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# Mammals

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## Overview

Many mammalian population studies have been initiated to determine a species' biological or ecological status because of its perceived economic importance, its abundance, its threatened or endangered state, or because it is viewed as our competitor. As a result, data on mammalian populations in North America have been amassed by researchers, naturalists, trappers, farmers, and land managers for years.

Inventory and monitoring programs that produce data about the status and trends of mammalian populations are significant for many reasons. One of the most important reasons, however, is that as fellow members of the most advanced class of organisms in the animal kingdom, the condition of mammal populations most closely reflects our condition. In essence, mammalian species are significant biological indicators for assessing the overall health of advanced organisms in an ecosystem.

Habitat changes, particularly those initiated by humans, have profoundly affected wildlife populations in North America. Though Native Americans used many wildlife species for food, clothing, and trade, their agricultural and land-use practices usually had minimal adverse effects on mammal populations during the pre-European settlement era. In general, during the post-Columbian era, most North American mammalian populations significantly declined, primarily because of their inability to adapt and compete with early European land-use practices and pressures.

Habitat modification and destruction during the settlement of North America occurred very slowly initially. Advances in agriculture and engineering accelerated the loss or modification of habitats that were critical to many species in climax communities. These landscape transformations often occurred before we had any knowledge of how these environmental changes would affect native flora and fauna. Habitat alterations were almost always economically driven and in the absence of land-use regulations and conservation measures many species were extirpated.

In addition to rapid and sustained habitat and landscape changes from agricultural and silvicultural practices, other factors such as unregulated hunting and trapping, indiscriminate predator and pest control, and urbanization also contributed significantly to the decline of once-bountiful mammalian populations. These practices, individually and collectively, have been directly correlated with the decline or extinction of many sensitive species.

The turn of the century brought a new focus on conservation efforts in this country.

Populations of some species, such as the white-tailed deer (*Odocoileus virginianus*), showed marked recovery after regulatory and conservation strategies began. Ardent wildlife management and conservation programs, started primarily for game species, have increased our knowledge and understanding of species and habitat interactions. Conservation programs have also positively affected many species that share habitat with the target species the programs are designed to aid. To complement these efforts, however, integrated regulatory legislation and conservation policies that specifically help sustain nontarget species and their habitats are still imperative.

The increased emphasis on the importance of managing for biological diversity and adopting an ecosystem approach to management has enhanced our efforts to move from resource-management practices that are oriented to single species to strategies that focus on the long-term conservation of native populations and their natural habitats. Thus, an integrated and comprehensive inventory and monitoring program that coordinates data on the status and trends of our natural resources is critical to successfully manage habitats that support a diverse array of plant and animal species.

This section provides knowledge on the status and trends of some higher vertebrate species that occupy some of this country's most diverse ecosystems. Many articles discuss historical and present species distribution, while others discuss the need for further research to fill our gaps of knowledge regarding the species. The articles cover a range of mammal species, some that have benefited greatly from past conservation efforts, and others that are now threatened or endangered, with the effort to recover them just beginning. Some species have been directly affected by habitat loss or modification, others by past hunting and trapping pressures.

We should not forget that our survival depends on wildlife, particularly higher vertebrates, nor should we forget that the status of wildlife populations serves as an advance indicator of overall environmental quality.

# Marine Mammals

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***Summarized from National Oceanic and Atmospheric Administration (1994)***

At least 35 species of marine mammals are found along the U.S. Atlantic coast and in the Gulf of Mexico: 2 seal species, 1 manatee, and 32 species of whales, dolphins, and porpoises (*see* Table 1 for status of selected species). Seven of these species are listed as endangered under the Endangered Species Act (ESA). At least 50 species of marine mammals are found in U.S. Pacific waters: 11 species of seals and sea lions; walrus; polar bear; sea otter; and 36 species of whales, dolphins, and porpoises; 11 species are listed as endangered or threatened under the ESA (*see* Table 2 for the status of selected species).

**Table 1.** Status of selected Atlantic and Gulf of Mexico coast species of marine mammals.

| Species and geographic area | Abundance | Status | Trends | Official status in designated U.S. waters |
|---|---|---|---|---|
| Fin whale, NE U.S. | 5,200 | Unknown | Unknown | Endangered* |
| Humpback whale, NW Atlantic | 5,100 (2,888-8,112) | Possibly 65% of 1850 population | Unknown | Endangered* |
| Northern right whale, NW Atlantic | 350 | Probably <5% of original number | Unknown | Endangered* |
| Pilot whales, NE U.S. | Unknown | Unknown | Unknown | |
| Bottlenose dolphin | | | | |
| NE U.S. coastal type | Unknown | Possibly down by 50% 1987-88 | Unknown | Depleted** |
| NE U.S. offshore type | 10,000-13,000 | Unknown | | |
| Gulf of Mexico (offshore and coastal types) | 35,000-45,000 | Possibly down by 50% 1987-88 | Unknown | |
| Whitesided dolphin, NE U.S. | 27,600 | Unknown | Unknown | |
| Spotted dolphin, NE U.S. | 200 | Unknown | Unknown | |
| Harbor porpoise, Gulf of Maine | 47,200 | Unknown | Unknown | Proposed as threatened* |
| Harbor seal, NE U.S. | 26,000 | Unknown | Increasing | |
| Beaked whales (six species in U.S. waters) | Unknown | Unknown | Unknown | |

*Endangered Species Act.
**Marine Mammal Protection Act.

## NMFS Assessments

The National Marine Fisheries Service (NMFS), an agency within the National Oceanic and Atmospheric Administration (NOAA), conducts research and status studies on many of these marine mammals under the authorities of the Magnuson Fisheries Conservation and Management Act, the Marine Mammal Protection Act (MMPA), and the ESA. The results of the status surveys include information required by the MMPA and the ESA on abundance (population size); status (as compared with historical levels or current viability); trends (changes in abundance); and status in U.S. waters. These results, published annually by NOAA, are the basis for this summary (NOAA 1994).

Estimates of abundance in U.S. waters are available for many, though not all, marine mammal species. Information on status and trends, however, is extremely limited because so little is known of the basic life history of many marine mammal species that scientists can determine neither status nor whether a population estimate represents a healthy, sustainable population. Moreover, long-term trends in many populations cannot be determined because historical population data are not available.

The NMFS provides assessments for 139 stocks (i.e., populations of species or groups of species that are treated together for management) of marine mammals; the status of 120 stocks is unknown, and trend data are only available for 19 stocks. The recently reauthorized MMPA requires the NMFS to conduct periodic assessments of marine mammal stocks that occur in U.S. waters. For this reason, better status and trends data are likely to become available over the next few years.

Abundance and status data for selected marine mammals are summarized in Table 1 (Atlantic species) and Table 2 (Pacific species). Trend data are mixed, but a number of conservation success stories have come from marine mammals. The bowhead and grey whales have shown significant population increases, as have California sea lions, the northern elephant seal, harbor seals in California, Oregon, Washington, and the Northeast, and the southern sea otter. These increases are largely the result of prohibition of commercial whaling by the International Whaling Commission (IWC) and by protection enacted under the MMPA and ESA. Other marine mammal populations, such as the Steller sea lion and the common dolphin in the eastern tropical Pacific, are still declining. Causes of decline in marine mammal populations include bycatch associated with commercial fishing, illegal killings, strandings, entanglement, disease, ship strikes, altered food sources, and possibly exposure to contaminants.

## Population Trends

### Whales

The eastern North Pacific stock of grey whale (*Eschrichtius robustus*) is rising (Fig. 1) and is one success story in species restoration. The NMFS estimates that the historical populations of grey whales in 1896 were around 15,000-20,000. While current population levels are below the estimated carrying capacity of 24,000, they appear higher than historical levels and represent a substantial gain. The population growth rate between 1968 and 1988 was 3.3% per year. After 3 years of review, on 15 June 1994, this species was removed from protection (delisted) under the ESA, an indication of successful management.

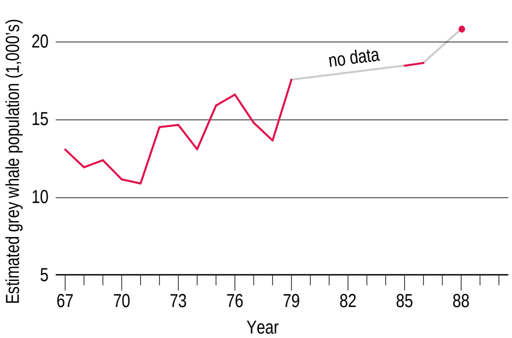


**Fig. 1.** Estimated population of grey whales, 1967-90 (NOAA 1994).

**Table 2.** Status of selected Pacific coast species of marine mammals.

| Species and area | Abundance | Status | Trends | Official status in designated U.S. waters |
|---|---|---|---|---|
| Fin whale | 935 | Unknown | Unknown | Endangered* |
| Humpback whale, E Pacific | ~1,400 | Probably less than 15% of 1850 population | Unknown | Endangered* |
| Northern right whale | Unknown | Unknown | Unknown | Endangered* |
| Bowhead whale, W. Arctic | 7,500 | About 40% of 1848 population size | Increasing at 3.1%/yr, 1978-88 | Endangered* |
| Grey whale | 20,869 (19,200-22,700) | Recovered to historical 1845 abundance levels | Increasing at 3.3%/yr, 1968-88 | Removed from ESA listing June 1994 |
| E. tropical Pacific dolphins | | | | |
| NE spotted | 731,000 | Depleted | Declining | |
| W/S spotted | 1,298,000 | Unknown | Stable | |
| Coastal spotted | 30,000 | Unknown | Stable | |
| E spinner | 631,800 | Depleted, 44% of late 1950's population | Stable | Depleted** |
| Whitebelly spinner | 1,019,000 | Unknown | Stable | |
| N common | 476,300 | Unknown | Declining | |
| Central common | 406,100 | Unknown | Stable | |
| S common | 2,210,900 | Unknown | Stable | |
| Common (pooled) | 3,093,300 | Unknown | Stable | |
| Striped | 1,918,000 | Unknown | Stable | |
| Harbor porpoise | | | | |
| SE Alaska | 2,052 | | | |
| W Gulf of Alaska | 1,273 | | | |
| N California | 10,000 | | | |
| Central California | 3,806 | | | |
| Inland Washington | 3,298 | | | |
| Oregon/Washington | 23,701 | | | |
| Hawaiian monk seal | 1,550 | Declined 50% since 1950's | Unknown, pup counts declining to variable | Endangered* |
| California sea lion (CA, OR, WA) | 111,016 | Unknown | Increasing 10.2%/yr since 1983 | |
| Harbor seal | | Unknown | Increasing? | |
| Alaska | 63,000 | | Declining | |
| California | 23,113 | | Increasing | |
| Oregon/Washington | 45,713 | | Increasing | |
| Northern fur seal | | | | |
| Pribilof Islands | 982,000 | < 40% of 1950's population | No significant trend since 1983 on St. Paul Is. | Depleted** |
| San Miguel | 6,000 | | Increasing | |
| Steller sea lion Northern Pacific | 116,000 | < 22% of late 1950's population | Declined 73% since 1960 | Threatened* |

*Endangered Species Act.
**Marine Mammal Protection Act.

The bowhead whale (*Balaena mysticetus*) is an endangered species that has shown a significant increase since the IWC adopted new rules in 1980 regulating its harvest for subsistence purposes by Native Americans (Fig. 2). The total prewhaling (before the mid-1800's) population of the bowhead whale is believed to have been 12,000-18,000; NMFS estimates that by 1900 it was probably in the low thousands. The current population of 7,500 is about 40% of its estimated 1848 population level (Table 2), more than 3 times the population low reached in 1980. The bowhead whale population has been growing by about 3% per year since 1978.

The endangered western North Atlantic population of right whales (*Eubalaena glacialis*) is considered by NMFS to be the only northern hemisphere right whale population with a significant number of individuals, about 300-350 animals (Table 1). Other stocks are considered virtually extinct: only five to seven sightings have been made in the last 25 years. Estimates

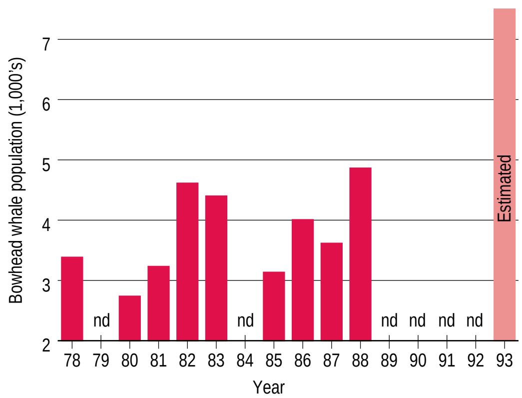


**Fig. 2.** Actual counts of bowhead whales, 1978-90 (NOAA 1994).

of the pre-18th century population are as high as 10,000. NMFS believes that human influences such as ship strikes and net entanglements are affecting about 60% of the population. The agency notes that the annual loss of even a single right whale has measurable effect on the population, by greatly inhibiting recovery of the species.

### Dolphins and Porpoises

The coastal migratory stock of Atlantic bottlenose dolphin (*Tursiops truncatus*) is listed as depleted under the MMPA (Table 1). This coastal stock incurred a loss of up to 50% during a 1987-88 die-off. Long-term trends are unknown, but the stock may require as many as 50 years to recover.

Harbor porpoises (*Phocoena phocoena*) occur on both U.S. coasts and are faring relatively well. The northwestern Atlantic harbor porpoise is found from Newfoundland, Canada, to Florida. The NMFS 1991-92 population estimate of the Gulf of Maine population is 47,200 (Table 1), but estimates of abundance for other populations do not exist. NMFS has found that harbor porpoise mortality from sink gill-net fisheries along the east coast of North America from Canada to North Carolina appears large compared with the species' natural reproduction rates. Management actions are being taken to address this issue, but long-term trends are unknown. On the west coast, NMFS's combined population estimate for northern California, Oregon, and Washington coastal stocks is 45,713.

The NMFS assesses 10 stocks of eastern tropical Pacific dolphins. Although population trends for most populations cannot be detected, the northeastern stocks of offshore spotted dolphin and the common dolphin may be declining (Table 2). These two stocks, as well as the eastern spinner and the striped dolphin, are incidentally taken in the international fishery for yellowfin tuna in the tropical Pacific waters off Mexico and Central America. Although mortality has been reduced in recent years, populations are still declining, or at best not increasing.

### Seals and Sea Lions

According to the NMFS, harbor seal (*Phoca vitulina*) populations have increased recently throughout much of their range because of protection by the MMPA. Recent NMFS surveys estimate that at least 26,000 harbor seals inhabit the Gulf of Maine (Table 1). Populations of California harbor seals are also increasing; a recent survey resulted in a count of about 23,000 harbor seals residing in the Channel Islands and along the California mainland (Table 2), an increase from about 12,000 in 1983. The population of harbor seals in Oregon and Washington has been estimated at 45,700, and is also increasing. Harbor seal counts in the Central Gulf of Alaska, however, have declined significantly in the past two decades; numbers are currently estimated by NOAA at 63,000 seals.

The northern fur seal (*Callorhinus ursinus*) is considered depleted under the MMPA. Production on one of its major breeding areas, Alaska's Pribilof Islands, dropped more than 60% between 1955 and 1980, but has since stabilized. The current population is less than 40% of the mid-1950's level; no significant trend in the Pribilof Islands population has been noted since 1983 (Table 2).

The northern sea lion or Steller sea lion (*Eumetopias jubatus*) is listed as threatened under the ESA. Species numbers have declined sharply throughout its range in the last 34 years (Table 2). The number of adults and juveniles in U.S. waters dropped from 154,000 in 1960 to 40,000 in 1992, a reduction of 73%. Most of this decline occurred in Alaska waters, and is believed due to a combination of factors, including incidental kills, illegal shooting, changes in prey availability and biomass, and perhaps other unidentified factors.

The U.S. population of California sea lions (*Zalophus californianus*) is increasing at a rate of about 10% annually. In 1990, NMFS estimated that the U.S. population was 111,000 individuals (Table 2). A number of human-related interactions, such as incidental take during fishing, entanglement, illegal killing, and pollutants, result in sea lion deaths.

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# Indiana Bats

*by*

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The Indiana bat (*Myotis sodalis*) is an endangered species that occurs throughout much of the eastern United States (Fig. 1). Although bats are sometimes viewed with disdain, they are of considerable ecological and economic importance. Bats consume a diet consisting largely of nocturnal insects and thereby are a natural control for both agricultural pests and insects that are annoying to humans. Furthermore, many forms of cave life depend upon nutrients brought into caves by bats in the form of guano or feces (Missouri Department of Conservation 1991).

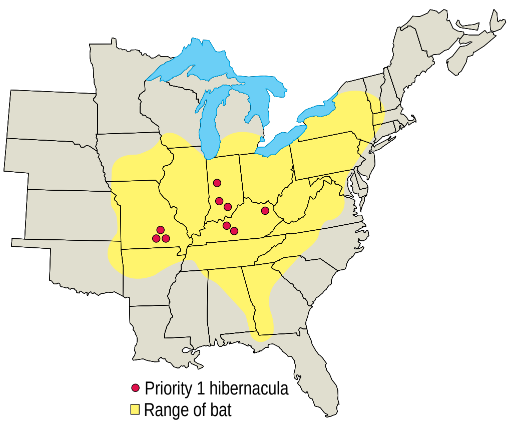


**Fig. 1.** Range of the Indiana bat and locations of Priority 1 hibernacula (*see* text for definitions).

Indiana bats use distinctly different habitats during summer and winter. In winter, bats congregate in a few large caves and mines for hibernation and have a more restricted distribution than at other times of the year. Nearly 85% of the known population winters in only seven caves and mines in Missouri, Indiana, and Kentucky, and approximately one-half of the population uses only two of these hibernacula.

In spring, females migrate north from their hibernacula and form maternity colonies in predominantly agricultural areas of Missouri, Iowa, Illinois, Indiana, and Michigan. These colonies, consisting of 50 to 150 adults and their young, normally roost under the loose bark of dead, large-diameter trees throughout summer; however, living shagbark hickories (*Carya ovata*) and tree cavities are also used occasionally (Humphrey et al. 1977; Gardner et al. 1991; Callahan 1993; Kurta et al. 1993).

As a consequence of their limited distribution, specific summer and winter habitat requirements, and tendency to congregate in large numbers during winter, Indiana bats are particularly vulnerable to rapid population reductions resulting from habitat change, environmental contaminants, and other human disturbances (Brady et al. 1983). Additionally, because females produce only one young per year, recovery following a population reduction occurs slowly. Concerns arising from the high potential vulnerability and slow recovery rate have led to a long-term population monitoring effort for this species.

## Bat Census Design

The first rangewide census of wintering Indiana bats was made in 1975. All subsequent population data were gathered according to standardized cave census techniques established by the Indiana Bat Recovery Team in 1983 (Brady et al. 1983). Data presented in this article are based upon counts made at 2-year intervals at Priority 1 hibernacula, which are caves where winter populations exceeding 30,000 bats have been recorded. We chose to use data only from Priority 1 caves because they contain the majority of bats in the population. During midwinter cave censuses, bats hanging singly and in small clusters of up to 25 were counted individually. The number of bats in larger clusters was determined by multiplying the surface area of the cluster by bat density (Fig. 2).

## Bat Populations: Trends and Recovery Prospects

Before the 1970's, the population status of Indiana bats was poorly understood because the locations of many of their winter hibernacula were unknown, and the counts that were conducted were made irregularly and inconsistently. The 1975 census established a benchmark of nearly 450,000 bats using Priority 1 hibernacula. Since 1983 the number of bats tallied has declined significantly, reaching a low of 347,890 during the most recent census in 1993 (Fig. 3).

Courtesy R. L. Clawson

**Fig. 2.** Hibernating cluster of Indiana bats.

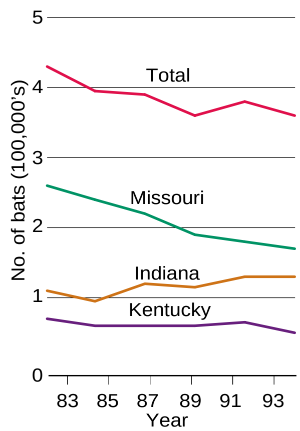


**Fig. 3.** State and national trends for Indiana bats, 1983-93.

Although the national trend indicates a 22% decline during the past 10 years, this decrease has not been consistent across the species' winter range (Fig. 3). Most of the decrease in the 10-year national census results can be accounted for by a precipitous 34% decline in the number of bats counted in Missouri. A more favorable pattern has been noted in Indiana, where numbers have increased, and in Kentucky, where the population has remained relatively stable.

Recovery efforts have included placing gates or fences across cave entrances to eliminate disturbances to hibernating bats. These exclusion devices have not halted population declines, suggesting that other factors are negatively influencing bat populations.

Another potential threat is the loss of habitat used by maternity colonies. Maternity roost sites in dead trees exposed to sunlight and located in upland forests and near streams are particularly important. Losses of these sites through streamside deforestation and stream channelization pose significant threats to population recovery.

Pesticides and other environmental contaminants represent additional hazards. Indiana bats are exposed to lingering residues of chlorinated hydrocarbon pesticides such as aldrin and heptachlor. These products have been banned since the 1970's, but persist in the soil and in insects upon which bats feed. Potential detrimental effects of the new generation of pesticides, including organophosphates, are unknown.

The long-term prognosis for Indiana bat populations is uncertain. The fact that wintering populations appear to be increasing in Indiana and are remaining relatively stable in Kentucky provides the basis for some optimism. A better understanding of their summer habitat requirements and factors affecting survival and reproduction is needed so that more effective recovery efforts can be formulated. It is important to recognize, however, that even if the factors that are negatively influencing Indiana bat populations are removed, recovery will occur slowly because this species has a low reproductive rate.

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# Gray Wolves

*by*
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The gray wolf (*Canis lupus*) originally occupied all habitats in North America north of about 20° north latitude (in Mexico), except for the southeastern United States, where the red wolf (*C. rufus*) lived. By 1960 the wolf was exterminated by federal and state governments from all of the United States except Alaska and northern Minnesota. Until recently, 24 subspecies of the gray wolf were recognized for North America, including 8 in the contiguous 48 states. After the gray wolf was listed as an endangered species in 1967, recovery plans were developed for the eastern timber wolf (*C.l. lycaon*), the northern Rocky Mountain wolf (*C.l. irremotus*), and the Mexican wolf (*C.l. baileyi*). The other subspecies in the contiguous United States were considered extinct.

The Eastern Timber Wolf Recovery Plan (U.S. Fish and Wildlife Service 1992) set as criteria for recovery the following conditions: a viable wolf population in Minnesota consisting of at least 200 animals, and either a population of at least 100 wolves in the United States within 160 km (100 mi) of the Minnesota population, or a population of at least 200 wolves if farther than 160 km (100 mi) from the Minnesota population. The Northern Rocky Mountain Wolf Recovery Plan (U.S. Fish and Wildlife Service 1987) defined recovery as when at least 10 breeding pairs of wolves inhabit each of three specified areas in the northern Rockies for 3 successive years. The Mexican Wolf Recovery Plan (U.S. Fish and Wildlife Service 1982) called for a self-sustaining population of at least 100 Mexican wolves in a 12,800-km$^2$ (4,941-mi$^2$) range.

A recent revision of wolf subspecies in North America (Nowak 1994), however, reduced the number of subspecies originally occupying the contiguous 48 states from eight to four. It classified the wolf currently inhabiting northern Montana as being *C.l. occidentalis*, primarily a Canadian and Alaskan wolf. It considered *C.l. nubilus* to be the wolf remaining in most of the range of the former northern Rocky Mountain wolf and the present range of the eastern timber wolf; this leaves the eastern timber wolf extinct in its former U.S. range, surviving now only in southeastern Canada. The new classification may have implications for the recovery criteria propounded by the Eastern Timber Wolf and Northern Rocky Mountain Wolf recovery plans. The reclassification did not change the status of the Mexican wolf.

This article is based on a review of the literature and recent personal communications. Most of the studies cited depended primarily on the use of aerial radio-tracking and observation (Mech 1974; Mech et al. 1988).

## Population Status by Region

### Lake Superior Region

After wolves were protected in 1974 by the Endangered Species Act of 1973, their numbers and distribution in Minnesota increased, and individuals began recolonizing Wisconsin (Mech and Nowak 1981). The population increased in Wisconsin and began recolonizing Michigan (Hammill 1993). The Minnesota population increased at about 3% per year (Fuller et al. 1992); its distribution continues to increase (Paul 1994). The best estimate of its current size is 1,740-2,030 wolves. Wisconsin and mainland Michigan each supported an estimated 50+ wolves in early 1994 (A.P. Wydeven, Wisconsin Department of Natural Resources, personal communication; J. Hammill, Michigan Department of Natural Resources, personal communication), and Isle Royale National Park about 14 wolves (Peterson 1994).

As wolves increased in Minnesota, they also began dispersing westward into North and South Dakota (Licht and Fritts 1994). The only records from these states involve 10 wolves killed from 1981 through 1992, but the possibility remains that small populations may occur in some of the more remote areas. Sufficient prey certainly exist there, so if dispersing wolves from Minnesota and Manitoba are not killed by humans, they should be able to breed and start populations.

### Western United States

Wolves were virtually absent in the western United States (other than an occasional animal that disperses from Canada) from the mid-1930's through 1980 (Ream and Mattson 1982). The nearest breeding population through this period was probably in Banff National Park, Alberta. Wolves were completely protected in extreme southeastern British Columbia in the 1960's (Pletscher et al. 1991). This led to recolonization of the area and adjacent northwestern Montana, and in 1986 a den was documented in Glacier National Park, Montana (Ream et al. 1989). This population, which straddles the Canadian border, has since grown to four packs and about 45 wolves.

Three breeding packs have been reported elsewhere in western Montana (Fritts et al. 1994), all probably founded by animals that dispersed from Glacier National Park. Additionally, an animal that dispersed from Glacier is in northeastern Idaho, and a wolf shot in 1992 just south of Yellowstone National Park was genetically related to Glacier wolves (Fritts et al. 1994). Animals that have dispersed, primarily from the Glacier area, have begun backfilling the area between Glacier National Park and Jasper National Park, Alberta (Boyd et al. 1994). This connection to larger wolf populations in Canada will enhance the viability of the U.S. population.

Courtesy L.D. Mech, NBS

Gray wolf (*Canis lupus*).

Although occasional wolves have been sighted in Wyoming and Washington and numerous sightings have been reported from central Idaho, no reproduction has been documented in these states, with the possible exception of litters in Washington in 1990 (S.H. Fritts, U.S. Fish and Wildlife Service, personal communication). An environmental impact statement on the reintroduction of wolves to Yellowstone and central Idaho was completed in early 1994.

### Factors Impeding Wolf Recovery

In small populations, the death of any individual can seriously impede recovery, meaning that factors that may not affect larger populations may hinder recovery of smaller ones. Such factors hindering the recovery of wolves include illegal and accidental killing of wolves by humans, canine parvovirus (Mech and Goyal 1993; Johnson et al. 1994; Wydeven et al. 1994), sarcoptic mange (A.P. Wydeven et al., Wisconsin Department of Natural Resources, personal communication), possibly Lyme disease (Thieking et al. 1992), and heartworm (*Dirofilaria immitis*; Mech and Fritts 1987). Of these, only killing by humans is subject to human control.

## Future Outlook

All wolf populations in the contiguous 48 states are increasing. Minnesota wolves occupy all suitable areas there and even have been colonizing agricultural regions where the Eastern Timber Wolf Recovery Team felt they should not be (U.S. Fish and Wildlife Service 1992). Thus, in 1993, the Department of Agriculture's Animal Damage Control Program destroyed a record 139 wolves for livestock depredation control (Paul 1994). As wolf populations continue to grow in other newly colonized areas, there may be an increasing need for control of those wolves preying on livestock (Fritts 1993). Because the public has so strongly supported wolf recovery and reintroduction, it may be difficult for many to understand the need for control. Thus, strong efforts at public education will be required.

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# Black Bears in North America

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Habitat loss, habitat fragmentation, and unrestricted harvest have significantly changed the distribution and abundance of black bears (*Ursus americanus*) in North America since colonial settlement. Although bears have been more carefully managed in the last 50 years and harvest levels are limited, threats from habitat alteration and fragmentation still exist and are particularly acute in the southeastern United States. In addition, the increased efficiency in hunting techniques and the illegal trade in bear parts, especially gall bladders, have raised concerns about the effect of poaching on some bear populations. Because bears have low reproductive rates, their populations recover more slowly from losses than do those of most other North American mammals.

Black bear populations are difficult to inventory and monitor because the animals occur in relatively low densities and are secretive by nature. Black bears are an important game species in many states and Canada and are an important component of their ecosystems. It is important that they be continuously and carefully monitored to ensure their continued existence.

## Black Bear Survey Data

Information on the distribution and status of black bears in North America came from several unpublished reports and scientific publications. *Traffic USA* (McCracken et al. 1995) reports periodically on the status of black bears in North America. Two reports on the status and conservation of the bears of the world were presented at meetings of the International Conference on Bear Research and Management in 1970 and 1989 (Cowan 1972; Servheen 1990). Finally, much of the information for this report is from data collected by survey for a report by the International Union for the Conservation of Nature and Natural Resources/Species Survival Commission (IUCN/SSC) Bear Specialist Group (Pelton et al. 1994).

## Range and Status

Black bears historically ranged over most of the forested regions of North America, including all Canadian provinces, Alaska, all states in the conterminous United States, and significant portions of northern Mexico (Hall 1981; Fig. 1). Their current distribution is restricted to relatively undisturbed forested regions (Pelton 1982; Pelton et al. 1994; Fig. 2). Black bears can still be found throughout Canada with the exception of Prince Edward Island (extirpated in 1937), and in at least 40 of the 50 states; their status in Mexico is uncertain (Leopold 1959; Fig. 2).

In the eastern United States black bear range is continuous throughout New England but becomes increasingly fragmented from the mid-Atlantic down through the Southeast (Maehr 1984). In the Southeast, most populations are now restricted to the Appalachian mountain chain or to coastal areas intermittently in all states from Virginia to Louisiana (J. Wooding, Florida Freshwater Fish and Game Commission, unpublished data).

**Fig. 1.** Historical distribution of the American black bear (modified from Hall 1981).

**Fig. 2.** Present distribution of the American black bear, based on survey responses from provinces and states (Pelton 1994) and research projects in Mexico (D. Doan, Texas A & I University, personal communication).

Recently, 11 Canadian provinces and territories reported stable black bear populations, and 10 provinces and territories estimated population sizes totaling about 359,000-373,000 (Pelton et al. 1994; McCracken et al. 1995; Table 1). Bears are legally harvested in all Canadian provinces and territories; total annual mortality from all sources (e.g., hunting, road kills, nuisance kills) is estimated at more than 23,000 (Pelton et al. 1994).

**Table 1.** Population estimates and trends of American black bears in Canada (adapted from Pelton et al. 1994).

| Province | Population estimate | Trend |
|---|---|---|
| Alberta | 39,600 | Stable |
| British Columbia | 121,600 | Stable |
| Manitoba | 25,000 | Stable |
| New Brunswick | Unknown | Stable/declining* |
| Newfoundland | 6,000-10,000 | Stable |
| Northwest Territories | 5,000+** | Stable |
| Nova Scotia | 3,000 | Stable |
| Ontario | 65,000-75,000 | Stable/increasing |
| Quebec | 60,000 | Stable |
| Saskatchewan | 24,000** | Stable |
| Yukon | 10,000 | Stable |
| Total | 359,200-373,200 | |

* Stable — East and Northeast; declining — West and Central.
**1991 or 1992 estimates from McCracken et al. (1995).

Thirty-eight of 40 states responding to a 1993 survey (Pelton et al. 1994) reported stable or increasing populations; only Idaho and New Mexico reported decreasing populations (Table 2). Based on data from 38 states, the total population estimate for black bears in the United States ranges from about 307,000 to 332,000 (excluding South Dakota and Wyoming). Black bears are listed as threatened or endangered in Florida, Louisiana, Mississippi, South Dakota, and Texas; rare in Missouri; and protected in Kentucky. They are unclassified in Connecticut. The remainder of the 40 states responding to the survey classify black bears as a game species (Table 2). In 1970 Arizona and Nevada listed black bears as a protected species and Texas listed them as game (Cowan 1972); thus the current classifications (Table 2) represent an upgrade in status for Arizona and Nevada and a downgrade for Texas. The status of bears in all remaining states covered in both surveys remained essentially unchanged.

The Southern Appalachian Region (Tennessee, North Carolina, South Carolina, and Georgia) is an area of special concern, and bear populations there have been routinely monitored since the late 1960's by the Southern Appalachian Bear Study Group. Initial estimates placed the population at 2,000-2,500 bears. The establishment of a network of black bear sanctuaries in the 1970's, scattered throughout the national forests in North Carolina, Tennessee, and Great Smoky Mountains National Park, provided protection for bears in the region, and estimates remain at 2,000-2,500 bears.

Courtesy J. Kasbohm

Black bear (*Ursus americanus*).

Two of 16 recognized subspecies of black bears (Hall 1981) require special mention: the Louisiana bear (*U.a. luteolus*), with a range of east Texas, all of Louisiana, and southern Mississippi; and the Florida bear (*U.a. floridanus*), with a range of Florida and southern Alabama. The U.S. Fish and Wildlife Service was petitioned in 1987 and 1990 to list the Louisiana bear and the Florida bear, respectively, as endangered species under the Endangered Species Act of 1973. In 1992 the Louisiana bear was officially placed on the federal endangered species list as a threatened species, and the Florida bear was placed in a "warranted but precluded" category. This latter category indicates that although biological evidence supports listing, several other species of higher priority are awaiting listing and will be listed before the Florida bear. At present, the U.S. Fish and Wildlife Service is considering listing bears in southern but not northern Florida.

Given the data available, the total minimum population of black bears reported in North America approaches 650,000-700,000. Total annual mortality (mostly from hunting) for the United States (more than 19,000) and Canada (more than 23,000) exceeds 42,000, which is less than 10% of the known population. Many state wildlife agencies accept that bear populations can sustain 20%-25% annual harvest mortality, with the understanding that some areas are more sensitive to overharvest than others. Thus, except for those in the southeastern United States and in Idaho and New Mexico, North American black bear populations appear stable or on the increase. Only concentrated research on isolated populations of bears remaining in the southeastern United States will answer questions concerning the long-term viability of those populations.

**Table 2.** Population estimates and trends of American black bears in the United States (adapted from Pelton et al. 1994).

| State | Estimated population size | Population trend | Status |
|---|---|---|---|
| Alabama | <50 | Stable | Game |
| Alaska | 100,000* | Stable | Game |
| Arizona | 2,500 | Stable | Game |
| Arkansas | 2,200 | Slightly increasing | Game |
| California | 20,000 | Slightly increasing | Game |
| Colorado | 8,000-12,000 | Unknown | Game |
| Connecticut | 15-30 | Increasing | Unclassified |
| Florida | 1,000-2,000 | Stable | Threatened |
| Georgia | 1,700 | Slightly increasing | Game |
| Idaho | 20,000-25,000* | Slightly decreasing | Game |
| Kentucky | <200 | Increasing | Protected |
| Louisiana | 200-400 | Slightly increasing | Threatened |
| Maine | 19,500-20,500 | Stable | Game |
| Maryland | 175-200 | Slightly increasing | Game |
| Massachusetts | 700-750 | Slightly increasing | Game |
| Michigan | 7,000-10,000 | Slightly increasing | Game |
| Minnesota | 15,000 | Increasing | Game |
| Mississippi | <50 | Slightly increasing | Endangered |
| Missouri | 50-130 | Increasing | Rare |
| Montana | 15,000-20,000 | Stable | Game |
| Nevada | 300 | Increasing | Game |
| New Hampshire | 3,500 | Increasing | Game |
| New Jersey | 275-325 | Increasing | Game |
| New Mexico | 3,000 | Decreasing | Game |
| New York | 4,000-5,000 | Slightly increasing | Game |
| North Carolina | 6,100 | Increasing | Game |
| Oklahoma | 120 | Increasing | Game |
| Oregon | 25,000 | Increasing | Game |
| Pennsylvania | 7,500 | Stable | Game |
| South Carolina | 200 | Slightly increasing | Game |
| South Dakota | Unknown | Unknown | Threatened |
| Tennessee | 750-1,500 | Increasing | Game |
| Texas | 50 * | Increasing | Threatened |
| Utah | 800-1,000 | Slightly increasing | Game |
| Vermont | 2,300 | Stable | Game |
| Virginia | 3,000-3,500 | Slightly increasing | Game |
| Washington | 27,000-30,000 | Increasing | Game |
| West Virginia | 3,500 | Increasing | Game |
| Wisconsin | 6,200 | Slightly increasing | Game |
| Wyoming | Unknown | Stable | Game |
| Total | 306,935-331,805 | | |

*1991 estimates from McCracken et al. (1995).

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# Grizzly Bears

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Grizzly bears (*Ursus arctos*) once roamed over most of the western United States from the high plains to the Pacific coast (Fig. 1). In the Great Plains, they seem to have favored areas near rivers and streams, where conflict with humans was also likely. These grassland grizzlies also probably spent considerable time searching out and consuming bison that died from drowning, birthing, or winter starvation, and so were undoubtedly affected by the elimination of bison from most of the Great Plains in the late 1800's. They are potential competitors for most foods valued by humans, including domesticated livestock and agricultural crops, and under certain limited conditions are also a potential threat to human safety. For these and other reasons, grizzly bears in the United States were vigorously sought out and killed by European settlers in the 1800's and early 1900's.

Between 1850 and 1920 grizzlies were eliminated from 95% of their original range, with extirpation occurring earliest on the Great Plains and later in remote mountainous areas (Fig. 1a). Unregulated killing of grizzlies continued in most places through the 1950's and resulted in a further 52% decline in their range between 1920 and 1970 (Fig. 1b). Grizzlies survived this last period of slaughter only in remote wilderness areas larger than 26,000 $km^2$ (10,000 $mi^2$). Altogether, grizzly bears were eliminated from 98% of their original range in the contiguous United States during a 100-year period.

Because of this dramatic decline and the uncertain status of grizzlies in areas where they had survived, their populations in the contiguous United States were listed as threatened under the Endangered Species Act in 1975. High levels of grizzly bear mortality in the Yellowstone area during the early 1970's were also a major impetus for this listing. Grizzly bears persist as identifiable populations in five areas (Fig. 1b): the Northern Continental Divide, Greater Yellowstone, Cabinet-Yaak, Selkirk, and North Cascade ecosystems. All these populations except Yellowstone's have some connection with grizzlies in southern Canada, although the current status and future prospects of Canadian bears are subject to debate. The U.S. portions of these five populations exist in designated recovery areas, where they receive full protection of the Endangered Species Act.

Grizzlies potentially occur in two other areas: the San Juan Mountains of southern Colorado and the Bitterroot ecosystem of Idaho and Montana. There are no plans for augmenting or recovering grizzlies in the San Juan Mountains, and serious consideration is being given to reintroducing grizzlies into the Bitterroots as an "experimental nonessential" population.

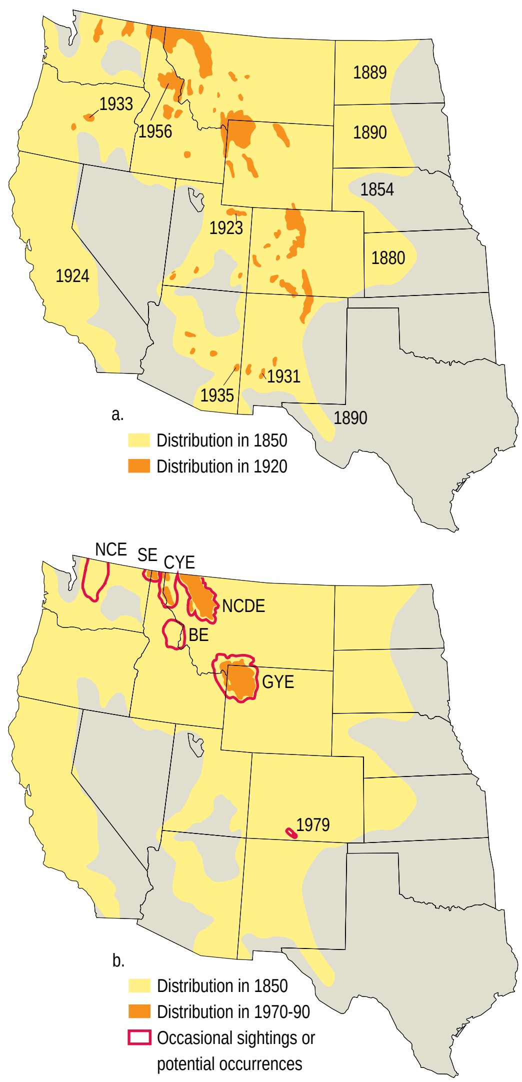


**Fig. 1.** Approximate distribution of grizzly bears in 1850 compared to 1920 (a; Merriam 1922) and 1970-90 (b). Local extinction dates, by state, appear in (a). Populations identified in (b) are NCE — North Cascades ecosystem, SE — Selkirk ecosystem, CYE — Cabinet-Yaak ecosystem, BE — Bitterroot ecosystem, NCDE — Northern Continental Divide ecosystem, GYE — Greater Yellowstone ecosystem. As indicated in (b), a grizzly was killed in the San Juan Mountains of southern Colorado in 1979.

## Status and Trends

Recent research in the Northern Continental Divide, Yellowstone, and Selkirk ecosystems has produced growth and size estimates for these grizzly bear populations. Study results, however, have been compromised by either small sample sizes, incomplete coverage, or possibly unrepresentative samples. These types of studies are also relatively expensive and require the capture and radio tagging of bears, although without the aid of radio tagging, it is even more difficult to directly count or otherwise monitor grizzly bear populations in their extensive, typically forested, ranges.

Because of these difficulties, we have only rough estimates of size for U.S. grizzly bear populations. Many grizzlies exist only in the Northern Continental Divide and Yellowstone ecosystems. We can be confident that there are at least 175 bears in the Northern Continental Divide ecosystem and 142 in the Yellowstone ecosystem, and a minimum of about 360 in the entire contiguous United States (Table). On the other hand, it is unlikely that more than 75 animals inhabit each of the Cabinet-Yaak, Selkirk, and North Cascade populations.

We have few reliable estimates of population trends for the same reasons that we have few reliable estimates of population size. In most cases we do not have any information on trends or the populations are so small (as in the Selkirks) that the death of only a few individuals can turn a growing population into a declining one (Table). Current best estimates for the Northern Continental Divide and Yellowstone areas suggest that these largest populations have been stable or slightly increasing in recent years. Even for these relatively well-studied populations, however, obtaining a reliable estimate of trends is difficult because of large and diverse study areas, small samples, and potentially biased observations.

Long-term viability of a population or species is achieved when there are enough animals and sufficient secure and productive habitat to ensure that the population or species will survive for the indefinite future. Certainly, direct mortality that accompanied the arrival of European settlers had catastropic consequences for grizzly bears. Other catastrophes related to disease, climate change, and changes in human values could yet be visited upon grizzlies.

Viability analysis is not an exact science, yet there are some rules of thumb that can be used to identify populations at substantially greater risk of extinction than others. For example, among existing isolated populations of brown bears (also *U. arctos)* and grizzly bears worldwide, only populations that were reduced to no fewer than about 450 bears responded with rapid growth when given protection. Conversely, even with protection, populations smaller than 200 continued to decline (Mattson and Reid 1991). All of these smaller populations also occupied areas less than 10,000 km$^2$ (3,900 mi$^2$) at the time they were given legal protection. This relationship between range size and vulnerability is consistent with the fact that only North American grizzly populations occupying areas larger than 26,000 km$^2$ (10,500 mi$^2$) in 1920 survived to the present. The Selkirk and Cabinet-Yaak ecosystems are about 5,200 km$^2$ (about 2,000 mi$^2$) and the remaining ecosystems are about 24,600-29,500 km$^2$ (about 9,500-11,400 mi$^2$). We expect populations with current ranges less than 29,500 km$^2$ (11,400 mi$^2$) to be at substantially greater risk of extinction.

Exchange of genes among individuals and populations is also important to survival of populations. Allendorf et al. (1991) estimated that populations of about 500 interbreeding grizzlies may be required to maintain normal levels of genetic diversity. This genetically effective population size equates to total population sizes of around 2,000 because not all bears breed. Given that the maximum documented movement of grizzly bears away from their mother's range is 45-105 km (28-65 mi; Blanchard and Knight 1991), it is unlikely that populations separated by a greater distance exchange breeding animals. Furthermore, bear movement across these gaps is entirely dependent upon their surviving often hostile conditions.

No grizzly bear population in the contiguous United States could be considered robust by our

**Table.** Recent population and trend estimates for areas in the contiguous United States occupied or potentially occupied by grizzly bears (NCDE — Northern Continental Divide ecosystem, GYE — Greater Yellowstone ecosystem, CYE — Cabinet-Yaak ecosystem, C — Cabinet portion only (95% confidence interval), SE — Selkirk ecosystem, NCE — North Cascades ecosystem, BE — Bitterroot ecosystem, SJE — San Juan ecosystem).

| | Minimum population estimate | | Population estimate assuming 60% sightability[a] | | | |
|---|---|---|---|---|---|---|
| Area | Average (mean) | Range (95% CI) | Average (mean) | Range (95% CI) | Trend estimate[b] | Long-term viability |
| NCDE[c] | 242[d] | 175-308 | 404 | 293-514 | Stable to slightly +[e] | ? |
| NCDE[f] | 302[d] | 219-384 | 502 | 365-640 | | |
| GYE[g] | 197[d] | 142-252 | 329 | 237-420 | ca. +0.01[h] | ? |
| CYE[i] | <15 | | 29 | 9-55 [j] | ? | Not viable |
| SE[k] | 26-36 [l] | | ? | | 0 to +0.02[k], recently-[m] | Not viable |
| NCE[n] | 10-20 [l] | | <50[l] | | ? | Not viable |
| BE | 0 | | Possible presence | | ? | Not viable |
| SJE | 0 | | Possible presence | | ? | Not viable |

[a]Based on results from Aune and Kasworm (1989) suggesting that 60% of adult females were observed in their study area. Accordingly, minimum population estimates are divided by 0.6.
[b]Expressed as an increasing (+) or decreasing (-) population, where available in terms of per capita rate of increase or decrease per year. A "?" indicates populations for which there are no substantive or reliable estimates of trend.
[c]Data from USFWS (1993) and MFWP (1993).
[d]Mean and 95% confidence intervals for 3-year sums of "unduplicated" adult females observed in an area ($n$ = 4 years, except for CYE $n$ = 3 years) minus known adult female mortality for the corresponding 3-year period, divided by 0.284 (the assumed proportion of adult females in the population) for NCDE and GYE.
[e]From Keating (1986), Aune and Kasworm (1989), McLellan (1989), and MFWP (1993).
[f]Using 22.8% adult females in the population and assuming a 1:1 adult sex ratio, based on the upper 95% confidence interval for estimates of percentage of adults in grizzly bear populations from the NCDE (MFWP 1993).
[g]Data from Knight et al. (1993).
[h]From Knight et al. (1988).
[i]Data for 1986-90 from MFWP (1993). Minimum population estimate is for the Cabinet portion only. Data from USFWS (1993).
[j]The lower confidence interval = 0, but 9 bears were radio-marked and known to be alive.
[k]From Wielgus (1993).
[l]Including bears in adjacent Canada.
[m]From R.B. Wielgus, University of British Columbia, Vancouver, personal communication (1994).
[n]From Almack et al. (1991).

rules of thumb for population viability. Clearly, the small populations of the North Cascade, Selkirk, and Bitterroot ecosystems, the San Juan Mountains, and the U.S. portion of the Cabinet-Yaak ecosystem are not viable. Although the North Cascade ecosystem is close to 26,000 $km^2$ (10,000 $mi^2$), its prospects are compromised by its isolation, even from populations in Canada. Similarly, although the Cabinet-Yaak and Selkirk populations can potentially receive bears that have dispersed from other populations, their 5,200-$km^2$ (2,000-$mi^2$) ranges are within the size boundaries of many U.S. populations that went extinct between 1920 and 1970 (Fig. 2) and are similar to those of European populations that appear to be declining toward extinction.

Prospects for the larger Northern Continental Divide and Greater Yellowstone populations are better but still uncertain. The Yellowstone population is probably no larger than 420 animals (Table) and is very isolated, making its long-term status tenuous. The Northern Continental Divide population probably has the best prospects because it is the largest population, in the largest area, and within the range of movement of other grizzly bear populations. Nonetheless, even this population is near the thresholds of 450 animals and the 26,000-$km^2$ (10,000-$mi^2$) range size historically associated with persistence of grizzlies in the United States and Europe.

The prognosis for the Selkirk, Cabinet-Yaak, and Northern Continental Divide populations might be improved if their connections with Canadian grizzly populations were considered. These southern Canadian grizzlies, however, do not have protection comparable to the U.S. Endangered Species Act and, outside of national parks, they are all hunted. There is also serious debate over the status of Canadian grizzly populations, especially in southwest Alberta and the northern Selkirks. Thus, there is no evidence that Canadian grizzlies will guarantee the long-term survival of neighboring U.S. populations.

## Implications

Since listing of the species under the Endangered Species Act in 1975, populations have probably stabilized in the Yellowstone and Northern Continental Divide ecosystems. Little if any of the former range has been reoccupied, however, and five of seven potential or existing populations do not have optimistic prospects, and even the two largest populations remain at risk.

About 88% of all grizzly bears that have been studied and died within the United States during the last 20 years were killed by humans, both legally and illegally. Humans remain the almost exclusive source of grizzly mortality, despite protection under the Endangered Species Act. Improved protection of these populations is accordingly dependent upon reducing the frequency of contact between grizzly bears and humans, primarily by managing levels of human activity in areas where we want grizzly bears to survive.

The Selkirk and Cabinet-Yaak grizzly bear populations may also need to be augmented by management if they are to survive beyond the next 100 years, whereas the North Cascade, Bitterroot, and San Juan populations will require the import of bears from elsewhere if they are to grow or persist even in the short term. The Yellowstone and Northern Continental Divide populations will need at least existing levels of protection, along with reliable monitoring and timely management.

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# Black-footed Ferrets

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The black-footed ferret (*Mustela nigripes*) was a charter member of endangered species lists for North America, recognized as rare long before the passage of the Endangered Species Act of 1973. This member of the weasel family is closely associated with prairie dogs (*Cynomys* spp.) of three species, a specialization that contributed to its downfall. Prairie dogs make up 90% of the ferret diet; in addition, ferrets dwell in prairie dog burrows during daylight, venturing out mostly during darkness. Trappers captured black-footed ferrets during their quests for other species of furbearers. Although the species received increased attention as it became increasingly rare, the number of documented ferrets fell steadily after 1940 (Fig. 1), and little was learned about the animals before large habitat declines made studies of them difficult. These declines were brought about mainly by prairie dog control campaigns begun before 1900 and reaching high intensity by the 1920's and 1930's.

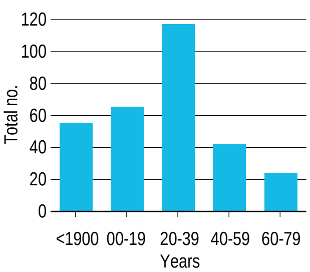



**Fig. 1.** Black-footed ferrets collected before 1980.

Much of what is known about black-footed ferret biology was learned from research during 1964-74 on a remnant population in South Dakota (Linder et al. 1972; Hillman and Linder 1973), and from 1981 to the present on a population found at Meeteetse, Wyoming, and later transferred to captivity (Biggins et al. 1985; Forrest et al. 1988; Williams et al. 1988). Nine ferrets from the sparse South Dakota population (only 11 ferret litters were located during 1964-72) were taken into captivity from 1971 to 1973, and captive breeding was undertaken at the U.S. Fish and Wildlife Service's Patuxent Wildlife Research Center in Maryland (Carpenter and Hillman 1978). Although litters were born there, no young were successfully raised. The last of the Patuxent captive ferrets died in 1978, and no animals were located in South Dakota after 1979.

Black-footed ferrets were "rediscovered" in prairie dog complexes at Meeteetse in 1981, giving conservationists what seemed a last chance to learn about the species and possibly save it from extinction. That population remained healthy (70 ferret litters were counted from 1982 to 1986) through 1984 (Fig. 2), a period when much was learned about ferret life history and behavior. In 1985, sylvatic plague, a disease deadly to prairie dogs, was confirmed in the prairie dogs at Meeteetse, creating fear that the prairie dog habitat vital for ferrets would be lost. In addition, field biologists were reporting a substantial decrease in the number of ferrets detected. The fear of plague was quickly overshadowed by the discovery of canine distemper in the ferrets themselves. It is a disease lethal to ferrets.

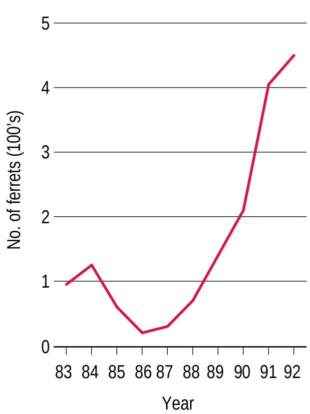



**Fig. 2.** Black-footed ferret population from Meeteetse, Wyoming, 1983-92 (all captive from 1986 to present).

In 1985 six ferrets were captured to begin captive breeding, but two of them brought the distemper virus into captivity, and all six died (Williams et al. 1988). A plan was formulated to place more animals from Meeteetse into captivity to protect them from distemper and to start the breeding program. By December 1985, only 10 ferrets were known to exist, 6 in captivity and 4 at Meeteetse. The following year, the surviving free-ranging ferrets at Meeteetse produced only two litters, a number thought too small to sustain the wild population. Because both the Meeteetse and captive populations were too small to sustain themselves, all remaining ferrets were removed from the wild, resulting in a captive population of 18 individuals by early 1987.

Courtesy D. Biggins, NBS

Black-footed ferrets, almost extinct by 1985, are being reintroduced from captive breeding but still lack genetic diversity.

Captive breeding of ferrets eventually became successful (Fig. 2). Although the captive population is growing, researchers fear the consequences of low genetic diversity (already documented by O'Brien et al. 1989) and of inbreeding depression (*see* glossary). A goal of the breeding program is to retain as much genetic diversity as possible, but the only practical way to increase diversity is to find more wild ferrets. In spite of intensive searches of the remaining good ferret habitat and investigations of sighting reports, no wild ferrets have been found.

The captive breeding program now is producing sufficient surplus ferrets for reintroduction into the wild; 187 ferrets were released into prairie dog colonies in Shirley Basin, Wyoming, during 1991-93. Challenges facing the black-footed ferret reintroduction include low survivorship of released ferrets due to high dispersal and losses to other predators; unknown influence of low genetic diversity; canine distemper hazard; indirect effect of plague on prairie dogs and possible direct effect on ferrets; and low availability of suitable habitat for reintroduction. The scarcity of habitat reflects a much larger problem with the prairie dog ecosystem and needs increased attention.

At the turn of this century, prairie dogs reportedly occupied more than 40 million ha

(100 million acres) of grasslands, but by 1960 that area had been reduced to about 607,500 ha (1.5 million acres; Marsh 1984). Much reduction was attributed to prairie dog control programs, which continue. For example, in South Dakota in the late 1980's, $6.2 million was spent to apply toxicants to prairie dog colonies on Pine Ridge Indian Reservation (Sharps 1988). At least two states (Nebraska and South Dakota) have laws prohibiting landowners from allowing prairie dogs to flourish on their properties; if the land manager does not "control" the "infestation," the state can do so and bill expenses to the owner (Clarke 1988).

Sylvatic plague also has been devastating to prairie dogs and was the likely cause of the dramatic decline in prairie dogs at Meeteetse. Although the Meeteetse complex recently supported the densest and most vigorous population of black-footed ferrets ever known, it cannot be considered as ferret habitat now because of plagued-induced losses of prairie dogs. Plague is present in most of the monitored white-tailed prairie dog (*Cynomys leucurus*) complexes, including the Shirley Basin ferret reintroduction site (Table). The plague's persistence could be responsible for the generally lower densities of white-tailed prairie dogs (averaging fewer than seven prairie dogs per hectare or fewer than three per acre).

Several prairie dog complexes have been evaluated as sites for reintroduction of black-footed ferrets (Table). The evaluation involves grouping clusters of colonies separated by fewer than 7 km (4.3 mi) into complexes, based on movement capabilities of ferrets (Biggins et al. 1993); these areas include some of the best prairie dog complexes remaining in the states. Nevertheless, other extensive prairie dog complexes were not considered for ferret reintroduction.

Most of the original range of the black-footed ferret was associated with black-tailed prairie dog (*Cynomys ludovicianus*) complexes, which now exhibit the highest population densities of all prairie dogs (Table). Black-footed ferret reintroductions recently began at black-tailed prairie dog complexes near Malta, Montana, and Badlands National Park, South Dakota (Table). At present, the best example of a large complex of black-tailed prairie dogs is near Nuevos Casas Grandes, Chihuahua, Mexico (Table). It supports an impressive associated fauna and is a potential reintroduction site for black-footed ferrets.

Ramifications of a healthy prairie dog ecosystem extend well beyond black-footed ferrets. The prairie dog is a keystone species of the North American prairies. It is an important primary consumer, converting plants to animal biomass at a higher rate than other vertebrate herbivores of the short-grass prairies, and its burrowing provides homes for many other species of animals and increases nutrients in surface soil. This animal also provides food for many predators. We estimated it takes 700-800 prairie dogs to annually support a reproducing pair of black-footed ferrets and a similar biomass of associated predators (Biggins et al. 1993), suggesting that large complexes of prairie dog colonies are necessary to support self-sustaining populations of these second-order consumers.

The 98% loss of the productive prairie dog ecosystem has not yet motivated legal protection or plans for management. There is no federal legislation directly promoting the welfare of the prairie dog ecosystem (even on public lands), and the only existing state legislation promotes poisoning.

To develop a plan for remedial action, several immediate research needs are apparent in the prairie dog ecosystem: determine the relative diversity and abundance of invertebrates and

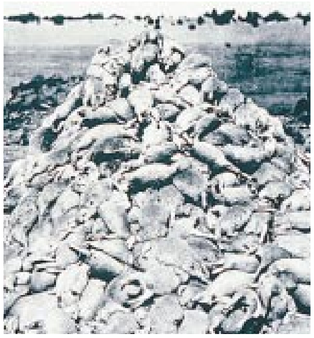
Courtesy D. Biggins, NBS

Three species of prairie dogs make up 90% of the black-footed ferret's diet; prairie dog burrows are also used by the ferrets during the day.

**Table.** Prairie dog complexes evaluated for black-footed ferret reintroductions. (Some data from Black-footed Ferret Interstate Coordinating Committee.)

| State | Site | Prairie dog species* | Complex size (ha) | Hectares of prairie dogs | Prairie dogs estimate | Prairie dogs/ha colony |
|---|---|---|---|---|---|---|
| **United States** | | | | | | |
| Arizona | Aubrey Valley | Gunnison's | 44,167 | 7,390 | 34,067 | 4.61 |
| Colorado | Little Snake | White-tailed | 252,075 | 31,624 | 36,875 | 1.17 |
| | Wolf Creek | White-tailed | 65,607 | 3,174 | 20,009 | 6.30 |
| | Sterling | Black-tailed | 57,824 | 2,366 | 16,786 | 7.10 |
| Montana | Custer Creek | Black-tailed | 38,879 | 425 | 16,750 | 39.39 |
| | Malta Bureau of Land Management | Black-tailed | 583,430 | 7,600 | 167,299 | 22.01 |
| | Charles M. Russell Refuge | Black-tailed | 28,508 | 896 | 22,371 | 25.00 |
| North Dakota | Roosevelt National Park | Black-tailed | 14,126 | 594 | 39,270 | 66.11 |
| | Marmarth National Park | Black-tailed | 7,257 | 548 | 21,208 | 38.70 |
| | Fort Yates | Black-tailed | 6,739 | 579 | 20,823 | 35.96 |
| South Dakota | Badlands National Park | Black-tailed | 17,016 | 1,669 | 74,081 | 44.39 |
| Wyoming | Meeteetse | White-tailed | 53,846 | 5,111 | 1,299 | 0.25 |
| | Shirley Basin | White-tailed | 48,987 | 20,612 | 75,155 | 3.65 |
| | Medicine Bow | White-tailed | 74,958 | 27,235 | 24,492 | 0.90 |
| | Recluse | Black-tailed | 98,802 | 7,181 | 59,895 | 8.34 |
| | Bolton Ranch | White-tailed | 28,068 | 4,420 | 7,858 | 1.78 |
| | Kinney Rim | White-tailed | 43,509 | 7,220 | 608 | 0.08 |
| **Mexico** | | | | | | |
| Chihuahua | Nuevos Casas Grandes | Black-tailed | 87,866 | 54,541 | 994,986 | 18.24 |

*Gunnison's prairie dog (*Cynomys gunnisoni*), white-tailed prairie dog (*C. leucurus*), and black-tailed prairie dog (*C. ludovicianus*).



Courtesy B. Miller

Prairie dog control campaigns, like this one in Arizona, circa 1913, contributed to the decline of the black-footed ferret.

small- to medium-sized vertebrates on prairie dog complexes, as well as the degree of dependence on prairie dogs of selected associated species; examine the effect of complex size, as well as constituent colony sizes, numbers, and juxtaposition on diversity and abundance of associated species; investigate the recent history of plague on selected complexes to determine the relation between complex size (and morphology) and resistance to decimation by plague; and develop methods for reestablishing prairie dog colonies and reconstructing complexes in suitable areas where prairie dogs have been extirpated.

The black-footed ferret cannot be reestablished on the grasslands of North America in viable self-sustaining populations without large complexes of prairie dog colonies. The importance of this system to other species is not completely understood, but large declines in some of its species should serve as a warning. The case of the black-footed ferret provides ample evidence that timely preventive action would be preferable to the inefficient “salvage” operations. Furthermore, there is considerable risk of irreversible damage (e.g., genetic impoverishment) with such rescue efforts.

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# American Badgers in Illinois

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The American badger (*Taxidea taxus*) is a medium-sized carnivore found in treeless areas across North America, such as the tallgrass prairie (Lindzey 1982). Badgers rely primarily on small burrowing mammals as a prey source; availability of badger prey may be affected by changes in land-use practices that alter prey habitat. In the midwestern United States most native prairie was plowed for agricultural use beginning in the mid-1800’s (Burger 1978). In the past 100 years, Midwest agriculture has shifted from a diverse system of small farms with row crops, small grains, hay, and livestock pasture to larger agricultural operations employing a mechanized and chemical approach to cropping. The result is a more uniform agricultural landscape dominated by two primary row crops, corn and soybeans. The effects of such land-use alterations on badgers are unknown. In addition, other human activities such as hunting and trapping have no doubt had an impact on native vertebrates such as the badger. Our ongoing study was initiated to determine the distribution and status of badgers in Illinois.

Trends in carnivore abundance are difficult to evaluate because most species are secretive or visually cryptic. Trapping records, one of the earliest historical data sources for furbearers, are virtually nonexistent for badgers in the 1800’s (Obbard et al. 1987). In Illinois, badgers have been protected from harvest since 1957. Furthermore, population estimates derived from furbearer harvest data are complicated by market price bias (Erickson 1982). Thus, data for estimating long-term population trends in Illinois badgers are few and flawed. Our approach is to document and evaluate current

Courtesy M. Georgi, Illinois Natural History Survey

American badger (*Taxidea taxus*).

population parameters, behavior, and habitat use in the context of present and historical habitat quality and availability.

Most research on badgers has been limited to the western United States. Although results have varied somewhat among these studies, average densities (estimated subjectively from mark-recapture and home range data) have ranged from 0.38 to 5 badgers/km$^2$ (0.98-12.95 badgers/mi$^2$). We use radio telemetry to collect intensive data at a field site in west-central Illinois. Preliminary results suggest that individual badger home range size in Illinois is an order of magnitude larger than that of western badgers, implying that badger density in Illinois is much lower. The home range size estimates of two badgers in Minnesota were also larger than those reported for western states (Sargeant and Warner 1972; Lampe and Sovada 1981).

More than 65% of the Illinois landscape is under intensive row-crop agriculture (Neely and Heister 1987). Although badger prey exist throughout Illinois, available prey in row crops is limited to small species such as the deer mouse (*Peromyscus maniculatus*), which occur in low uniform densities. Important prey species reported in the West, such as ground squirrels (*Spermophilus* spp.), have average densities similar to Illinois deer mice, but they are much larger animals and may be concentrated into easily hunted loose colonies (Messick and Hornocker 1981; Minta 1990).

In Illinois, badgers appear to use most frequently cover types that are relatively undisturbed by plowing, including hayfields, pastures, and linear habitats such as roadsides and fencelines. These habitats offer the greatest concentration of small mammalian prey and the lowest frequency of agricultural disturbance. If badgers are limited by available prey, it is possible that the current badger population density is lower than when native prairie and its accompanying prey species' populations dominated the landscape.

Although badgers are legally protected in Illinois, human-induced mortality such as vehicle collisions and agricultural accidents take a toll on populations. Large predators that might prey on adult badgers, such as the black bear (*Ursus americanus*), gray wolf (*Canis lupus*), and mountain lion (*Felis concolor*), have been extirpated since the 19th century (Hoffmeister 1989). However, our study shows that predation by coyotes (*Canis latrans*) and domestic dogs significantly affects juvenile badgers; fewer than 70% of juveniles survive to dispersal, reducing overall recruitment.

The badger's range may be expanding eastward from its former boundaries within the Midwest; observations of range expansion in Missouri, southern Illinois, Indiana, and Ohio suggest that agricultural practices have converted previously forested acres to more suitable badger habitat (Moseley 1934; Leedy 1947; Mumford 1969; Hubert 1980; Mumford and Whitaker 1982; Long and Killingley 1983; Gremillion-Smith 1985; Whitaker and Gammon 1988).

Our study revealed that badgers are distributed and breeding throughout Illinois. The dynamics of badger range expansion are difficult to pinpoint, in part because of the cryptic nature of the species. In Illinois and probably the agricultural Midwest in general, individual badgers move over such large areas that live sightings or indications of badger presence are few and far between. Opportunistic observations to evaluate local badger distribution underestimate geographic range; thus, a focused regionwide attempt to evaluate badger range in the Midwest might demonstrate a wider distribution than expected.

Badgers in Illinois appear to be a species with intermediate status: though they are neither abundant nor of high economic value, they are widely distributed and have adapted to a greatly altered environment. Understanding what factors cause a species such as the badger to become more or less abundant is vitally important in conservation biology and wildlife management.

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# California Sea Otters

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Information on the size, distribution, and productivity of the California sea otter population is broadly relevant to two federally mandated goals: removing the population's listing as threatened under the Endangered Species Act (ESA) and obtaining an "optimal sustainable population" under the Marine Mammal Protection Act. Except for the population in central California, sea otters (*Enhydra lutris*) were hunted to extinction between Prince William Sound, Alaska, and Baja California (Kenyon 1969). Wilson et al. (1991), based on variations in cranial morphology, recently assigned subspecific status (*E. l. nereis*) to the California sea otter. Furthermore, mitochondrial DNA analysis has revealed genetic differences among populations in California, Alaska, and Asia (NBS, unpublished data).

In 1977, the California sea otter was listed as threatened under the ESA, largely because of its small population size and perceived risks from such factors as human disturbance, competition with fisheries, and pollution. Because of unique threats and growth characteristics, the California population is treated separately from sea otter populations elsewhere in the North Pacific.

Courtesy D. Buchich

Sea otter (*Enhydra lutris*).

## Survey Design

Data on the size and distribution of the California sea otter population have been gathered for more than 50 years. In 1982 we developed a survey technique in which individuals in most of the California sea otter's range are counted from shore by groups of two observers using binoculars and spotting scopes. Supplemental data for each sighting include group size, activity, number and size of pups, and habitat. Areas that cannot be counted from shore are surveyed from a low-flying aircraft. Rangewide surveys are done in late spring and mid-autumn.

## Population Trends, 1914-93

The California sea otter population has increased steadily through most of the 1900's (Fig. 1). Rate of increase was about 5% per year until the mid-1970's. Although only one survey was completed between 1976 and 1982, the collective data suggest that population growth had ceased by the mid-1970's, and that the population may have declined by as much as 30% between the mid-1970's and early 1980's. Counts made since 1983 have increased at about 5%-6% per year. In spring 1993, 2,239 California sea otters were counted.

The California sea otter's lineal range (distance along the 9-m [5-fathom] isobath between the northernmost and southernmost sightings) has also increased, although more slowly and erratically than the population size (data summarized by Riedman and Estes 1990). The

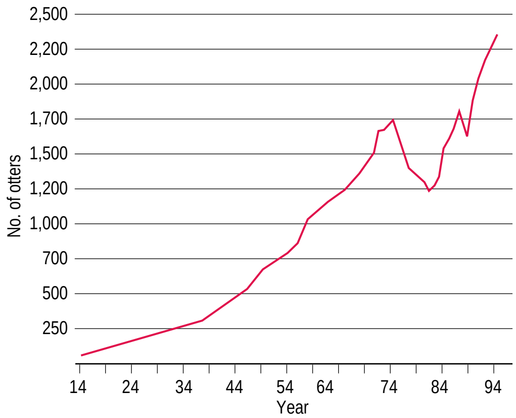


**Fig. 1.** Trends in abundance of the California sea otter population, 1914-93.

direction of range expansion was predominately southward before 1981, but northward thereafter. Comparison between spring surveys conducted in 1983 and 1993 (Fig. 2) is sufficient to draw several conclusions. First, the population's range limits changed little during this 10-year period, even though large numbers of individuals accumulated near the range peripheries. Second, population density increased throughout this time, although rates of increase were lowest near the center of the range. Finally, the relative abundance of individuals has remained largely unchanged (compare Fig. 2a [1983] with Fig. 2b [1993], noting the similarity in forms of distributions for kilometer segments 10-21).

Although the number of dependent pups counted in spring surveys almost doubled between 1983 and 1993, the geographic range within which these pups were born has changed very little (Fig. 2). Rate of annual pup production ranged from 0.14 to 0.28, but in most years it varied between 0.18 and 0.21. There are no obvious trends in rate of annual pup production between 1983 and 1993. Although the incremental change in the population from one year to the next appeared positively related to the annual number of births, this relationship cannot be shown to be statistically significant.

## Implications

From the mid-1970's to the early 1980's, the California sea otter population ceased growing and probably declined. Entanglement mortality in a coastal set-net fishery was the likely cause of this decline (Wendell et al. 1985). Restrictions were imposed on the fishery in 1982, and the population apparently responded by resuming its prior rate of increase.

The maximum rate of increase for sea otter populations is about 20% per year. Except for the California otters, all increasing populations for which data are available have grown at about this rate (Estes 1990). These patterns, coupled with the absence of any size- or density-related reduction in growth rates, make the relatively slow rate of increase in the California population perplexing.

Although the ultimate reason for disparate growth rates among sea otter populations is unknown, we believe that causes relate more to increased mortality than diminished reproduction. While it is difficult to compare population-level reproductive rates between sea otters in Alaska and California, longitudinal studies of

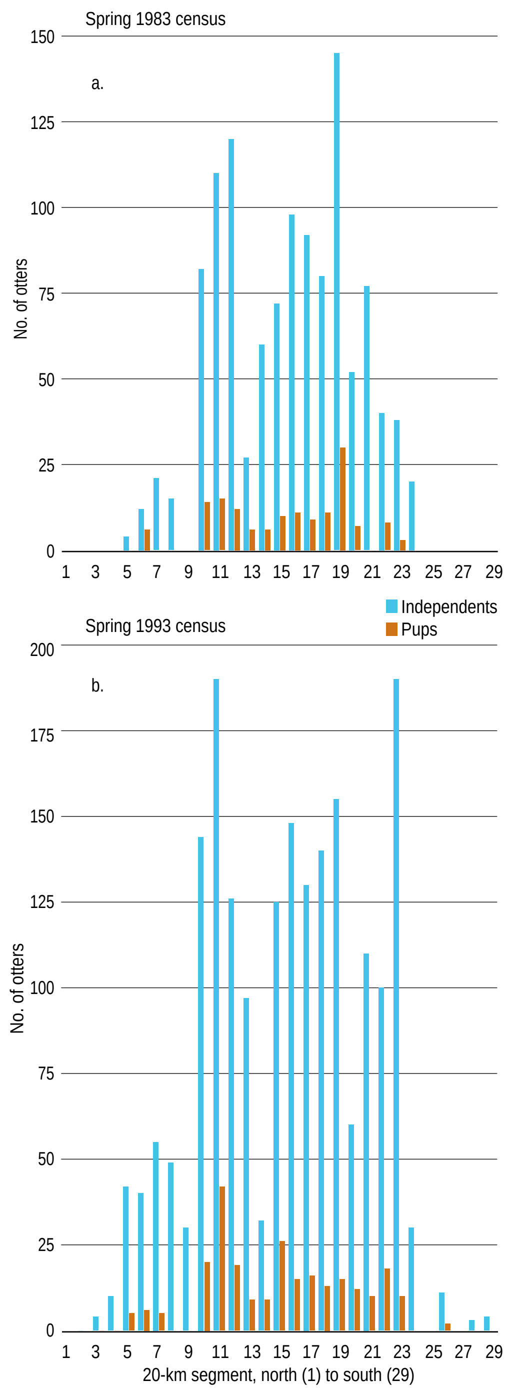


**Fig. 2.** Distribution and abundance of California sea otters in 1983 (a) and 1993 (b). Data are from the spring surveys.

marked individuals in the two regions indicate that both age of first reproduction and annual birth rate of adult females are similar. Furthermore, the close similarity between the theoretical maximum rate of increase and observed rates of population increase for sea otters in Washington, Canada, and portions of Alaska suggests that mortality from birth to senescence in these populations is quite low. In contrast, rates of mortality in the California sea otter are comparatively high, with an estimated 40%-50% of newborns lost before weaning (Siniff and Ralls 1991; Jameson and Johnson 1993; Riedman et al. 1994). This alone would significantly depress a population's potential rate of increase. Furthermore, the age composition of beach-cast carcasses in California indicates that most postweaning deaths occur well in advance of physiological senescence (Pietz et al. 1988; Bodkin and Jameson 1991). These patterns likely explain the depressed rate of increase in the California sea otter population.

Although the demographic patterns of mortality in California sea otters are becoming clear, the causes of deaths remain uncertain. There is growing evidence for the importance of predation by great white sharks (*Carcharodon carcharias*). Contaminants may also be having a detrimental effect on California sea otters, although as yet there is no direct evidence for this. However, polychlorinated biphenyl (PCB) and DDT levels, known to be high in the California Current, are also high in the liver and muscle tissues of California sea otters (Bacon 1994). Of particular concern are that average PCB levels in California sea otters approach those that cause reproductive failure in mink, which are in the same family as otters; and preweaning pup losses are especially high in primiparous (*see* glossary) females. This latter point may be significant because environmental contaminants that accumulate in fat can be transferred via milk in extraordinarily high concentrations, especially to the first-born young in species such as the sea otter which has prolonged sexual immaturity.

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# White-tailed Deer in the Northeast


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Populations of white-tailed deer (*Odocoileus virginianus*) have changed significantly during the past 100 years in the eastern United States (Halls 1984). After near extirpation in the eastern states by 1900, deer numbers increased during the first quarter of this century. The effects of growing deer populations on forest regeneration and farm crops have been a concern to foresters and farmers for the past 50 years.

In recent years, deer management plans have been designed to maintain deer populations at levels compatible with all land uses. Conflicts, however, between deer and forest management or agriculture still exist in the Northeast. Areas that were once exclusively forests are now a mixture of forest, farm, and urban environments that create increased interactions and conflicts between humans and deer, including deer-vehicle collisions. Management of deer near urban environments presents a unique challenge for local resource managers (Porter 1991).

This report describes trends in abundance of white-tailed deer in the northeastern United States, relationships between harvest and population estimates, and conflicts between deer and other resources.

## Data Surveys

We contacted biologists in each of 13 northeastern states to acquire estimates of deer population size, harvest, and deer-vehicle collisions. We featured harvest data for antlered deer from all 13 states to describe deer abundance during 1983-92, as well as data from selected states to describe relations between deer harvests and population size.

Biologists in the northeastern states also provided information on trends in reported conflicts between deer and land use and other natural resources. We determined the proportion of states that expressed conflicts for particular categories such as deer and agriculture, deer and forestry, or deer and other resources.

## Population Estimates and Management Implications

White-tailed deer populations have increased in all 13 northeastern states during 1983-92, based on either population estimates or number of antlered deer harvested. Population estimates for nine states indicated an increase from less than 1.5 million in the early 1980's to 1.8 million in the early 1990's (Fig. 1). Deer density in the deer range of these states

## Deer Management at Parks and Refuges

Even though states are responsible for managing deer within their boundaries, they do not control all land areas. The level of management for a state may be an ecological or political unit. However, states usually lack data on deer and their habitats for small units such as municipalities, parks, refuges, or military facilities, and they are not directly responsible for management of these special areas. Presented here are examples of two state parks, two national parks and a national historic site, and three national wildlife refuges.

### Parks

Ridley Creek and Tyler state parks in Pennsylvania provide two examples of where attempts have been made to manage high deer densities in and around urban areas. Such high densities pose significant problems because of deer feeding on ornamental plants and deer-vehicle collisions. At Ridley Creek State Park, a 1,052-ha (2,600-acre) area near Philadelphia, hunters harvested 97-344 deer per year during eight controlled hunts held between 1983 and 1992. From 160 to 491 deer were observed during annual counts made from helicopters (no count was made in 1990). A count of 491 in 1983 indicated that the deer density was in excess of 46.7 deer/km$^2$ (121 deer/mi$^2$) in the park. Hunter harvests resulted in a significant herd reduction, as 160 deer were counted in 1992 compared to 491 in 1983.

Controlled hunts were conducted during 4 years—1987, 1988, 1989, and 1991—at Tyler State Park in eastern Pennsylvania. The hunts in December 1987 and January 1988 yielded a kill of 487 deer; this number equates to 70.3 deer harvested per km$^2$ (182 deer/mi$^2$) on the 692-ha (1,710-acre) park. During 1987, 455 deer were counted during aerial surveys compared to 49 during 1992, indicating that controlled hunts resulted in a significant reduction in deer abundance at Tyler State Park.

### National Parks

The 2,335-ha (5,770-acre) Catoctin Mountain National Park, administered by the National Park Service in Maryland, has been noticeably affected by deer since at least 1981. Estimates of deer density indicated an increase from 9.6 to 23.5 deer/km$^2$ (25 to 61 deer/mi$^2$) between 1986 and 1989. The presence of deer at this density has led to concern over the effect of deer on native plants, including rare species. The National Park Service is preparing an environmental assessment to review various management alternatives and to select a strategy to manage deer at Catoctin Mountain Park. Unlike in state parks, harvest of deer from National Park Service lands is difficult, if not illegal, to implement; hence, management options are more limited.

Estimates of deer abundance at Gettysburg National Military Park and Eisenhower National Historic Site from 1987 through 1992 indicated an increase from 721 to 1,018 deer on a 2,862-ha (7,072-acre) area near Gettysburg in Adams County, Pennsylvania (Storm et al. 1992; Tzilkowski and Storm 1993). The 1992 population equates to a density of 35.5 deer/km$^2$ (92 deer/mi$^2$), which is 10 times higher than that prescribed by the Pennsylvania Game Commission for Adams County. The deer herd at Gettysburg has been associated with high levels of damage to farm crops and forest plant communities, as well as deer-vehicle collisions. An environmental impact statement is being prepared to develop a strategy for managing the Gettysburg deer population.

### Refuges

The number of deer harvested by hunters increased twofold between 1983 and 1992 at each of the three national wildlife refuges examined. During 1992, the number of deer taken by hunters was 165 (17.8/km$^2$ [46/mi$^2$]) for Eastern Neck, 210 (7.7/km$^2$ [20/mi$^2$]) for Great Swamp, and 109 (4.2/km$^2$ [11/mi$^2$]) at Montezuma. Although we did not obtain estimates of prehunt populations at these three refuges, if we assume that 35% of the population was killed, the prehunt herd size at the Great Swamp Refuge was 600 deer, which equates to 22 deer/km$^2$ (57 deer/mi$^2$).

Harvests by hunters appear to control deer at national wildlife refuges, despite the fact that each refuge manager has a unique set of cultural and biological attributes to consider in deer management. Although hunting is a viable deer management alternative for most refuges, there is still a need to monitor the size of deer herds, determine the most suitable technique to survey deer at each refuge and the most useful demographic data, and monitor plant communities to assess the effect of feeding by deer on plant resources.

Courtesy The Pennsylvania Game Commission

White-tailed deer fawn.

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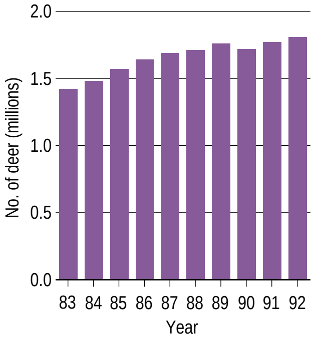


**Fig. 1.** The trend in the size of the white-tailed deer population in nine northeastern states (Connecticut, Delaware, Maine, Massachusetts, New Hampshire, New York, Pennsylvania, Rhode Island, Vermont), 1983-92.

has increased from 4.3 deer/km$^2$ (11.1 deer/mi$^2$) in 1983 to 5.5 deer/km$^2$ (14.2 deer/mi$^2$) in 1992. Density estimates ranged from 2.7 deer/km$^2$ (7.1 deer/mi$^2$) in Rhode Island to 9.7 deer/km$^2$ (25.1 deer/mi$^2$) in Pennsylvania. The total 1992 population of white-tailed deer in the Northeast (including estimates provided by personal communication with biologists from Maryland, New Jersey, Virginia, and West Virginia) was estimated at about 3.0 million.

The total antlered (Fig. 2) and antlerless harvest for all 13 states was estimated at 600,000 in 1983 and 900,000 in 1992. Managers manipulate the harvest of antlered to antlerless deer to obtain a desired population (i.e., appropriate age and sex ratios). During the past decade, deer populations in the Northeast have continued to increase except in states that harvested markedly more antlerless than antlered deer. In Pennsylvania, for example, the deer population increased until the harvest of antlerless deer reached levels necessary to curb the upward trend in the population. In contrast, Massachusetts has consistently harvested more antlered than antlerless deer and the population

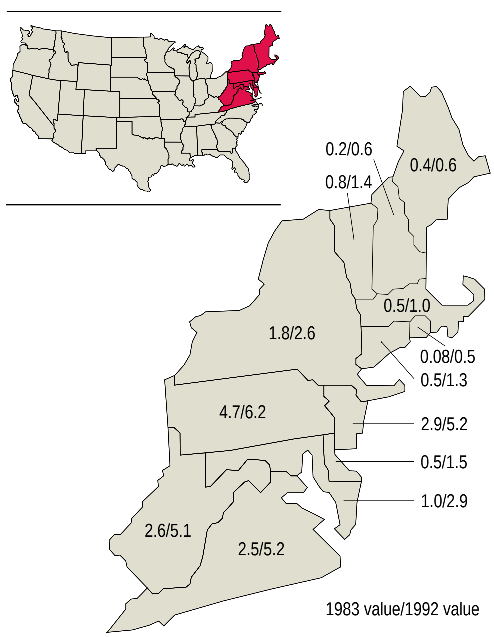


**Fig. 2.** The harvest of antlered white-tailed deer (number per square mi or 259 ha of deer range) in 13 northeastern states in 1983 (first value) and in 1992 (second value); estimates for Virginia and West Virginia include young-of-the-year males (button bucks).

continues to increase. These two examples illustrate how a prescribed harvest of antlerless deer can be used to achieve a population response that is consistent with each state's management objective. The magnitude of the antlerless and antlered deer harvest is a key factor for adjusting populations. The actual female-male ratio in the population, reproductive rates, and the sex-specific mortality caused by nonhunting factors also affect the population trends of each state.

Ten of 13 states responded to the request for


White-tailed deer (*Odocoileus virginianus*).

information on deer conflicts during the past decade; only two of these indicated no conflict between current deer populations and land use or other natural resources. Four of the eight states with conflicts indicated increasing trends in agriculture-deer conflicts. Conflicts increased between deer and urban habitats in eight states, and vehicle-deer collisions increased in seven of the states. Seven states indicated they had problems between deer and forest regeneration, and two of these states indicated the problem was becoming commoner. Seven states reported deer conflicts with parks and refuges; such problems included lack of forest regeneration as well as deer feeding on ornamental shrubs on private property. Four of these states indicated increasing trends in these kinds of problems.

## Conclusions and Present Outlook

The trends in abundance of deer in northeastern states are largely a function of regulated harvests by hunters. A significant amount of information on annual harvest by hunters and deer demographics is available in each northeastern state. Thus, the process of managing white-tailed deer may serve as a model to evaluate monitoring techniques, population dynamics, and effects of wildlife on cultural and other natural resources.

Managers of parks and refuges need better information to predict trends in regeneration and development of forests and the role of deer in forest regeneration. This will require the use of new and appropriate survey techniques (Wiggers and Beckerman 1993) and the ability to evaluate, interpret, and manage data acquired during long-term monitoring of deer and habitats used by deer (Tzilkowski and Storm 1993). Management goals can only be achieved through knowledge of trends in deer abundance and a better understanding of public attitudes toward natural resources.

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# North American Elk

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North American elk or wapiti (*Cervus elaphus*) represent how a wildlife species can recover even after heavy exploitation of populations and habitats around the turn of the century. This species is highly prized by wildlife enthusiasts and by the hunting public, which has provided the various state wildlife agencies with ample support to restore populations to previously occupied habitats and to manage populations effectively. Additionally, the Rocky Mountain Elk Foundation, founded in 1984, has promoted habitat management, acquisition, and proper hunting ethics among many segments of the hunting public.

Current population size is estimated at 782,500 animals for the entire elk range (Rocky Mountain Elk Foundation 1989). Projections of population trends for the national forests and for the entire U.S. elk range are for continued increases through the year 2040 (Flather and Hoekstra 1989).

This species occupies more suitable habitat than at any time in the century, and populations are at all-time highs (Figure). Elk populations in the United States primarily occupy federally managed lands, including national forests, public lands, national parks, and several wildlife refuges. Substantial populations occur on private holdings, including large ranches and reservations owned by Native Americans. Populations have been introduced into Michigan and Pennsylvania and recently have expanded in Nevada and California. In Canada, elk have increased their range into northern British Columbia since 1950 and occupy crown lands in Alberta, British Columbia, and Manitoba. Elk populations in the mountain parks of Jasper, Yoho, Kootenay, and Banff are an important part of the fauna, and the populations in Elk Island National Park and Riding Mountain National Park have been extensively investigated. In Alberta and the western United States, an industry centered around ranching elk has proliferated in recent years.

Perhaps the most spectacular improvement in elk populations is in California, where one population that originally consisted of about 600 individuals in the Owens Valley has now grown to over 2,500 Tule elk in 22 different populations (Phillips 1993). Aquiring habitat and reintroducing elk are the major reasons for the increase.

Problems associated with elk management include the reduced life expectancies of males, which in some areas are attributable to hunting. This problem has been aggravated by increased access to formerly inaccessible habitat, allowing more bulls to be hunted. Additionally, elk have moved into more accessible habitats that provide less cover during hunting seasons. In some cases, hunting has increased enough to lower bull elk life expectancies even in areas where access has not increased. Means to address these issues include reductions in season lengths, quotas on bulls either through hunter registration or limited-entry permit hunts, closures of extensive areas to vehicle access during the hunting season, and more integrated management of timber harvest to accommodate the needs of elk for escape cover.

Such restrictions vary in their effectiveness, depending upon numbers and distribution of hunters, other human disturbances, and the amount and kind of forest involved. In open pine forests, for example, restricting access may be less effective than in denser fir forests, making other hunting regulations, such as limited-entry hunts, necessary. Elk occupying open rangelands where conifer cover is poorly distributed are largely subject to limited-entry hunting. Elk are sensitive to human activity

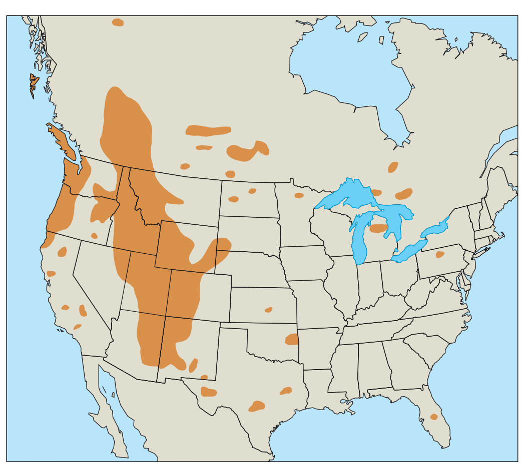


**Figure.** Distribution of elk in North America as of 1978, based on information provided by provincial and state wildlife agencies (modified from Thomas and Toweill 1982, used with permission, Wildlife Management Institute).

even in national parks where they are not hunted and may become partially conditioned to human presence. Recreational, logging, grazing, seismic, and mining activities must be restricted to times and places where animals are least affected.

As elk numbers have increased in farming areas, depredation on cash crops has also increased. Efforts to address this issue include special "depredation" hunts designed to move animals away from problem areas or to reduce populations, planting less palatable crops, fencing hay and valuable crops to prevent access by elk, feeding elk, and hazing to discourage use. An integrated and specially tailored approach is often necessary to address this important problem.

Whether the high densities of elk that occur within Yellowstone National Park are perceived to be a problem depends upon one's viewpoint. Current research on the condition of park plant communities heavily used by wintering elk suggests that factors interact to influence these communities. Grasslands that have been protected for more than 30 years did not exhibit changes in productivity when compared with grazed grasslands (Coughenour 1991). On the other hand, when protected stands are compared with stands open to browsing, it appears that woody plants may have been adversely altered through prolonged heavy grazing (Chadde and Kay 1991). Past actions that affected plants include fire protection, concentrated grazing pressure by bison (*Bison bison*) in some areas, and altered grizzly bear (*Ursus arctos*) feeding behavior. Within Yellowstone Park, the prospective restoration of wolf (*Canis lupus*) populations and changes in grizzly bear populations since the elimination of artifical food sources will undoubtedly affect elk populations that exist primarily within the park.

Natural changes in habitat across the western elk range have largely benefited elk. Efforts to improve range conditions by modifying livestock grazing practices will provide more forage for elk, even if losses in woody plants may reduce the habitat quality for deer. Better livestock management should also mean accommodating elk habitat use by providing ungrazed pastures within grazing allotments and by manipulating livestock grazing so plants retain their palatability to elk. As livestock is managed more effectively across western public lands, forage plants that wildlife use will benefit, thus also benefiting elk.

On the other hand, some traditional high-quality elk winter habitats, which contain seral (*see* glossary) shrub ranges that developed after large fires earlier this century, are now growing into conifer stands. Some conifers like Douglas fir (*Pseudotsuga menziesii*) are palatable and highly digestible for elk, and even pole-size stands can provide needed cover during severe winters or hunting seasons. As conifers dominate a larger proportion of the winter ranges and associated spring habitats, however, they shade out other species and habitat quality may deteriorate, eventually hurting elk populations. These long-term changes are not easily dealt with in short-term management efforts.

Nevertheless, the future of elk populations in North America seems secure. Demand for hunting as well as the nonconsumptive values of elk will ensure the success of substantial populations. Elk populations will benefit from improved habitat conditions on arid portions of the range, improved livestock management, more effective integrated management of forested habitats, and continued implementation of fire management policies in the major wilderness areas and national parks.

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