazatzal Mountains in central Arizona primarily ate berries and fruits, but scats also contained rabbits (6.5%–23.5%), rodents (17.1%–62.8%), birds (4.3%–17.5%), insects (7.8%–25.4%), and reptiles (1.1%–5.9%) (Cunningham et al. 2001, 2006). Relative abundance of gray foxes likely is linked to availability of food resources, which in turn is influenced by habitat type (Wood et al. 1958, Novaro et al. 1995). Reduced abundance of gray foxes following a wildfire in central Arizona likely corresponded with reductions in availability of cover and food (Cunningham et al. 2001, 2006).

Behavior

Although gray foxes are primarily nocturnal, they are often crepuscular and active in early morning or evenings (Trapp 1978, Haroldson and Fritzell 1984, Hoffmeister 1986, Fedriani et al. 2000). They reportedly are active from 25% to 54% of the time during daylight, and from 77% to 87% of the time at night (Yearsley and Samuel 1980, Haroldson and Fritzell 1984). They also move more rapidly at night than during the day (Follmann 1973, Yearsley and Samuel

1980, Hallberg and Trapp 1984, Haroldson and Fritzell 1984). Gray foxes might tend to avoid habitats with higher densities of coyotes, and thus avoid or reduce risk of predation by the larger canid (Fedriani et al. 2000, Chamberlain and Leopold 2005).

Feces and urine apparently play a role in communications between gray foxes, and excretions often are deposited in conspicuous locations on bare ground, logs, rocks, elevated sites, or along trails (Grinnell et al. 1937, Richards and Hine 1953, Trapp 1978, Fritzell 1987). Scats usually are deposited singly, but might be deposited in groups of 2 to 8 ball-shaped masses (boluses) (Trapp 1978). Barking is the most common vocalization, and gray foxes might vocalize more in February during the breeding season and in late summer when pups are learning to hunt (Rue 1968).

Gray foxes are well adapted to climbing and can climb branchless, vertical tree trunks to more than 50 feet by grasping the trunk with forefeet and pushing with hind feet. They can climb by jumping from branch to branch (Seton 1929, Grinnell et al. 1937, Terres 1939, Taylor 1943, Leopold 1959). They also might become nuisances to humans in some situations by preying on poultry or pets (Hockman and Chapman 1983).



Gray fox in ponderosa pine forest habitat in northern Arizona. Foxes are best known for capturing their prey by pouncing, jumping up to three feet above the ground and diving with their front paws onto the prey. The pouncing action either stuns or flushes the prey.

Reproduction

Gray foxes are assumed to be monogamous, but conclusive evidence is lacking (Fritzell 1987). Gray foxes breed between January and April, but tend to breed earlier in southern than in northern areas of their range (Samuel and Nelson 1982), and females are monestrous (they have one estrous cycle per year) (Rowlands and Parkes 1935). Breeding season in California is in February; in Texas it occurs during February and March (Hoffmeister 1986). Estrus (sexual receptivity) lasts 1 to 6 days (Asdell 1964), and females might breed at about 10

months of age, but all females likely do not breed their first year of age. Some yearlings might reach puberty too late for successful breeding (Layne 1958), but timing of breeding might not differ between yearling and older females (Wood 1958). About 8% of females may not produce young their first year (Wood 1958); a female that does not produce offspring may have unfertilized eggs, experience mortality of embryos, or may not have a mate (Layne and McKeon 1956).

Spermatozoa form in testes of males during October and November, and most males are capable of breeding their first year, although the penis is not completely ossified (the process of bone formation, in which connective tissues such as cartilage are turned to bone or bone-like tissue) (Lloyd and Englund 1973, Storm et al. 1976). Throughout the reproductive cycle, testes of adults and yearlings had equivalent numbers of spermatogonia (the cells that sperm originate from), primary spermatocytes, spermatids, and spermatozoa (Follman 1978). Weights of testes are maximal from December to January and decrease until quiescence (inactivity) between late June and September (Layne 1958).

Gestation period is thought to be about 53 days (Sheldon 1949) but in captivity might be 59 days (Fritzell 1987). Mean litter size is about 4 pups



(Fritzell 1987), likely does not vary among gray foxes less than 4.5 years old (Root 1981), and sex ratio of pups is equal (Samuel and Nelson 1982, Fritzell 1987). Females and males care for pups, the pair remains intact through denning, and adults might attempt to decoy observers away from young (Turkowski 1969). Adult males might forage separately from females at night, and return to dens at daylight, but only females likely make repeated trips to dens at night (Nicholson et al. 1985). Dens are vacated when pups are 2.5 to 3 months old, but pups remain with adult females until about 4 months of age, when they begin foraging on their own and are independent of their mothers (Wood 1958, Nicholson et al. 1985). Failure to excavate all pups from a den and communal denning might bias estimations of litter sizes and sex ratios (Hoffman and Kirkpatrick 1954). Intrauterine mortality might bias estimations of size of

litters based on placental scars (Englund 1970, Allen 1975). Size of litters, sex ratios, gestation period and development of young are poorly understood for gray foxes in Arizona (Hoffmeister 1986).

Home Range, Density, and Dispersal

Sizes of home ranges of gray foxes vary greatly between habitat types and seasons. Depending on sexual maturity, sex, and period of measurement, sizes of home ranges among studies have ranged from 0.1 to 2.6 square miles (Lord 1961, Follman 1973, Trapp and Hallberg 1975, Haroldson and Fritzell 1984, Sawyer and Fendley 1990). Size of home range likely is larger for males than females in each season, and increases for males progressively from denning season through pup rearing, dispersal, and mating. Home range of females likely is smallest during pup rearing and largest during dispersal and mating seasons



Bobcats may compete with and occasionally prey on gray foxes in some areas of the United States, but bobcats and gray foxes seem to share food and space resources in Arizona's Sonoran Desert.

(Sawyer and Fendley 1990). Abundance of food resources, competition (intra-specific and interspecific), type and diversity of habitat, and physical barriers to movements seem to influence sizes of home ranges (Samuel and Nelson 1982). However, differences in sizes of home ranges might be related to differences in quality of habitat, rather than age, breeding season, and weather (Fuller 1978), although birth of pups likely restricts movements by females (Follman 1973).

Few studies have estimated densities of gray foxes, but published estimates range from 0.5 to 0.8/square mile, depending on location, method of estimation, and season (Errington 1933, Gier 1948, Lord 1961, Trapp 1978).

Gray fox

Scent-station surveys might be the most widely used method for indexing relative abundance of populations of mammalian carnivores (Sargeant et al. 2003), and the method has been used to estimate relative abundance of gray foxes (Wood 1959, Hatcher and Shaw 1981, Hon 1981, Conner et al. 1983).

Survival

Longevity of gray foxes in the wild generally might be about 4 years (Wood 1958, Lord 1961, Storm et al. 1976); despite relatively high annual mortality, gray foxes occasionally might live 14 to 15 years (Seton 1929). However, few gray foxes likely live longer than 4 to 5 years. Nearly half of newborns might die within their first 7 months, 90% of juveniles might die within their first year, and half to two-thirds of adults might die each year. Human harvest by hunting and trapping undoubtedly represent the single most important source of mortality of gray foxes (Fritzell 1987). Predation of gray foxes by other predators likely do not have population-level effects, but golden eagles (*Aquila chryseatos*), coyotes, bobcats (*Lynx rufus*), and mountain lions (*Puma concolor*) occasionally prey on gray foxes (Grinnell et al. 1937, Gander 1966, Mollhagen et al. 1972, McKinney et al. 2006).



Arizona Game and Fish Department biologist Thorry Smith prepares a scent-station tracking bed in the Sonoran Desert. Scent stations are widely used in monitoring abundance of carnivores and can provide indices of relative abundance and population changes of gray foxes.

Gray foxes may be exposed to numerous infectious diseases that might be sources of mortality, including brucellosis, distemper, infectious canine hepatitis, leptospirosis, rabies, tularemia, and parvovirus. Gray foxes also host numerous parasites, such as fleas, lice, ticks, chiggers, mites, cestodes, nematodes, trematodes, and acanthocephalans (Nicholson and Hill 1984, Fritzell 1987, Samuel et al. 2001, Williams and Barker 2001). Gray foxes are highly resistant to infestation by sarcoptic mange mites (*Sarcoptes scabiei*) and infections with heartworms (*Dirofilaria immitis*) which might affect other canids (Stone et al. 1972, Monson

et al. 1973, Simmons et al. 1980). Gray foxes might spread enzootic rabies to domestic animals, and rabies in some cases might affect populations of the predator (Parker et al. 1957, Jennings et al. 1960, Trapp and Hallberg 1975). Gray foxes in wild populations might regularly associate with human developments, potentially increasing their exposure to diseases such as canine parvovirus (Riley et al. 2004).

Management and Conservation

Population Ecology and Trends

Few management programs have focused specifically on populations of gray foxes (Trippensee 1953, Pils and Martin 1978), and little is known regarding ecology and trends of populations in Arizona. Inadequate levels of precipitation in deserts reduce production of small mammals (Beatley 1969, Reichman and Van De Graaff 1975, Whitford 1976), potentially influencing diets and relative abundance of gray foxes.

Harvest Strategies

Management of gray foxes in Arizona is guided by Arizona Game and Fish Department's Wildlife 2006 (Arizona Game and Fish Department 2001). The goal for all predatory mammals is to maintain historic range and distribution of species and to allow for maximum recreational, economic, and aesthetic uses commensurate with existing populations. No specific objective is outlined for gray foxes, but the desired annual harvest level for all 3 species of foxes found in Arizona (gray fox, kit fox, red fox) is 3,000 to 5,000 animals. Annual harvest by hunters and trappers in Arizona between 1990 and 2002 averaged about 4,800 foxes.

Research Needs

Gray foxes seldom have been studied in Arizona, and very little information is available regarding abundance, trends, effects of harvest levels, and status of the species in arid or semi-arid environments. No studies have provided broad-based ecological research of gray foxes in Arizona, and there are many gaps in knowledge and understanding of this species. The following actions and information are needed in Arizona, and ostensibly would enhance development of management prescriptions for gray foxes.

- 1) Separate annual harvest levels for all 3 species of foxes.
- 2) Monitor habitat-specific harvest and trends of relative abundance.
- 3) Determine age and sex structure of annual harvest, at least at 3-year

intervals.

- 4) Monitor nuisance activities and human interactions, including depredations and disease-related issues.
- 5) Evaluate habitat-specific association between abundance of gray foxes, bobcats, and coyotes.
- 6) Determine effects of environmental variables such as drought and habitat fragmentation on relationships between gray foxes, their prey, and other food resources.
- 7) In general, increase knowledge about life history, particularly with respect to harvest levels, habitats, dispersal and environmental variables such as drought and habitat fragmentation.
- 8) Current fox species management guidelines in Arizona address research needs for gray foxes, and include:
 - a) Develop and implement standardized statewide survey methods to index abundance;
 - b) Assess status and impacts of harvest;
 - c) Develop annual harvest surveys; and
 - d) Evaluate age structure of harvested gray foxes using jaws submitted by hunters and trappers.

Literature Cited

- Allen, S. H. 1975. The influence of age and other factors on red fox reproduction. Paper Presented at 37th Midwest Wildlife Conference, Toronto, Canada.
- Arizona Game and Fish Department. 2001. Wildlife 2006. Arizona Game and Fish Department. Phoenix.
- Asdell, S. A. 1964. Patterns of mammalian reproduction. Second edition. Cornell University Press, Ithaca, N.Y.
- Beatley, J. C. 1969. Dependence of desert rodents on winter annuals and precipitation. *Ecology* 50:721-724.
- Chamberlain, M. J., and B. D. Leopold. 2005. Overlap in space use among

Biological Foundations for Management of Mammalian Predators in Arizona

- bobcats (*Lynx rufus*), coyotes (*Canis latrans*), and gray foxes (*Urocyon cinereoargenteus*). *American Midland Naturalist* 153:171–179.
- Churcher, C. S. 1959. The specific status of the new world red fox. *Journal of Mammalogy* 40:513–520.
- Conner, M. C., R. F. Labisky, and D. R. Progulsk, Jr. 1983. Scent-station indices as measures of population abundance for bobcats, raccoons, gray foxes, and opossums. *Wildlife Society Bulletin* 11:146–152.
- Cunningham, S. C., L. M. Monroe, L. Kirkendall, and C. L. Ticer. 2001. Effects of the catastrophic Lone Fire on low, medium, and high mobility wildlife species. Technical Guidance Bulletin No. 5, Arizona Game and Fish Department, Phoenix.
- Cunningham, S. C., L-B. Kirkendall, and W. B. Ballard. 2006. Gray fox and coyote abundance and diet responses after a wildfire in central Arizona. *Western North American Naturalist* 66:169–180.
- Dorf, E. 1959. Climatic changes of the past and present. *Contribution of the University of Michigan Museum of Paleontology* 13:199.
- Englund, J. 1970. Some aspects of reproduction and mortality rates in Swedish foxes (*Vulpes vulpes*), 1961–63 and 1966–69. *Viltrevy* 8:1–82.
- Errington, P. L. 1933. Bobwhite winter survival in an area heavily populated with gray foxes. *Iowa State College Journal of Science* 8:127–130.
- Fedriani, J. M., T. K. Fuller, R. M. Sauvajot, and E. C. York. 2000. Competition and intraguild predation among three sympatric carnivores. *Oecologia* 125:258–270.
- Follman, E. H. 1973. Comparative ecology and behavior of red and gray foxes. Ph. D. Dissertation, Southern Illinois University, Carbondale.
- Follman, E. H. 1978. Annual reproductive cycle of the male gray fox. *Transactions of the Illinois State Academy of Science* 71:304–311.
- Fritzell, E. K. 1987. Gray fox and island gray fox. Pages 408–420 in M. J. Novak, J. A. Baker, M. E. Obbard, and B. Malloch, editors. *Wild furbearer management and conservation in North America*. Ontario Ministry of Natural Resources, Toronto, Ontario, Canada.

Gray fox

- Fuller, E. K. 1978. Variable home-range sizes of female gray foxes. *Journal of Mammalogy* 59:446–449.
- Gander, E. F. 1966. Friendly foxes. *Pacific Discovery* 19:28–31.
- Gier, H. T. 1948. Rabies in the wild. *Journal of Wildlife Management* 12:142–153.
- Grayce, R. L. 1957. Checklist of New England mammals. *Bulletin of the Massachusetts Audubon Society* 41:15–24.
- Grinnell, J., J. S. Dixon, and J. M. Linsdale. 1937. Fur-bearing mammals of California. Two volumes. University of California Press, Berkeley.
- Guilday, J. E., and M. S. Bender. 1958. A recent fissure deposit in Bedford County, Pennsylvania. *Annals of the Carnegie Museum* 35:127–138.
- Hall, E. R., and K. R. Kelson. 1959. *The mammals on North America*. Volume 2. Ronald Press, New York, New York.
- Hall, E. R. 1981. *The mammals of North America*. Second edition. Two volumes. John Wiley and Sons, New York, New York.
- Hallberg, D. L., and G. R. Trapp. 1984. Gray fox (*Urocyon cinereoargenteus*) temporal and special activity in a riparian-agricultural habitat in California's central valley. Pages 920–928 in R. E. Warner and K. Hendrix, editors. *California Riparian Systems Conference*, University of California Press, Berkeley.
- Haroldson, K. J., and E. K. Fritzell. 1984. Home ranges, activity, and habitat use by gray foxes in an oak-hickory forest. *Journal of Wildlife Management* 48:222–227.
- Hatcher, R. T., and J. H. Shaw. 1981. A comparison of three indices to furbearer populations. *Wildlife Society Bulletin* 9:153–156.
- Hildebrand, M. 1952. The integument in Canidae. *Journal of Mammalogy* 33:419–428.
- Hockman, J. G., and J. A. Chapman. 1983. Comparative feeding habits of red foxes (*Vulpes vulpes*) and gray foxes (*Urocyon cinereoargenteus*) in Maryland. *American Midland Naturalist* 102:276–285.

Biological Foundations for Management of Mammalian Predators in Arizona

- Hoffman, R. A., and C. M. Kirkpatrick. 1954. Red fox weights and reproduction in Tippecanoe County, Indiana. *Journal of Mammalogy* 55:504–509.
- Hoffmeister, D. F. 1986. *Mammals of Arizona*. University of Arizona Press, Tucson, and Arizona Game and Fish Department, Phoenix.
- Hon, T. 1981. Relative abundance of bobcats in Georgia: survey techniques and preliminary results. *National Wildlife Federation Scientific and Technical Series* 6:104–106.
- Jackson, H. H. T. 1961. *Mammals of Wisconsin*. University of Wisconsin Press, Madison.
- Jennings, W. L., N. J. Schneider, A. L. Lewis, and J. E. Scatterday. 1960. Fox rabies in Florida. *Journal of Wildlife Management* 24:171–179.
- Layne, J. N. 1958. Reproductive characteristics of the gray fox in Southern Illinois. *Journal of Wildlife Management* 22:157–163.
- Layne, J. N., and W. H. McKeon. 1956. Some aspects of red fox and gray fox reproduction in New York. *New York Fish Game Journal* 3:44–74.
- Leopold, A. S. 1959. *Wildlife of Mexico*. University of California Press, Berkeley.
- Linhart, S. B. 1968. Dentition and pelage in the juvenile red fox (*Vulpes vulpes*). *Journal of Mammalogy* 49:526–528.
- Llewellyn, L. M., and F. M. Uhler. 1952. The foods of fur animals of the Patuxent Research Refuge, Maryland. *American Midland Naturalist* 48:193–204.
- Lloyd, H. G., and J. Englund. 1973. The reproductive cycle of the red fox in Europe. *Journal of Reproduction and Fertility Supplement* 19:119–130.
- Lord, R. D. 1961. A population study of the gray fox. *American Midland Naturalist* 66:87–109.
- McKinney, T., T. W. Smith, and J. C. deVos, Jr. 2006. Evaluation of factors potentially influencing a desert bighorn sheep population. *Wildlife Monographs* 164:1–36.
- Mollhagen, T. R., R. W. Wiley, and R. L. Packard. 1972. Prey remains in golden eagle nests: Texas and New Mexico. *Journal of Wildlife Management*

Gray fox

36:784–792.

- Monson, R. A., W. B. Stone, and B. L. Weber. 1973. Heartworms in foxes and wild canids in New York. *New York Fish and Game Journal* 20:48–53.
- Murie, O. J. 1974. *Animal tracks*. Second edition. Houghton Mifflin Company, New York, New York.
- Nicholson, W. S., and E. P. Hill. 1984. Mortality in gray foxes from east-central Alabama. *Journal of Wildlife Management* 48:1429–1432.
- Nicholson, W. S., E. P. Hill, and D. Briggs. 1985. Denning, pup-rearing, and dispersal in the gray fox in east-central Alabama. *Journal of Wildlife Management* 49:33–37.
- Novaro, A. J., R. S. Walker, and M. Suarez. 1995. Dry-season food habits of the gray fox (*Urocyon cinereoargenteus fraterculus*) in the Belizean Peten. *Mammalia* 59:19–24.
- Parker, R. L., J. W. Kelly, E. L. Cheatum, and D. J. Dean. 1957. Fox population densities in relation to rabies. *New York Fish Game Journal* 4:219–228.
- Pils, C. M., and M. A. Martin. 1978. Population dynamics, predator-prey relationships and management of the red fox in Wisconsin. Report No. 105, Wisconsin Department of Natural Resources, Madison.
- Reichman, O. J., and K. M. Van De Graaff. 1975. Associations between ingestion of green vegetation and desert rodent reproduction. *Journal of Mammalogy* 56:503–506.
- Richards, W. W., and R. L. Hine. 1953. Wisconsin fox populations. Wisconsin Conservation Department Technical Wildlife Bulletin 6 :1–78.
- Riley, S. P. D., J. Foley, and B. Chomel. 2004. Exposure to feline and canine pathogens in bobcats and gray foxes in urban and rural zones of a National Park in California. *Journal of Wildlife Diseases* 40:11–22.
- Root, D. A. 1981. Productivity and mortality of gray foxes and raccoons in southwestern Wisconsin. M.S. Thesis, University of Wisconsin, Stevens Point.
- Rowlands, I. W., and A. S. Parkes. 1935. The reproductive proceeds of certain

Biological Foundations for Management of Mammalian Predators in Arizona

- mammals—reproduction in foxes (*Vulpes* spp.). Proceedings of the Zoological Society of London. Part 8:823–841.
- Rue, L. L. 1968. Sportsman's guide to game animals. Harper and Row, New York, New York.
- Samuel, D. E., and B. B. Nelson. 1982. Foxes: *Vulpes vulpes* and allies. Pages 475–490 in J. A. Chapman and G. A. Feldhammer, editors. Wild mammals of North America. Johns Hopkins University Press, Baltimore, Maryland.
- Samuel, W. M., M. J. Pybus, and A. A. Kocan. 2001. Parasitic diseases of wild mammals. Second edition. Iowa State University Press, Ames.
- Sargeant, G. A., D. H. Johnson, and W. E. Berg. 2003. Sampling designs for carnivore scent-station surveys. Journal of Wildlife Management 67:289–298.
- Sawyer, D. T., and T. T. Fendley. 1990. Seasonal home range size and movement behavior of the gray fox on the Savannah River site, South Carolina. Proceedings of the Annual Conference SEAFW. 1990:380–389.
- Scott, T. G. 1955. Dietary patterns of red and gray foxes. Ecology 36:366–367.
- Seton, E. T. 1923. The mane and tail of the gray fox. Journal of Mammalogy 4:180–182.
- Seton, E. T. 1929. Lives of game animals. Volume 1. Doubleday, Duran and Company. Garden City, New Jersey.
- Sheldon, W. G. 1949. Reproductive behavior of foxes in New York state. Journal of Mammalogy 30:236–246.
- Simmons, J. M., W. S. Nicholson, E. P. Hill, and D. B. Briggs. 1980. Occurrence of *Dirofilaria immitis* in gray fox (*Urocyon cinereoargenteus*) in Alabama and Georgia. Journal of Wildlife Diseases 16:225–228.
- Stains, H. J. 1979. Primeness in North American furbearers. Wildlife Society Bulletin 7:120–124.
- Stone, W. B., Jr., E. Parks, B. L. Weber, and F. J. Parks. 1972. Experimental transfer of sarcoptic mange from red foxes and wild canids to captive wildlife and domestic animals. New York Fish and Game Journal 19:1–11.

Gray fox

- Storm, G. L., R. D. Andrews, R. L. Phillips, R. A. Bishop, D. B. Siniff, and J. R. Testes. 1976. Morphology, reproduction, dispersal and mortality of Midwestern red fox populations. Wildlife Monographs 49:1–82.
- Sullivan, E. G. 1956. Gray fox reproduction, denning, range and weights in Alabama. Journal of Mammalogy 37:346–351.
- Taylor, W. P. 1943. The gray fox in captivity. Texas Game and Fish 1:12–13, 19.
- Terres, J. K. 1939. Tree climbing technique of a gray fox. Journal of Mammalogy 20:256.
- Trapp, G. R., and D. L. Hallberg. 1975. Ecology of the gray fox (*Urocyon cinereoargenteus*). Pages 164–178, in M. F. Fox, editor. The wild canids. Van Nostrand Reinhold, New York, New York.
- Trapp, G. R. 1978. Comparative behavioral ecology of the ringtail (*Bassariscus astutus*) and gray fox (*Urocyon cinereoargenteus*) in southwestern Utah. Carnivore 1:3–32.
- Trippensee, R. E. 1953. Wildlife management, furbearers, waterfowl and fish. Volume 2. McGraw-Hill, New York, New York.
- Turkowski, F. J. 1969. Food habits and behavior of the gray fox (*Urocyon cinereoargenteus*) in the lower and upper Sonoran life zones of southwestern United States. Ph. D. Dissertation, Arizona State University, Tempe.
- Waters, J. H. 1964. Red and gray fox from New England archeological sites. Journal of Mammalogy 45:307–308.
- Whitford, W. G. 1976. Temporal fluctuations in density and diversity of desert rodent populations. Journal of Mammalogy 57:351–359.
- Williams, E. S., and I. K. Barker. 2001. Infectious diseases of wild mammals. Third edition. Iowa State University Press, Ames.
- Wood, J. E. 1952. The effects of agriculture (ranching and farming) on the habitat and food supply of furbearers in the post oak region of Texas. Transactions of the North American Wildlife Conference 17:427–437.
- Wood, J. E. 1954. Food habits of furbearers of the upland post oak region in Texas. Journal of Mammalogy 35:406–415.

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- Wood, J. E. 1958. Age structure and productivity of a gray fox population. *Journal of Mammalogy* 39:74–86.
- Wood, J. E., D. E. Davis, and E. V. Komarek. 1958. The distribution of fox populations in relation to vegetation in southern Georgia. *Ecology* 39:160–162.
- Wood, J. E. 1959. Relative estimates of fox population levels. *Journal of Wildlife Management* 23:53–63.
- Yearsley, E. F., and D. E. Samuel. 1980. Use of reclaimed surface mines by foxes in West Virginia. *Journal of Wildlife Management* 44:729–734.