Population Status of Native Species

The principal objective of this report is to describe recent trends and status of California's wildlife resources. In general, the data on which this assessment is based were collected for purposes other than a statewide assessment. Therefore, the information reported originates from a variety of federal and State agencies and non-governmental cooperators. The California Department of Fish and Game (DFG) developed data on population and harvest trends for the State's large mammal game species that are a product of extensive field sampling and harvest monitoring and modeling to infer population size. Figure 1 is a guide to county and topographical feature locations mentioned in this section.

Figure 1. Topographic feature locations

Source: compiled by FRAP from Barbour and Major, 1977

Findings on land cover and habitat

This section reviews broad trends in wildlife habitat extent and condition, which give an indication of availability and quality of wildlife habitat. Scientific literature contains many definitions of what constitutes habitat for plant and animal species. Morrison et al. (1992) defines habitat as "the area with the combination of resources and environmental conditions that promotes occupancy by individuals of a given species and allows those individuals to survive and reproduce." Thus, habitat is different from land cover, which refers to land units having approximately the same capacity to produce vegetation. Patton (1992) describes habitat as simply "the environment of and the specific place where an organism lives."

This assessment considers, for the most part, the landscape features associated with the distribution and abundance of species and the habitat relationship models that seek to describe that relationship. Technological advancements in map-making, data storage, and retrieval now make it possible to begin to examine the geographic distribution and arrangement of habitat needs for a larger number of species. For example, the California Wildlife Habitat Relationships System (CWHR) is an informational database frequently used to associate wildlife occurrence with the habitats on which they depend. CWHR contains life history, management, and habitat relationship information for 675 species of amphibians, reptiles, birds and mammals known to occur in California's many habitat types.

The most basic descriptor of species diversity is species richness (the number of species occurring in an area). In general terms, the relative importance of forest and rangeland habitat types to terrestrial vertebrate species is now readily evaluated with the California Wildlife Habitat Relationships System. For California's terrestrial vertebrate species, richness is potentially highest in the shrub-dominated habitats and lowest in urban areas (Table 1).

Table 1. Species richness by land cover class*

	Number of species				
Land cover	**Amphibians**	**Reptiles**	**Birds**	**Mammals**	**Total**
Agriculture	9	12	194	61	276
Conifer Forest	32	31	177	114	354
Conifer Woodland	6	51	141	85	287
Desert Shrub	11	53	102	85	251
Desert Woodland	13	50	156	67	286
Hardwood Forest	30	26	175	102	333
Hardwood Woodland	30	45	205	98	378
Grassland	20	38	135	114	307
Shrub	27	68	186	133	414
Urban	4	8	169	43	224
Wetland	29	22	186	89	326

**Optimal (high) or suitable (medium) breeding habitat suitability ratings*
Source: California Department of Fish and Game and California Interagency Wildlife Task Group, 2001

Similarly, forest and rangeland condition is being increasingly evaluated based on measures associated with ecosystem structure and function. This perspective incorporates a broader understanding of the role of habitat availability and species occurrence as a measure of biological diversity (see Habitat Diversity, Rangeland Area and Condition).

Findings on large mammal game species

Population numbers reported and the trends derived in this section must be interpreted with caution. A variety of variables influences the accuracy and comparability of data collected over time. These include improved information concerning age structure of the population, levels of mortality, estimates of extent and quality of habitat, and other potentially significant and locally specific demographic considerations.

Black Bear

In California, black bears are found in mountainous areas and most commonly inhabit forested and chaparral dominated plant communities. Two subspecies are recognized, the northwestern black bear (*Ursus americanus altifrontalis*) and the California black bear (*Ursus americanus californiensis*).

Black bear. Photo courtesy of California Department of Water Resources.

Black bear numbers in California are now increasing. Important demographic measures such as sex ratio of harvested bears, median age, and number of bears harvested indicate increasing population levels. In addition, the illegal take of bears has been greatly reduced from levels seen prior to 1985. Current population levels are estimated between 17,000 and 23,000. This is up from an estimated population of 10,000 to 15,000 in the early 1980s.

Mixed conifer forests, montane hardwood conifer, chaparral, and hardwood are important habitat types and support the greatest bear densities. Substantial amounts of these habitats have been converted to other uses. However, most of the conversion has been associated with urban centers and/or in habitat types that are of little or no value as bear habitat (DFG, 2001a).

In the early 1980s, the illegal killing of black bear in the spring and summer was considered a problem, and in some areas approached legal harvest. Poaching, in combination with other sources of mortality, was nearing dangerous levels for the bear population. Evidence for this trend included decline in legal harvest, hunter success, median age of hunter-killed bears, and the increased success of law enforcement operations.

The adoption and enforcement of additional regulations and laws to reduce illegal bear hunting in California appear to have successfully lowered bear mortality to sustainable levels and improved bear population status.

The adoption and enforcement of additional regulations and laws to reduce illegal bear hunting in California appear to have successfully lowered bear mortality to sustainable levels and improved bear population status. The regulations include controlling the use of dogs to pursue bear in the spring and summer; making the selling of bear parts a felony; mandatory bear tag return for both successful and unsuccessful hunters; and bear tooth sampling to improve demographic data.

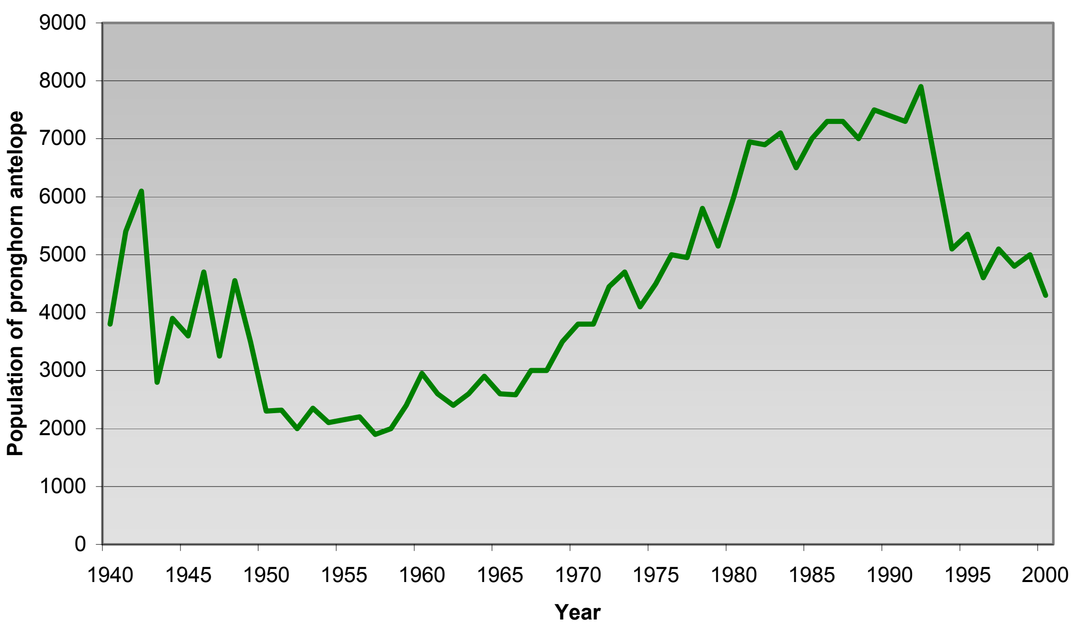
Pronghorn Antelope

Historically, the pronghorn antelope (*Antilocapra americana*) inhabited most of the grassland, oak woodland, and sagebrush-steppe plant communities in California. The antelope was likely the most abundant big game animal in the State (Pyshora, 1977) and population densities in the San Joaquin Valley may have been the highest in North America. However, by the early 1870s their numbers were significantly reduced due to market hunting, livestock competition, and changing land use practices. In 1923, it was estimated that less than 1,100 pronghorn were present in seven areas of California. By 1943, pronghorn were found only in northeastern California (DFG, 2001f).

Pronghorn antelope. Photo by Gerald and Buff Corsi. California Academy of Sciences.

Population levels have increased from the mid-1940s due to generally favorable weather conditions, increases in acreage devoted to alfalfa and grain crops, reductions in competition for forage with livestock on public lands, and species management practices (Figure 2). DFG and other cooperators are actively involved in establishing new herds in suitable habitat.

Figure 2. Pronghorn antelope population estimate, 1940-2000



Source: DFG, 2001f

Elk

Three subspecies of elk occur in California. The Roosevelt elk (*Cervus elaphus roosevelti*) inhabits coastal areas in Mendocino, Humboldt, and Del Norte counties in addition to the Cascade and Klamath Mountain Ranges in Siskiyou County. The introduced Rocky Mountain elk (*Cervus elaphus nelsoni*) is found in the Warner Mountains of Modoc County, and southern Kern, western San Luis Obispo, and Shasta counties. The tule elk (*Cervus elaphus nannodes*) occurs in a number of individual herd areas in the coast range, valley floor, and Owens Valley. Collectively, numbers of elk have exhibited increasing trends since the mid 1960s. The greatest increases have occurred within the tule and Roosevelt elk subspecies due primarily to the establishment of new herds by DFG and other cooperators.

Elk. Photo by Dr. Lloyd Glenn Ingles, California Academy of Sciences.

Roosevelt elk were once widely distributed throughout northern California. However, by 1925, they were reduced to a small area of Humboldt and Del Norte counties. Elimination of market hunting and public ownership of large tracts of habitat contributed to significant population increases. Relocation efforts by DFG (280 elk since 1985) and natural movement of elk from Oregon into California have resulted in range expansion. Elk now occupy new areas in Mendocino County and the Klamath and Cascade Mountain Ranges of Siskiyou and Trinity counties with significant population increases. The DFG currently estimates Roosevelt elk numbers at approximately 4,000 (DFG, 2001d).

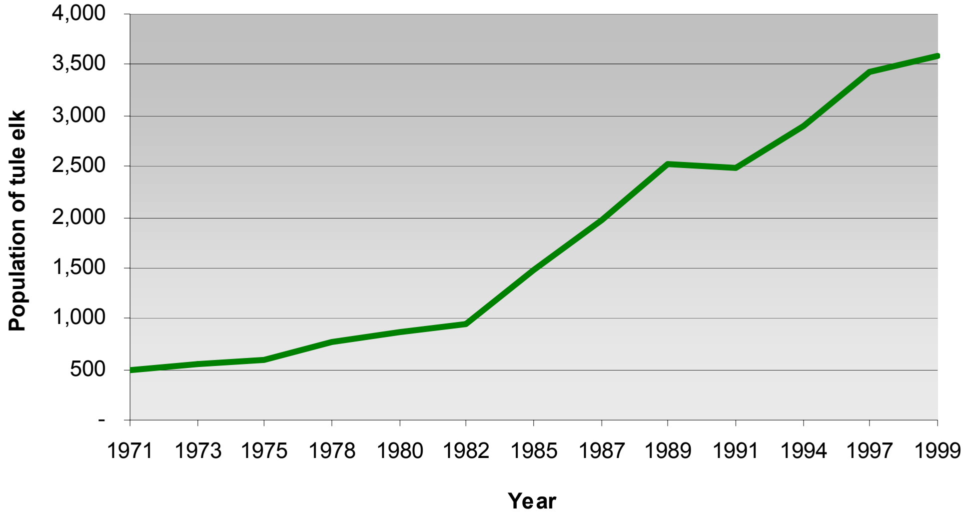
Of the four populations of Rocky Mountain elk in California, only the population in the Warner Mountains appears to have originated by animals moving into suitable habitat from southeastern Oregon. DFG estimates a total population of 1,000-1,500 including 300-400 in two populations in the southern portion of the State (DFG, 2001d).

Tule elk were undoubtedly the most numerous elk subspecies in California. Historical estimates of early explorers suggest as many as 500,000 elk inhabiting the oak woodland, savannah, and valley floor. However, by the 1860s the effects of habitat conversion to agriculture, market hunting, and competition with livestock reduced numbers and distribution to a small herd in the southern San Joaquin Valley (McCullough, 1969).

Complete protection of the elk remaining on private lands and subsequent relocation efforts by the California Academy of Sciences contributed to an increase in numbers. This resulted in established herds in three locations by 1940: Cache Creek herd in Colusa and Lake counties; the Owens Valley herd in Inyo county; and the 953 acre enclosure at the Tupman Tule Elk Reserve in Kern county.

Legislation passed in 1971 (the Behr Bill) and amendments to the California Fish and Game Code, established a tule elk carrying capacity of 490 head in Owens Valley. Furthermore, the new legislation restricted the ability of the Fish and Game Commission to regulate elk numbers through hunting. Passage of Public Law 94-389 in 1976 required the Secretaries of Agriculture, Interior, and Defense to cooperate with the State to identify suitable relocation sites. In addition, a population of 2,000 tule elk was recognized as an appropriate national goal, which resulted in renewed efforts at the identification and relocation of elk to new areas within their historic range. Tule elk are now established in 22 herd areas (as of 2000) with a total population of approximately 3,580 (1999 estimate); up from only 500 in 1971 (Figure 3).

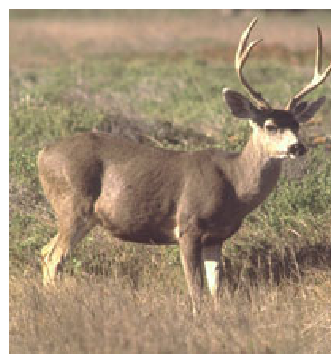
Figure 3. Tule elk population estimate, 1971-1999



Source: DFG, 2001d

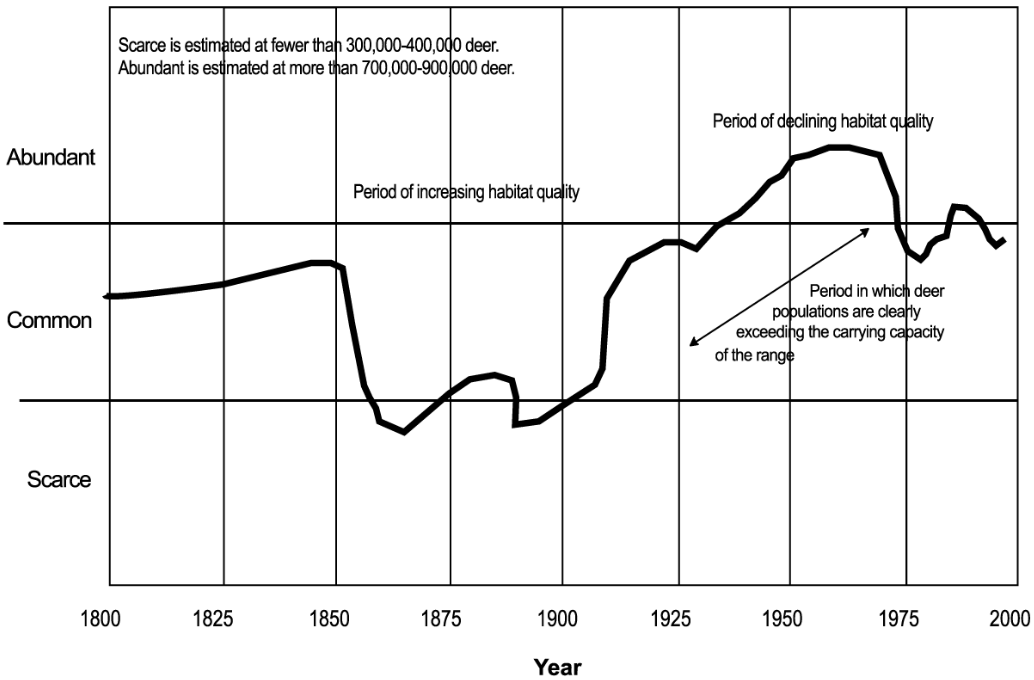
Deer

California's deer (*Odocoileus hemionus*) population peaked during the late 1950s and early 1960s (Figure 4). The high deer population levels during that period are the product of large-scale land use and management policy changes that influenced forage quality and direct mortality in the early to mid-1900s. These include the elimination of unrestricted hunting; reduction in predator populations as a result of unregulated trapping and hunting; significant reduction in numbers of domestic livestock grazing on public lands; and the spread of timber harvest and subsequent use of fire as elements in the establishment of shrub fields and other early successional habitats.



Mule deer. Photo by Dr. Lloyd Glenn Ingles, California Academy of Sciences.

Figure 4. Deer populations as a function of habitat quality from 1800-2000



Source: DFG, 2001c

Since the mid-1970s, the total deer population in California has remained relatively stable. However, on a local herd or Deer Assessment Unit (several deer herds showing similar management needs and herd conditions) basis, marked declines in deer numbers and habitat quality and availability are evident.

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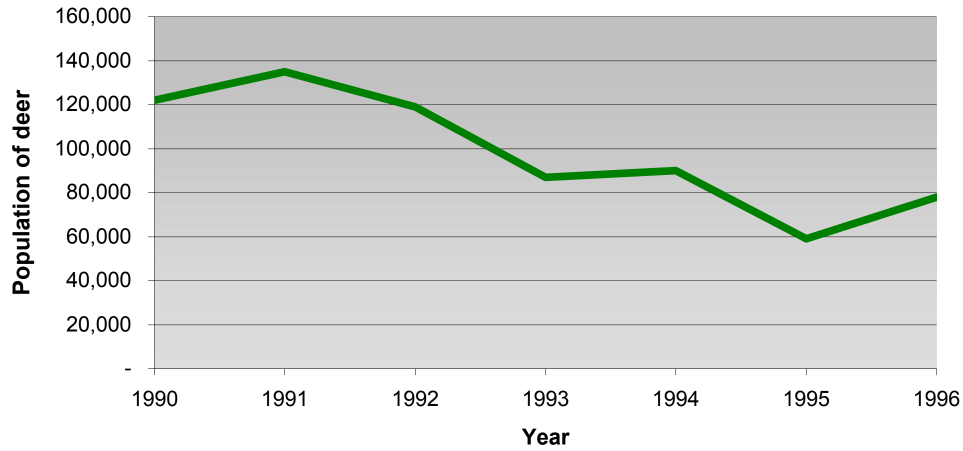
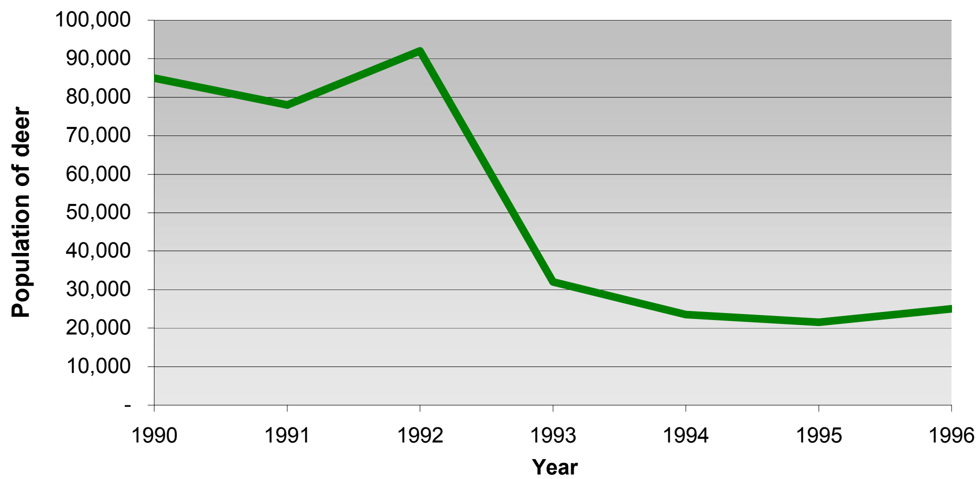
In recent years, deer populations have shown the most marked declines in northeastern California and the northern and central Sierra Nevada Mountains (Figures 5-10). Several factors are responsible for these declines including habitat loss in quality and quantity, predation, competition with livestock, urban and agricultural development, and illegal hunting. In general, the principal factor influencing deer populations is the availability of quality forage. Habitat quantity and quality continues to decline in the wake of urbanization and other agricultural development in deer habitats (DFG, 2001c).

Figure 5. Deer population estimates in the central Sierra, 1990-1996



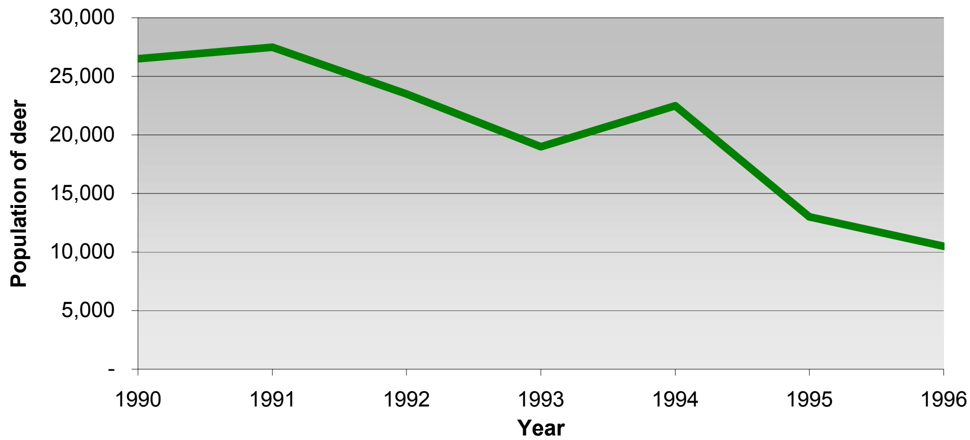
Source: DFG, 2001c

Figure 6. Deer population estimates in northeast California, 1990-1996



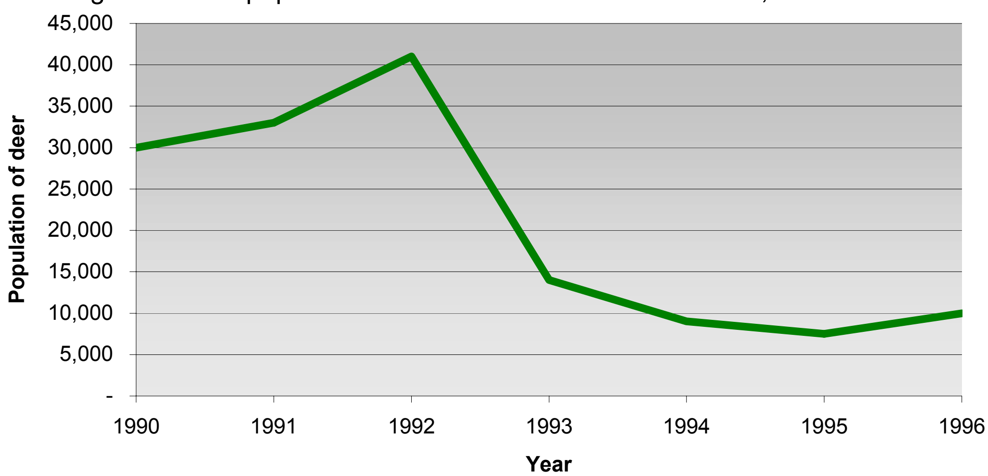
Source: DFG, 2001c

Figure 7. Deer population estimates in the east Sierra, 1990-1996



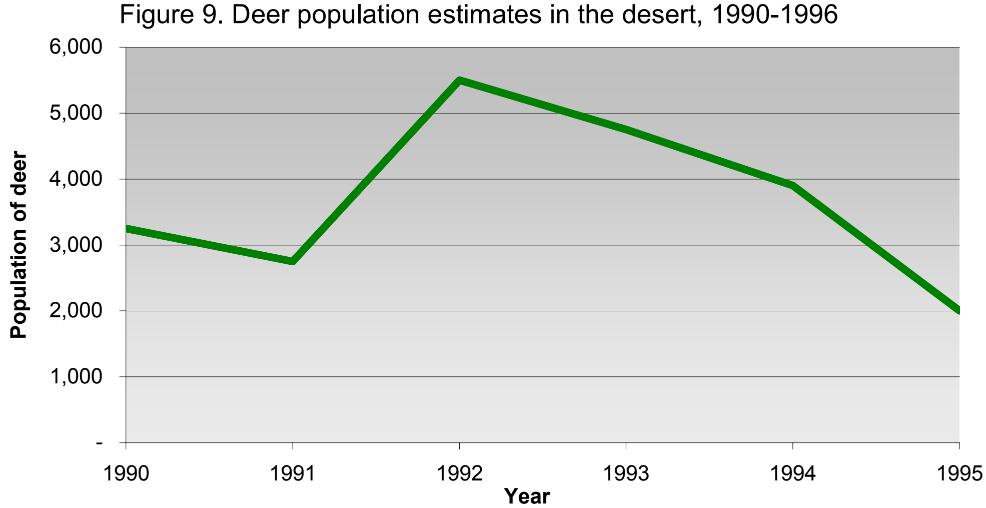
Source: DFG, 2001c

Figure 8. Deer population estimates in the northeast Sierra, 1990-1996



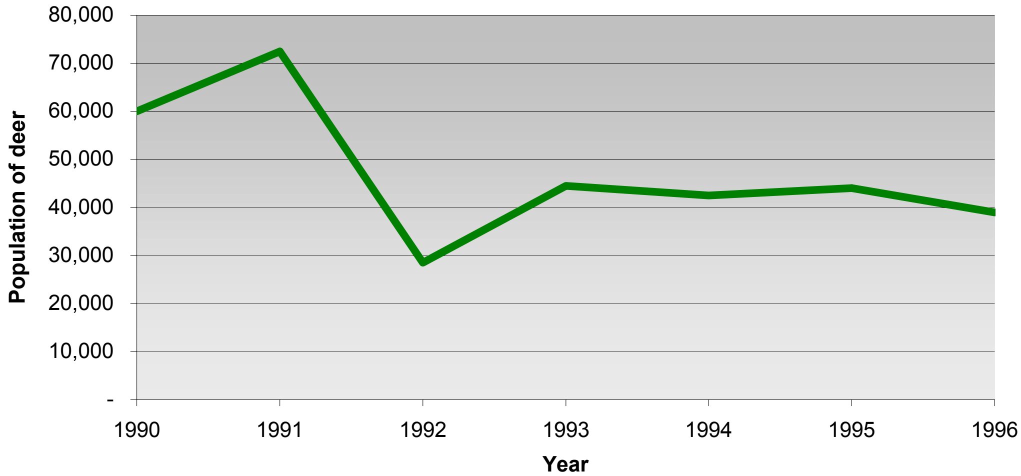
Source DFG, 2001c

Figure 9. Deer population estimates in the desert, 1990-1996



Source: DFG, 2001c

Figure 10. Deer population estimates in the Cascade/north Sierra, 1990-1996



Source: DFG, 2001c

Periodic disturbance of closed canopy shrub or second-growth tree habitats that improves forage and cover value can improve deer habitat. However, a variety of land management activities can negatively affect deer and their habitat (DFG, 2001c):

- Fire is beneficial to deer in most California habitat types with notable exceptions in the bitterbrush and sagebrush communities of the eastern Sierra and Great Basin during summer months. The lack of a sufficiently large-scale prescribed fire program to regenerate deer habitats will likely limit future deer populations.
- Prescribed fire timing and fire intensity that is designed to primarily meet fuel management objectives may not be conducive to shrub regeneration.
- Large and intense summer fires in the Great Basin and eastern Sierra Nevada Mountains deer winter range and transition ranges reduces the availability of desirable forage species and expands the range of undesirable non-native annual grasses.
- Necessary fire suppression in the private/urban and public/rural interface results in reduced vegetation diversity and habitat quality.
- Forest thinning can reduce thermal cover and when followed with herbicide treatments minimize the availability of forage.
- The general decline in total acreage and quality of deer habitat has likely increased the level of competition with livestock for available forage in preferred deer habitats on summer and winter ranges.
- Over the next 20 years, habitat conversion to agricultural and urban uses on private lands in central Sierra Nevada Mountains is expected to reduce deer populations by 10-20 percent. Public lands in these areas are becoming increasingly important to meet deer herd requirements.

Deer population levels can be improved through management of habitat quality and availability. Additionally, populations can be improved by reducing deer numbers to achieve a more appropriate “balance” with desired or current habitat conditions. Effective management strategies will use both of these approaches where possible.

Wildland fire effects on winter range condition—Great Basin interstate deer herds: Shrub-steppe plant communities in the Great Basin of California and other states have been affected by two events over the past century that has markedly altered composition of the vegetation and fire frequency. First, unrestricted grazing practices in the late 1800s resulted in reduction in occurrence or loss of perennial bunchgrass species and increase in sagebrush and bitterbrush (Salwasser, 1979; Neal, 1981; Gruell, 1986). Second, alien annual species were introduced and rapidly expanded their range and dominance on the degraded rangelands (Young et al., 1972). Cheatgrass (*Bromus tectorum*) may be the most widespread non-native annual grass species in the Great Basin. This species and other introduced annuals begin their cycle of growth early in the year and consequently mature and cure in early summer. In this condition, the introduced annuals provide an easily ignited fuel with a rapid rate of fire spread that under the soil moisture regime and fire conditions of late summer, essentially remove native shrub species used by deer. Cheatgrass and medusahead (*Taeniatherum asperum*) then invade the fire-disturbed area, inhibit the reestablishment of native shrubs, and perpetuate the introduced annual-wildlife cycle.

Health and condition of deer winter range is a key parameter that dictates individual survivability as well as recruitment of young the following spring. Deer herd activities during the winter period are geared toward maintaining a balance between the expenditure of energy and that available in the forage base. Because winter range forage condition is frequently the key factor for maintenance of deer condition, the duration and severity of the winter period plays an important role in determining the length of time deer survive and their spring condition (Moen, 1973; Wallmo et al., 1977). It follows that with decreases in the quality or availability of winter forage and thermal cover, the length of time that deer can energetically maintain themselves will also be reduced (Hobbs, 1989; Updike et al., 1990).

Bitterbush (*Purshia* spp.) has been documented as the most important winter range forage plant from September through December for mule deer in Lassen and Washoe counties of California and Nevada (Lassen et al., 1952; Dasmann and Blaisdell, 1954; Leach, 1956). Sagebush (*Artemisia* spp.) provides the primary forage source for the remainder of the winter period (Leach, 1956). Fire occurrence in this portion of Lassen and Washoe counties is generally due to lightning strikes during summer thunderstorms. Summer wildfire on the winter range results in loss of big sagebush (*A. tridentata*) and bitterbush. Regeneration of bitterbush is poor to non-existent when burned under these conditions and in this portion of the plants distribution (Nord, 1965; Rice, 1983; Martin and Driver, 1983). An additional effect of the fire related disturbance is the proliferation of cheatgrass that competes with the establishment of bitterbush seedlings arising from dormant seed (Young et al., 1972).

Updike and others (1990) suggest that summer fires in this portion of the Lassen-Washoe range is detrimental to the energy balance of wintering deer and their survival because of the extensive conversion of shrubland to annual non-native grasses and the associated loss of browse species and thermal cover. In their study, fire records for incidents greater than 300 acres were reviewed from 1957 to 1988 for southern Lassen and Washoe counties. Fires burned about 49,000 acres within recognized winter range boundaries between 1957 and 1982 (20,000 acres per year). From 1983 through 1988, 159,000 acres burned (32,000 per year).

These fires in combination with competition for forage with livestock have had a marked influence on the diet of wintering deer as well as overall population (Updike et al., 1990). Mule deer food habits were compared between December of 1951 and 1952 (Leach, 1956) and December 1987 (Leach, 1988) on adjacent winter range sites in Washoe County, Nevada. Diets of mule deer collected in December of 1951 and 1952 were 28 percent grass, 63 percent browse, and 9 percent forbs. These percentages differed markedly from those collected in December 1987. In the latter sample, 79 percent was dry cheatgrass and about 20 percent browse. No bitterbrush was found in the diet. Body fat indices of 31 deer collected in 1987 on the winter range indicated that 28 were in poor to fair condition (Shor, 1988). Deer condition was attributed to a negative energy balance due to consumption of dry cheatgrass of low digestibility and low crude protein.

In 1964, the December deer population in the winter range area reached a high of approximately 15,500 animals and has declined since. Between 1983 and 1988, the December population estimate declined by approximately 28 percent to a low of 7,100 deer (Updike et al., 1990). The severe winter conditions and heavy snows of 1992-1993 further reduced herd numbers to approximately half of the 1988 estimate (3600 deer) (Hall, 2001). Large areas of winter range, because of summer wildlife, are now relatively homogeneous stands of herbaceous vegetation lacking in adequate thermal and hiding cover. Lack of rainfall in conjunction with mild temperatures during late fall and early winter limits herbaceous forage to the dry annuals from the previous spring. Similarly, snow depth or below average temperatures on the winter range can influence herbaceous vegetation availability and energetic demands. In either case, annual herbaceous forage is unreliable as a food source in some years (Updike et al., 1990) and contains crude protein levels below that required for body maintenance (Welch, et al., 1986). The result is a periodic but significant decline in deer herd numbers with the greatest influence on young age classes and recruitment to the adult deer population.

Bighorn sheep

Three subspecies of bighorn sheep occur in California: Nelson's (*Ovis canadensis nelsoni*) from the Transverse Ranges, Mojave Desert, and eastern Sonoran Desert; Peninsular (*Ovis canadensiscremnobates*) found in the western Sonoran Desert, western Imperial, central Riverside, and eastern San Diego counties; and Sierra Nevada (*Ovis canadensis californica*) from the eastern Sierra Nevada Mountains and in recent history from northeastern California. The Peninsular and Sierra Nevada subspecies are currently state-listed as threatened and endangered under the federal Endangered Species Act.

California bighorn sheep. Photo by Gerald & Buff Corsi, California Academy of Sciences.

It is estimated that 10,000 bighorn, distributed across approximately 100 populations, were present in California in 1800 (DFG, 2001b). However, in the decades following gold discovery, unregulated market and subsistence hunting, and grazing and associated disease transmission from domestic livestock resulted in the loss of several populations in the Sierra Nevada Mountains. Recent reintroduction efforts in the Lava Beds and Warner Mountains of Modoc County have been unsuccessful due in large part to respiratory diseases contracted from domestic sheep.

Population estimates for bighorn sheep in California were infrequent until passage of Senate Resolution 43 and funding of an intensive survey effort from 1968-1972. That survey estimated the bighorn sheep population at 3,700. Since 1979, DFG has reestablished 11 new populations and augmented four small populations through translocation projects.

As of 1996, approximately 3,200 bighorn sheep occupied several Mojave and Sonoran Colorado desert mountain ranges in the southeastern portion of the State. They are also found in five populations of 160 animals within the eastern Sierra Nevada Mountains and three populations of about 300 individuals in the Transverse Ranges of Ventura, Los Angeles, and San Bernardino counties (DFG, 2001b). Individual population management plans are being developed to identify and protect important habitats, identify future reintroduction sites and limiting factors, and collect demographic data.

Since the late 1970s, mountain lion have preyed more heavily on bighorn sheep. From 1977 to 1987, 50 sheep from the Mount Baxter herd in the Sierra Nevada Mountains were lost to mountain lion predation on low elevation winter range. In response to predation pressure and drought conditions of the late 1980s, bighorn sheep are increasingly using higher elevation ranges and low quality forage. In addition to this behavioral change, there has been a consistent decline in Sierra Nevada bighorn sheep numbers.

In 1996, the Sierra Nevada bighorn sheep population fell to 150 individuals from the 250 recorded in 1979 (Graber, 1996). Compounding the problem is the lack of suitable reintroduction sites in the eastern Sierra Nevada Mountains given domestic sheep and cattle allotments on public lands and potential for disease transmission.

Mountain lion

The mountain lion (*Puma concolor*) is widespread in California and can be found from sea level to alpine meadows, with the general exception of dry areas of the Colorado and Mojave Deserts that do not support mule deer populations and the agricultural areas of the central valley. Approximately 62 percent of California is within the known range of mountain lions. Of that total, approximately 62 percent is considered moderate to highly suitable habitat (Torres et al., 1996). The species is rarely observed in the wild, given their habitat requirements, large home range size, relatively low population densities, and secretive nature.

Mountain lion. Photo by Gerald & Buff Corsi, California Academy of Sciences.

Although the mountain lion is not currently designated a game animal in California, it has held a variety of management classifications. These range from bountied predator in 1907 to non-game and non-protected animal in 1963 to game mammal in 1969 and through passage of Proposition 117 to "specially protected mammal" in 1990. Only one complete regulated hunting season (1970-1971) as part of a management program has occurred for this species (Torres et al., 1996).

Estimating the statewide mountain lion population and trends is a difficult task. Regional variation in population densities, habitats used, and effectiveness of population monitoring techniques complicate the estimations. Sitton and Wallen (1976) conducted the first major mountain lion study in California and found no evidence to suggest a change from the 1973 DFG estimate of 2,400 mountain lions. However, in the late 1980s, the DFG population estimate was revised to approximately 5,100 with a likely range of 4,000 to 6,000.

These data would suggest that mountain lion numbers have increased over the last 30 years. Coincidentally, there has been an increase in conflicts with California's growing population in rural and largely undeveloped areas.

Verified occurrences of mountain lion depredation increased from 21 in 1973 to 331 in 1995. While no verified attacks on humans occurred in California during the period of 1910-1985, nine were recorded in the 1986-1996 period.

Mountain lion depredations, interactions with people and predation events have decreased in many regions of the State, suggesting regional declines in populations from the mid-1990s.

These trends in conflict with rural or urban interface residents and bighorn sheep and other species of concern (Torres, 2000), are regional scale phenomena. They may be representative of growing mountain lion populations, change in habitat conditions, and/or movement of people into suitable mountain lion habitats and do not necessarily suggest a trend in lion populations identifiable at a statewide scale (Torres et al., 1996). More recently, mountain lion depredations, interactions with people, and predation events have decreased in many regions of the State, suggesting regional declines in populations from the mid-1990s (Torres, 2000).

Findings on fur bearing and non-game mammals

Information on trends in furbearer and non-game mammal populations is limited in California. Thirteen species are harvested for their pelts and the regulations pertaining to their take are similar. Although all 13 species are biologically furbearers, the Fish and Game Code identifies six species as furbearers and seven as non-game mammals. Yearly harvest data comes from annual reports from fur buyers, trapper catch reports by county, bobcat tagging requirements, and take by federal animal damage control efforts.

Population density estimates from a variety of habitats provide an indication of animal numbers. Variation in the market price of pelts has historically had a major effect on the number of trappers. The market price also affects the number of animals of a particular species taken. Similarly, market prices for pelts can be independent of availability of animals in the wild.

A recent compilation of furbearer population status was completed for the Fur Resources Committee of the International Association of Fish and Wildlife Agencies (Flather et al., 1999). Population status for the three most significant species, as determined by state biologists, was evaluated based on population relative to estimated habitat carrying capacity.

Population status for the muskrat, coyote, and bobcat is considered to be at the maximum capacity of the habitat. Of those three, only bobcats show an expected decreasing population over the next ten years (Flather et al., 1999). Analysis of biological data and population estimates for each species concluded that furbearer and non-game mammal hunting and trapping would not have a significant impact on the California population (DFG, 2001e).

The conservation of forest carnivores is also a topic of some concern among wildlife managers. Populations of marten and fisher within California are near the southernmost limits of the species range and likely occupy marginally suitable habitat. In addition, these areas are where human disturbance on habitat values is the most significant (Lyon et al., 1994). Current populations of marten and fisher may be particularly vulnerable to local extirpation resulting from random demographic or environmental events. They are particularly susceptible to these events given their relatively low ability to colonize new areas of suitable habitat (Lyon et al., 1994).

In California, two populations of fisher are known and occur in northwestern California and the southern Sierra Nevada Mountains and are considered isolated from one another and from populations in other parts of the species distribution. The status of the Humboldt marten (*Martes americana humboldtensis*) in northwestern California is uncertain (Lyon et al., 1994).

The wolverine (*Gulo gulo*), currently State listed as threatened, was likely never numerous in California relative to densities found in other parts of the species range. No specimen has been found and no photographs taken for more than 50 years. The number of reported sightings has declined in recent years and survey efforts in the early 1990s in areas of suitable habitat of the Sierra Nevada Mountains were not successful in documenting presence (BioSystems Analysis, 1994).

Findings on resident game birds

Resident game birds are native as well as introduced species and include chukar partridge (*Alectoris chukar*), ring-necked pheasant (*Phasianus colchicus*), blue grouse (*Dendragapus obscurus*), white-tailed ptarmigan (*Lagopus leucurus*), ruffed grouse (*Bonasa umbellus*), sage grouse (*Centrocercus urophasianus*), wild turkey (*Meleagris gallopavo*), Gambel's quail (*Callipela gambelii*), California quail (*Callipella californica*), and mountain quail (*Oreortyx pictus*).


California quail. Photo by H. Vannoy Davis, California Academy of Sciences

Population status of resident game bird species varies with habitat extent and condition. Habitat condition is primarily determined by amounts and timing of annual precipitation and effect on grasses, forbs, and insect populations. Annual surveys to assess population status are conducted on established routes in selected locales by DFG volunteers. Methodologies vary depending on the game bird species, but include water source counts, roadside transects and brood counts, and crowing counts.

Annual hunter kill data and hunter surveys are used as an additional means to assess abundance and to help guide season and bag limit determinations. However, long-term trends in resident game bird hunter participation and associated harvest have exhibited a steady decline since peaking in the 1960s. These trends suggest that both the number of hunters and their harvest will continue to decline or remain stable. For example, the number of quail hunters has declined from a high of 230,000 in 1967 to 85,500 in 1996.

The number of quail harvested has paced hunter numbers with 2.75 million quail bagged in 1964 but only 800,000 in 1996. Wild turkey harvest represents a notable difference in this trend with a gradual increase in hunter participation since 1967. In 1996, 22,000 hunters harvested approximately 17,000 wild turkeys (DFG, 2001g).

In general, population numbers of these species cannot be precisely determined. Wide variation in numbers in different parts of the species range, level of inventory or census effort, and occupancy of a variety of habitat types in varying degrees of condition make expression of population levels in anything other than a range of expected numbers impossible. North American Breeding Bird Survey (see description below) results for the period of 1966 through 1999 (Table 2) give a general indication of population trend over time.

Wild turkey and California quail exhibit significant positive trends in population over the 1966-1998 period. The remaining species exhibit upward trends though at non-significant levels.

Wild turkey and California quail exhibit significant positive trends in population over the 1966-1998 period. The remaining species exhibit upward trends though at non-significant levels. Adult spring populations for white-tailed ptarmigan and ruffed grouse are estimated at 3,250 and 7,900, respectively. Based on area counts, adult spring populations of sage grouse are 2,770 birds (DFG, 2001g).

Ruffed grouse are at the southernmost limit of their distribution in northwestern California. Recently, sage grouse populations have fluctuated widely due to habitat alteration. As such, both species are considered species of special concern, a classification that has no statutory or regulatory authority but highlights consideration of the species during planning efforts.

Table 2. Resident game bird population trends for California 1966-2000

Species	Trend*	P-Value**	Number of survey routes with species detected
Chukar	5.95	0.17	8
Ring-Necked Pheasant	-0.27	0.87	59
Blue Grouse	7.33	0.43	15
Sage Grouse	1.99	0.80	3
Mountain Quail	0.31	0.34	108
California Quail	1.12	0.07	164
Gambel's Quail	-0.20	0.96	24
Wild Turkey	31.72	0.00	23
Mourning Dove***	-1.13	0.00	214

** Annual percentage increase or decrease*

*** P-Value > 0.1 is a non-significant trend, positive or negative*

**** Considered a migratory game bird but common in a variety of habitats throughout the State*

Note: Only ring-necked pheasant, California quail, and mourning dove represent species where sufficient sample size, abundance on survey routes, and precision of results provide for a high level of confidence in results.

Source: Sauer et al., 2000

Findings on non-game wildlife

Non-game wildlife is, for the purposes of this assessment, those vertebrate species that are not taken for sport, subsistence, or profit (Flather et al., 1999). Non-game wildlife, as treated here, also includes imperiled species of one categorization or another (see Species of Concern). Few data sources, with the exception of those for game species, are available that provide an assessment of statewide or bioregional population trends. Bioregional assessments for a particular class of vertebrate (fish, amphibians, etc.) have been completed but generally report on the presence or absence of the species. These vertebrate assessments cover a significant span of time with little data collection in intervening years.

Birds represent one taxonomic group where data is available to examine broad scale abundance trends over long periods due to the efforts of the North American Breeding Bird Survey (BBS). The North American Amphibian Monitoring Program, although of much more recent origin, is similar in concept to the BBS and also uses a group of volunteers who complete calling surveys along established routes to determine relative abundance of amphibians.

Birds

The North American Breeding Bird Survey was established in 1966 to provide breeding bird data in the United States and southern Canada. Bird counting stops are established along secondary roads at 0.5-mile intervals for a distance of 24.5 miles and are visited annually. Because of these systematically collected data, breeding landbirds can provide a useful indicator of the status and health of those ecosystems sampled.

Breeding land birds can provide a useful indicator of the status and health of those ecosystems sampled.

Relative abundance trends for California were summarized in two ways (Flather et al., 1999). The numbers of species with statistically significant increasing, decreasing, or stable trends for the State are estimated. Second, species are grouped according to life history characteristics including: nest type and location (cavity, open cup, ground/low, midstory/canopy), migration status (neotropical migrant, short distance, and permanent resident), and breeding habitat (woodland, shrubland, grassland, wetland, and urban) (Peterjohn and Sauer, 1993).

The number of species with increasing, decreasing, or stable trends was also estimated for each life history characteristic. In order to maintain statistical significance, each bird species must have been detected on at least 15 survey routes. For most species, sampling is insufficient to determine population trends; nonetheless, the systematically collected data do provide a useful indicator for those species that are adequately represented on survey routes and at the scale of the State.

Statewide, the majority of all surveyed species for the period 1966-1999 (117 species, 66 percent of all species analyzed) were stable in relative abundance. The number of species with increasing (29 species, 16 percent) and decreasing trends (31 species, 18 percent) were similar.

The percentage of bird species considered stable decreased between the 1969-1979 and 1980-1999 periods.

Trends in relative abundance of all birds analyzed may also be compared for the periods 1966-1979 and 1980-1999. A markedly higher percentage of all bird species (83 percent) are considered stable in relative population abundance during 1966-1979 than the most recent period of 1980-1999 (67 percent). Conversely, a greater percentage of all bird species are considered increasing (13 percent versus 8 percent) or decreasing (20 percent versus 9 percent) in relative abundance during the 1980-1999 period.

Of the 12 life history groups examined, those with the greatest proportion of declining species for the 1966-1999 period are found in the urban (33 percent) and mid-story/canopy (30 percent) groups.

In general, the percentage of bird species considered stable decreased for each of the 12 life history groups represented when the 1969-1979 and 1980-1999 survey periods are compared (Table 3). Likewise, the percentage of bird species showing declining trends increased in the most recent period with the exception of grassland inhabiting species. When sampling periods are compared (1969-1979 versus 1980-1999), the greatest increase in declining species also occurred in the urban (37 percent) and mid-story/canopy (25 percent) groups. These urban and mid-story/canopy groups are followed closely by cavity nesting (22 percent) and woodland inhabiting (21 percent) species. The greatest percentage increase for increasing species occurred in wetland (23 percent) and grassland (13 percent) life history groups.

Table 3. Number and percentage of bird species with increasing, decreasing, and stable population trends by life history group, 1966-1999

Life history group	Increasing species			Decreasing species			Stable species			Total species		
	1966-1999	1966-1979	1980-1999	1966-1999	1966-1979	1980-1999	1966-1999	1966-1979	1980-1999	1966-1999	1966-1979	1980-1999
Cavity	5 (17%)	2 (10%)	2 (8%)	7 (24%)	1 (5%)	7 (27%)	17 (59%)	18 (85%)	17 (65%)	29	21	26
Open Cup	10 (14%)	2 (3%)	3 (4%)	17 (23%)	9 (14%)	6 (23%)	47 (64%)	52 (83%)	52 (73%)	74	63	71
Ground/low	7 (15%)	1 (3%)	4 (9%)	8 (17%)	3 (8%)	5 (11%)	32 (68%)	34 (89%)	37 (80%)	47	38	46
Midstory/Canopy	7 (12%)	4 (8%)	3 (5%)	18 (30%)	7 (13%)	21 (38%)	35 (58%)	41 (79%)	32 (57%)	60	52	56
Neotropical	7 (14%)	3 (8%)	2 (4%)	11 (22%)	5 (13%)	11 (23%)	32 (64%)	31 (79%)	35 (73%)	50	39	48
Short distance	6 (11%)	2 (5%)	7 (14%)	14 (25%)	5 (12%)	12 (24%)	35 (64%)	36 (83%)	32 (62%)	55	43	51
Permanent resident	8 (17%)	6 (15%)	7 (16%)	4 (9%)	2 (5%)	9 (20%)	34 (74%)	31 (80%)	28 (64%)	46	39	44
Woodland	6 (13%)	1 (3%)	4 (9%)	10 (21%)	3 (8%)	13 (29%)	31 (66%)	32 (89%)	28 (62%)	47	36	45
Shrubland	6 (13%)	3 (8%)	3 (7%)	8 (18%)	4 (11%)	8 (18%)	31 (69%)	30 (81%)	33 (75%)	45	37	44
Grassland	2 (22%)	0 (0%)	1 (13%)	1 (11%)	1 (14%)	1 (13%)	6 (67%)	6 (86%)	6 (74%)	9	7	8
Wetland	9 (29%)	0 (0%)	7 (23%)	2 (6%)	0 (0%)	2 (7%)	20 (65%)	12 (100%)	21 (70%)	31	12	30
Urban	1 (11%)	1 (13%)	1 (13%)	3 (33%)	1 (13%)	4 (50%)	5 (56%)	6 (74%)	3 (37%)	9	8	8
All birds	29 (16%)	11 (8%)	22 (13%)	31 (18%)	12 (9%)	34 (20%)	117 (66%)	107 (83%)	112 (67%)	177	130	168

Source: U.S. Geological Survey, 2001

Bird species within the cavity, open cup nesting, and neotropical migrant life history groups are frequently the object of conservation and management initiatives. Managers are concerned for these species given loss of snags, nest parasitism by other bird species, and tropical deforestation and habitat loss respectively. Seventy-nine percent of neotropical migrant species exhibited stable populations during the 1966-1979 period. This declined to 73 percent for the 1989-1999 period.

The percentage of open cup nesting bird species populations considered stable also decreased from the 1966-1979 (83 percent) and 1980-1999 (73 percent) periods. Similarly, the percentage of populations categorized as decreasing, increased from 14 percent for the 1966-1979 period to 23 percent for the 1980-1999 period. Cavity nesting species regarded as stable decreased from 85 percent to 65 percent over the two time periods. The percentage of species showing decreasing population trends increased from 5 percent during the 1966-1979 period to 27 percent during the 1980-1999 period. Overall, the percentage of species exhibiting decreasing trends was larger during 1980-1999 (23 percent) versus the 1966-1979 (13 percent) period.

The North American Breeding Bird Survey (BBS) is a key source of information regarding population trends for the majority of North American bird species. However, many features of the survey complicate analyses that seek to go beyond general trends and identify cause and affect relationships influencing bird populations (Sauer et al., 1996).

A variety of environmental factors affects bird populations. Weather conditions, competition with other species, predation, and habitat condition, working either independently or cumulatively affect bird numbers. Similarly, within survey route observer effects, where there is a change in observer or a change in regional survey route coverage, can confound the determination of cause and effect relationships (Temple and Wiens, 1989; Barker and Sauer, 1992).

Although BBS data have been collected in a standardized manner since initiation of the survey, methods of data analysis have changed over time. Several statistical methods have been used to estimate population trends but there is no consensus on which method is most reliable and additional research in this area is required. Prioritization of species conservation efforts based on the statistical significance of trends may vary depending on the data analysis method selected (Thomas and Martin, 1996). Statistical analyses of data and subsequent interpretation are best focused on gross pattern of population change instead of magnitude of calculated trends and variances (Droege, 1990).

Amphibians and reptiles

Over the last decade, there has been a general decline in many amphibian species in California, North America, and other parts of the world. In some cases, the cause of the decline is proportional to loss of habitat.

Recolonization of areas formerly occupied by some Sierra Nevada frog species is unlikely due to the widespread loss of populations and the presence of introduced predators.

For other amphibians, the reasons are much less clear. This is particularly true when the decline is noted in undisturbed areas. A variety of factors have been suggested by way of explanation and include ionizing radiation from a depleted ozone layer, estrogenic effects of pesticides as an influence on reproduction, acid precipitation, application of fertilizers and herbicides, introduction of exotic competitors and predators, and infectious diseases (Declining Amphibian Populations Task Force, 2002). Unlike the variety of disturbances that influence aquatic amphibian and reptile (western pond turtle) species, reptile populations are most influenced by habitat conversions (Veirs and Opler, 1998).

True frog and toad species have exhibited the most significant declines. Conservation practices that were previously thought effective, such as setting aside lands from development or reliance on parks or other reserved lands, may not provide the desired results in the face of ecosystem-wide or trans-regional effects. Forty percent of the toad species (four of ten) and 88 percent of the native frog taxa (seven of eight) have been removed from at least 45 percent of their historic California distribution (Jennings, 1995; Veirs and Opler, 1998).

Little comparative baseline data is available to address long-term amphibian population trends in the western United States and California. The documentation of an entire frog fauna declining in a large, diverse region is unprecedented. However, in 1996 Drost and Fellers re-surveyed a Sierra Nevada Mountains transect first conducted by Grinnell and Storer (1924). They included Yosemite National Park in this new survey and showed marked declines in the amphibian fauna. Their re-survey indicated that at least five of the seven frog and toad species observed in the original survey have exhibited "serious declines." Two species, the foothill yellow-legged frog (*Rana boylii*) and great basin spadefoot toad (*Scaphiopus intermontanus*), were not observed in the survey area and the mountain yellow-legged frog (*Rana mucosa*), once the most abundant amphibian, has been reduced to a few small remnant populations.

It is likely that a number of different factors are contributing to the documented declines. One possible explanation suggests that the long-term cumulative effects of multiple factors, where natural low points in amphibian population cycles synergize with widespread environmental alterations (e.g., extended drought, chemical pollutants, predation by and competition with non-native species, and disease) will create extinction events (Jennings, 1996; Drost and Fellers, 1996). Recolonization of areas formerly occupied by some Sierra Nevada frog species is unlikely due to the widespread loss of populations and the presence of introduced predators (salmonids and char) (Bradford et al., 1993; Jennings, 1996).

Habitat

In a review of currently unlisted amphibian and reptile species, Jennings and Hayes (1994) concluded that 48 of the 80 taxa examined warranted reconsideration of their status. Species occurring in aquatic habitat types such as springs, seeps, marshes, and small headwater streams are at the greatest risk for continued population decline. Degradation and reduction of aquatic habitats has occurred statewide but some regions have experienced greater levels of habitat loss.

The fate of most forest amphibians will be determined outside of reserves (deMaynadier and Hunter, 1995). When considering the conservation of amphibians, the fundamental challenge for forest management is to minimize the differences between the quantity and quality of forest floor microhabitats.

The fate of most forest amphibians will be determined outside of reserves in the managed landscape.

Forest managers must also assess the differences between recently harvested stands and stands regenerated through natural disturbances. Forest management can increase soil temperature and decrease soil moisture by typically reducing canopy closure. Many terrestrial amphibians have narrow thermal tolerances given their respiratory physiology, and stable microclimates are important habitat attributes. The relative abundance of amphibians and soil moisture increased while soil temperature decreased in northwestern California along a gradient from clear cuts into adjacent thinned and unthinned forests (Karraker and Welsh, 2001). Welsh and Lind (1995) documented an important relationship with canopy closure for the Del Norte salamander (*Plethodon elongatus*) on interior forest sites in northwestern California. In contrast, Diller and Wallace (1994) found no relationship between canopy closure and this species presence in wetter coastal redwood sites.

Introduced exotics

A survey of 80 percent of the Trinity Alps Wilderness including 288 water bodies found that amphibian populations were either depressed or absent in a significant number of water bodies supporting either rainbow trout (*Oncorhynchus mykiss*), easten brook trout (*Salvelinus fontinalis*), or both species. Reproduction of Cascade frogs (*Rana cascadae*) rarely occurs in the presence of high fish populations. The species is seldom found in lakes of otherwise suitable habitat if introduced trout are also present in high numbers. Similarly, long-toed salamander (*Ambystoma macrodactylum*) larvae are particularly susceptible to predation by introduced trout in the Trinity Alps (Welsh and Boiano, 2001).

The mountain yellow-legged frog (*Rana mucosa*) has declined precipitously during the past century, and studies have implicated predation by introduced trout (*Salmo* spp.) as being at least partially responsible for the declines. Frog populations increased dramatically following removal of trout from Sierra Nevada lakes because of increased frog immigration and within lake recruitment of young (Knapp et al., 2001).

Introduced tiger salamanders (*Ambystoma tigrinum*) hybridize with the California tiger salamander (*Ambystoma californiense*) producing fertile offspring that threaten the genetic integrity of the California native (Fitzpatrick et al., 2001).

Estrogenic effects

Bioassays on seven populations of northern red-legged frogs (*Rana aurora aurora*) in northwestern California to determine the presence of endocrine-disrupting contaminants showed that four populations (57 percent) contained male frogs producing the protein vitellogenin. This protein is produced by mature female egg-laying vertebrates and is normally controlled by estrogen, an endocrine system hormone. The presence of vitellogenin in the serum of male and sub-adult frogs indicates an additional source of estrogen in the environment (Bettaso et al., 2001).

Diseases

Recently, the widespread occurrence of a fungus (Batrachochytrium) affecting Ranid frogs and causing Chytridiomycosis has been recognized as a particularly significant influence on population status. Fungal infections in amphibians are generally considered opportunistic as they infect animals stressed or weakened by other factors (Morell, 1999). One indicator of the presence of the disease is underdeveloped or malformed tadpole mouthparts. These abnormalities have been documented in six species from 73 sites in 10 California counties. Key questions currently being researched concern mechanisms that spread the virus and factors that might predispose populations to infection (Fellers, 2001).

Agrochemicals

Prevailing westerly winds convey airborne pesticides from California's Central Valley into the Sierra Nevada Mountains where they are deposited with rain or snow. More than 50 percent of the Pacific treefrogs (*Hyla regilla*) collected at Yosemite National Park had measurable chlorpyrifos or diazinon concentrations in their tissues. This is compared to only nine percent tissue concentrations at coastal sites. Other pesticides detected at high levels include endosulfan, DDT and derivatives, and hexachlorocyclohexanes. The residues indicate both a historic and current exposure to pesticides and high cholinesterase activities in the sampled frogs signify biological effects from this exposure (Sparling et al., 2001).

Glossary

bioassay: Biological evaluation of conditions as compared to a standard.

carrying capacity: The maximum population of a given organism that a particular environment or habitat can sustain; implies continuing yield without environmental damage. The carrying capacity changes over time according to the abundance of predators and resources (food and habitat).

Cavity Life History Group: Bird species nesting in natural or constructed cavities generally found in trees.

Chytridiomycosi: A fungal disease found in amphibians and associated with large population declines.

depredation: Removals or depletions of native, domestic, or commercial animals resulting in a loss in value.

DFG: California Department of Fish and Game.

endocrine-disrupting contaminants: Environmental pollutants altering the production of hormones and other chemical compounds from the endocrine system.

furbearer: Mammals harvested for the economic value of their fur.

game bird: Birds hunted for food or sport.

Habitat Relationship Model: A model developed to simulate or describe the complex relationship between an animal and it's habitat in a specific environmental setting.

Neotropical Migrant: Refers to bird species that nest in temperate regions and migrate to the Neotropical faunal region, which includes the West Indies, Mexico, Central America, and that part of South America within the tropics.

non-game mammal: Species that are not hunted for food or sport although some are harvested for their fur.

Open-cup nesting: Refers to bird species that construct nests on the ground or in a shrub or tree that is shaped like a cup and accessed from the top.

Literature cited

Barbour, M. and J. Major (editors). 1977. Terrestrial vegetation of California. New York: Wiley.

Barker, R. J. and J. R. Sauer. 1992. Modeling population change from time series data. In: McCullough, D.R. and R.H. Barrett (editors). Wildlife 2001: populations. New York: Elsevier Applied Science. pp. 182-194.

Bettaso, J.B., H.H. Welsh Jr., and B.D. Palmer. 2001. Test of an endocrine model: red-legged frogs as bioindicators of endocrine-disrupting contaminants. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

BioSystems Books.1994. Life on the Edge. Santa Cruz, CA.

Bradford, D.F., D.M. Graber, and F. Tabatabai. 1993. Isolation of remaining populations of the native frog, *Rana mucosa*, by introduced fish in Sequoia and Kings Canyon National Parks, California. Conservation Biology 7(4):882-888.

California Department of Fish and Game. 2001a. Bear hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001b. Bighorn sheep hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001c. Deer hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001d. Elk hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001e. Furbearing and non-game mammal hunting and trapping: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001f. Pronghorn antelope hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game. 2001g. Resident game bird hunting: final environmental document. Sacramento, CA.

California Department of Fish and Game and California Interagency Wildlife Task Group. 2001. California Wildlife Habitat Relationships System Version 7.0. Sacramento, CA. Web site accessed June 12, 2003. http://www.dfg.ca.gov/whdab/html/cwhr.html.

Dasmann, W.P. and J.A. Blaisdell. 1954. Deer and forage relationships on the Lassen-Washoe interstate winter deer range. California Department of Fish and Game. 40(3):215-234.

Declining Amphibian Populations Task Force. 2002. Home page of the Declining Amphibian Populations Task Force. Web site accessed June 12, 2003. http://www.open.ac.uk/daptf/.

deMaynadier, P.G. and M.L. Hunter Jr. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Review 3:230-261.

Diller, L.V. and R.L. Wallace. 1994. Distribution and habitat of *Plethodon elongatus* on managed, young growth forests in north coastal California. Journal of Herpetology 28:310-318.

Droege, S. 1990. The North American breeding bird survey. In: Sauer, J.R. and S. Droege (editors). Survey designs and statistical methods for the estimation of avian population trends. Biological report 90(1). Washington DC: U.S. Fish and Wildlife Service. pp. 1-4.

Drost, C.A. and G.M. Fellers. 1996. Collapse of a regional frog fauna in the Yosemite area of the California Sierra Nevada, USA. Conservation Biology 10(2):414-425.

Fellers, G.M. 2001. Recent observations of chytridiomycosis among amphibians in California. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Fitzpatrick, B.M., S.P.D. Riley, B.H. Shaffer, and V.S. Randal. 2001. Hybridization between an invasive exotic and a declining native amphibian: ecological, genetic, and political components of the threat. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Flather, C.H., S.J. Brady, and M.S. Knowles. 1999. Wildlife resource trends in the United States: a technical document supporting the 2000 USDA Forest Service RPA Assessment. General Technical Report RMRS-GTR-33. Fort Collins, CO: U.S. Forest Service, Rocky Mountain Research Station.

Graber, D.M. 1996. Status of terrestrial vertebrates. In: Centers for Water and Wildland Resources. 1996. Sierra Nevada Ecosystem Project final report to Congress: status of the Sierra Nevada; volume II: assessments and scientific basis for management options. Wildland Resources Center Report No. 37. pp. 709-734. Davis, CA: University of California, Davis.

Grinnell, J. and T.I. Storer. 1924. Animal life in Yosemite. Berkeley, CA: University of California Press.

Gruell, G.E. 1986. Post-1900 mule deer irruptions in the intermountain west: principal causes and influences. General Technical Report INT-206. Ogden, UT: U.S. Forest Service, Intermountain Research Station.

Hall, F. 2001. Personal communication. California Department of Fish and Game. December 13, 2001.

Hobbs, N.T. 1989. Linking energy balance to survival in mule deer: development and test of a simulation model. Wildlife Monograph Number 101.

Jennings, M.R. 1995. Native ranid frogs in California. In: LaRoe, E.T., G.S. Farris, C.E. Puckett, P.D. Doran, and M.J. Mac (editors). Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. Washington, DC: U.S. Department of the Interior, National Biological Service. pp. 131-134.

Jennings, M.R. 1996. Status of amphibians. In: Centers for Water and Wildland Resources. 1996. Sierra Nevada Ecosystem Project final report to Congress: status of the Sierra Nevada; volume II: assessments and scientific basis for management options. Wildland Resources Center Report No. 37. pp. 921-944. Davis, CA: University of California, Davis.

Jennings, M.R. and M.P. Hayes. 1994. Amphibian and reptile species of special concern in California. Rancho Cordova, CA: Inland Fisheries Division, California Department of Fish and Game.

Karraker, N.E. and H.H. Welsh, Jr. 2001. Effects of silvicultural edges on terrestrial amphibian abundance and microclimate in northwestern California. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Knapp, R.A., V. Vredenberg, and C. Milliron. 2001. Removal of non-native trout from Sierra Nevada lakes as a means to recover mountain yellow-legged frog populations. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Lassen, R.W., C.M. Ferrel, and H. Leach. 1952. Food habits, productivity and condition of the Doyle mule deer herd. California Department of Fish and Game. 38(3):211-224.

Leach, H.R. 1956. Food habits of the Great Basin deer herd of California. California Fish and Game 42(3):243-308.

Leach, H.R. 1988. Food items eaten by 16 Rocky Mountain mule deer from the Buffalo Creek area, Washoe Co., Nevada. California Department of Fish and Game. 1p.

Lyon, J.L., K.B. Aubry, W.J. Zielinski, S.W. Buskirk, and L.F. Ruggiero. 1994. The scientific basis for conserving forest carnivores: considerations for management. Pg 128-137. In: L.F. Ruggiero, K.B. Aubry, S.W. Buskirk, L.J. Lyon, and W.J. Zielinski (eds.). The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. Gen. Tech. Rep. RM-254. Fort Collins, CO: Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture; 184pp

Martin, R.E. and C.H. Driver. 1983. Factors affecting antelope bitterbrush reestablishment following fire. In: Tiedemann, A.R. and K.L. Johnson (compilers). Proceedings-research and management of bitterbrush and cliffrose in western North America. General Technical Report INT-152. Odgen, UT: U.S. Forest Service, Intermountain Forest and Range Experiment Station. pp. 266-279.

McCullough, D.R. 1969. The tule elk: its history, behavior and ecology. Univ. Calif. Pub. Zool. 88:1-209.

Moen, A.N. 1973. Wildlife ecology: an analytical approach. San Francisco, CA: W.H. Freeman and Company.

Morell, V. 1999. Are pathogens felling frogs? Science 284:728-731.

Morrison, M.L., B.G. Marcot and R.W. Mannan. 1992. Wildlife habitat relationships: concepts and applications. Madison, WI: University of Wisconsin Press.

Nord, E.C. 1965. Autecology of bitterbrush in California. Ecological Monographs 35(3):307-344.

Neal, D.L. 1981. Improvement of Great Basin deer winter range with livestock grazing. In: Nelson, L., Jr and J.M. Peek. Proceedings of the wildlife-livestock relationships symposium, April 20-22, 1981, Coeur d'Alene, Idaho. Moscow, ID: University of Idaho. pp. 61-73.

Patton, D.R. 1992. Wildlife habitat relationships in forested ecosystems. Portland, OR: Timber Press.

Peterjohn, B.G. and J.R. Sauer. 1993. North American breeding bird survey annual summary, 1990-1991. Bird Populations 1(1):1-15.

Pyshora, L. 1977. The pronghorn antelope in northeastern California. Wildlife Management Administrative Report Number 77-2. Sacramento, CA: California Department of Fish and Game.

Rice, C.L. 1983. A literature review of the fire relationships of antelope bitterbrush. In: Tiedemann, A.R. and K.L. Johnson (compilers). Proceedings-research and management of bitterbrush and cliffrose in Western North America. General Technical Report INT-152. Odgen, UT: U.S. Forest Service, Intermountain Forest and Range Experiment Station. pp. 256-265.

Salwasser, H. 1979. The ecology and management of the Devil's Garden interstate deer herd and its range. Thesis, PhD. Berkeley, CA: University of California, Berkeley.

Sauer, J.R., G.W. Pendelton, and B.G. Peterjohn. 1996. Evaluating causes of population change in North American insectivorous songbirds. Conservation Biology 10(2):465-478.

Sauer, J.R., J.E. Hines, I. Thomas, J. Fallon, and G. Gough. 2000. The North American Breeding Bird Survey, results and analysis 1966 -1999. Version 98. Laurel, MD: U.S. Geologic Survey, Patuxent Wildlife Research Center.

Sitton, L.W. and S. Wallen. 1976. California mountain lion study. Sacramento, CA: California Department of Fish and Game.

Shor, B. 1988. Body fat indices of deer collected in the Buffalo Creek area, Washoe Co., Nevada. California Department of Fish and Game. 2p.

Sparling, D.W., G. Fellers, L. McConnell, and D. Cowman. 2001. Biological effects of pesticides on declining amphibian populations in the Sierra Nevada. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Temple, S.A. and J.A. Wiens. 1989. Bird populations and environmental changes: can birds be bio-indicators? American Birds 43:260-270.

Thomas, L. and K. Martin. 1996. The importance of analysis method for breeding bird survey population trend estimates. Conservation Biology 10(2):479-490.

Torres, S. 2000. Counting cougars in California. Outdoor California 61(3):7-9.

Torres, S.G., T.M. Mansfield, J.E. Foley, T. Lupo, and A. Brinkhaus. 1996. Mountain lion and human activity in California: testing speculations. Wildlife Society Bulletin 24(3):451-460.

U.S. Geological Survey. 2001. North American Breeding Bird Survey. Patuxent Wildlife Research Center. Web site accessed June 12, 2003. http://www.mp2-pwrc.usgs.gov/bbs/.

Updike, D.R., E.R. Loft, and F.A. Hall. 1990. Wildfires on big sagebrush/antelope bitterbrush range in northeastern California: implications for deer populations. In: McArthur, E.D., E.M. Romney, S.D. Smith, and P.T. Tueller (compilers). Proceedings-symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 1989 April 5-7; Las Vegas, NV. General Technical Report INT-276. Ogden, UT: U.S. Forest Service, Intermountain Research Station. pp. 41-46.

Veirs, Jr., S.D. and P.A. Opler. 1998. California. In: Mac, M.J., P.A. Opler, C.E. Puckett Haecker, and P.D. Doran. Status and trends of the nation's biological resources. Reston, VA: U.S. Geological Survey. pp. 593-644.

Wallmo, O.C., L.H. Carpenter, W.L. Regelin, R.B. Gill, and D.L. Baker. 1977. Evaluation of deer habitat on a nutritional basis. Journal of Range Management. 30(2):122-127.

Welch, B.L., E.D. McArthur, D.L. Nelson, J.C. Pederson, and J.N. Davis. 1986. Hobble Creek—a superior selection of low-elevation mountain big sagebrush. Research Paper INT-370. Ogden, UT: U.S. Forest Service, Intermountain Research Station.

Welsh, H.H., Jr. and A.J. Lind. 1995. Habitat correlates of the Del Norte salamander, *Plethodon elongatus* (Caudata: Plethodontidae), in northwestern California. Journal of Herpetology 29:198-210.

Welsh, H.H., Jr. and D.M. Boiano. 2001. Amphibian assemblages and introduced fishes in the Trinity Alps Wilderness of the Klamath Mountains of northwestern California. Program and abstracts. 2001 Annual meeting of the Western Section of The Wildlife Society. Sacramento, CA: The Wildlife Society-Western Section.

Young, J.A., R.A. Evans, and J. Major. 1972. Alien plants in the Great Basin. Journal of Range Management 25(3):194-201.