Courtesy D. Stevens, NPS

Interior West

by
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Overview

The articles in this section reveal the critical need for ecosystem science to direct ecosystem management in areas ranging from the Colorado Rockies, south to the Colorado Plateau, west to the Great Basin and the Pacific Northwest, and north to the Greater Yellowstone Ecosystem. Ecosystems in the Interior West are challenged by severe climatic fluctuations superimposed on rapidly changing land-use patterns and anthropogenic (human-caused) threats. Because scientists and resource managers now recognize the prohibitive cost and difficulty of a species-by-species approach to biological conservation and wise stewardship, their efforts are moving increasingly toward an ecosystem and landscape approach to conservation.

My colleagues and I begin this section by identifying and quantifying anthropogenic threats to ecosystem integrity in Rocky Mountain National Park and the Colorado Rockies (Stohlgren et al.). The article by Schullery continues this common theme by describing alarming trends in plant and animal populations in the Greater Yellowstone Ecosystem. By taking a broad view of subalpine forest dynamics in the Pacific Northwest, Peterson shows that treeline communities may be adversely influenced by rapid environmental change. Warshall examines the southwestern sky island ecosystems (the mountaintops of the Great Basin) with respect to threats from nonindigenous species, recreation and military practices, and fire-management activities.

The status and trends of many plant and animal populations are uncertain in the Interior West. Scoppettone and Rissler, however, report successful population increases of the endangered cui-ui fish (*Chasmistes cujus*) in Pyramid Lake, Nevada: the population has doubled between 1990 and 1993. Mueller and Marsh focus on how loss of critical riparian habitat through water development, pollution, and the introduction of nonindigenous species have caused population declines of the threatened and endangered razorback sucker (*Xyrauchen texanus*) and bonytail (*Gila elegans*) in the Colorado River Basin. The article by Drost and Deshler on the diversity of reptiles and amphibians on the Colorado Plateau reminds us that much inventory and monitoring work lies ahead. Van Riper III et al. also remind us that human activities in the past (e.g., pesticide use, water diversion, and the introduction of nonindigenous trout) continue to affect the status and trends of bald eagle (*Haliaeetus leucocephalus*) populations on the southern Colorado Plateau. And, Willey demonstrates that 90% of the threatened Mexican spotted owl (*Strix occidentalis lucida*) habitat on the Colorado Plateau is on timber-management sites.

The last two articles focus on restoring ecosystem integrity by reintroducing extirpated species. Singer reports that the success of restoration efforts of bighorn sheep (*Ovis canadensis*) in the Rocky Mountains is influenced negatively by their proximity to domestic sheep and by small, translocated groups of bighorn sheep that are too genetically similar. McCutchen discusses the history and status of desert bighorn (*O.c. nelsoni*) and shows that sheep translocations have been fairly successful, except in New Mexico and southern California.

It is important to note the overriding theme in this section: modern humans continue to alter ecosystem components and processes. To manage natural resources in a sustainable way to meet the needs of the American people, we must first understand the inseparable link of human and resource ecology. The perpetuation of biological diversity in the Interior West depends largely on coordinated, multiscale ecosystem science, and resource inventory and monitoring efforts.

Ecosystem Trends in the Colorado Rockies

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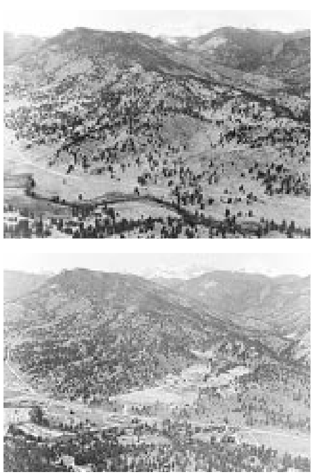


Fig. 1. Drastically increased urbanization in Estes Park/Rocky Mountain National Park, 1921 (above) to 1986 (below). The photographs also show, however, forest recovery from turn-of-the-century logging and human-caused fires (Veblen and Lorenz 1991).

Biological conservation is increasingly moving toward an ecosystem and landscape approach, recognizing the prohibitive cost and difficulty of a species-by-species approach (LaRoe 1993). Also, statewide (e.g., Gap Analysis Program) and national surveys (e.g., Environmental Monitoring and Assessment Program or EMAP) are conducted at a scale and level of resolution that do not meet the needs of most small land-management units that require detailed information at the ecosystem and landscape scale (Stohlgren 1994). The Colorado Rockies are an ideal outdoor laboratory for ecosystem science and management. The escalating environmental threats described in this article compelled us to design a landscape-scale assessment of the status and trends of biotic resources.

Our guiding principle is that a strong ecosystem science program provides crucial information for ecosystem management and wise stewardship. We define ecosystem science as the long-term, interdisciplinary study of ecosystem components and processes and their interactions at multiple spatial, temporal, and organizational scales, to meet management needs.

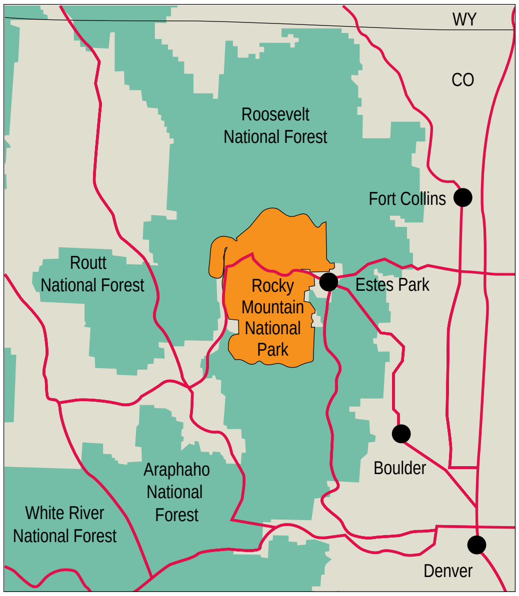
About 76% of the land adjacent to Rocky Mountain National Park is federal land. While the area has not received as much attention as the Greater Yellowstone Ecosystem, there may be as many internal and external threats to the natural resources in the area. The Colorado Rockies are an archetypal ecosystem under siege. Like many national parks, wilderness areas, wildlife refuges, and other natural areas, common threats include encroachment from urbanization and development, habitat fragmentation, fire suppression, nonindigenous species' invasion, and global change (e.g., climate change, bordering land-use changes, and air and water pollution). Since all these threats transcend ownership or stewardship borders, so have interagency concerns for conservation, inventory and monitoring, and research.

Here we identify and quantify trends that threaten ecosystem integrity in Rocky Mountain National Park and the Colorado Rockies. Our specific objectives include presenting qualitative information on vegetation change over the past 65 years, documenting quantitative trends of an ecosystem under siege, showing preliminary results of a long-term global change research program, and discussing the role of ecosystem science in assessing long-term trends in ecosystem condition.

Status and Trends

There is little doubt that the ecosystems of the Colorado Rockies have been altered significantly by humans. The density of ponderosa pine woodlands has increased (Fig. 1) as has suburban development (Veblen and Lorenz 1991). These qualitative changes are supported by qualitative measures (Fig. 2). The response of the forest from turn-of-the-century logging and fires showed a 5-fold increase in ponderosa pine bole (*see* glossary) biomass. In addition, the human population in Estes Park and the number of visitors in Rocky Mountain National Park have almost doubled since 1960. Urban development throughout the Front Range of Colorado has resulted in increased air pollution. Annual wet deposition values for nitrate, ammonium, and sulfate in the Loch Vale watershed of Rocky Mountain National Park are significantly greater than the average values of 2-4 kg/ha (about 2-4 lb/acre) in remote areas of the world (Fig. 2).

Elk and moose populations continue to increase in the park (Fig. 2) for many reasons including reduced predation (wolves have been extirpated) and hunting as well as diminished habitat and migratory corridors outside the park. Researchers are now quantifying ungulate (hooved herbivores) habitat relationships and aspen-willow community conservation. Although agricultural land use in Larimer County has declined slightly in recent years (Fig. 2), landscape and ecosystem integrity is



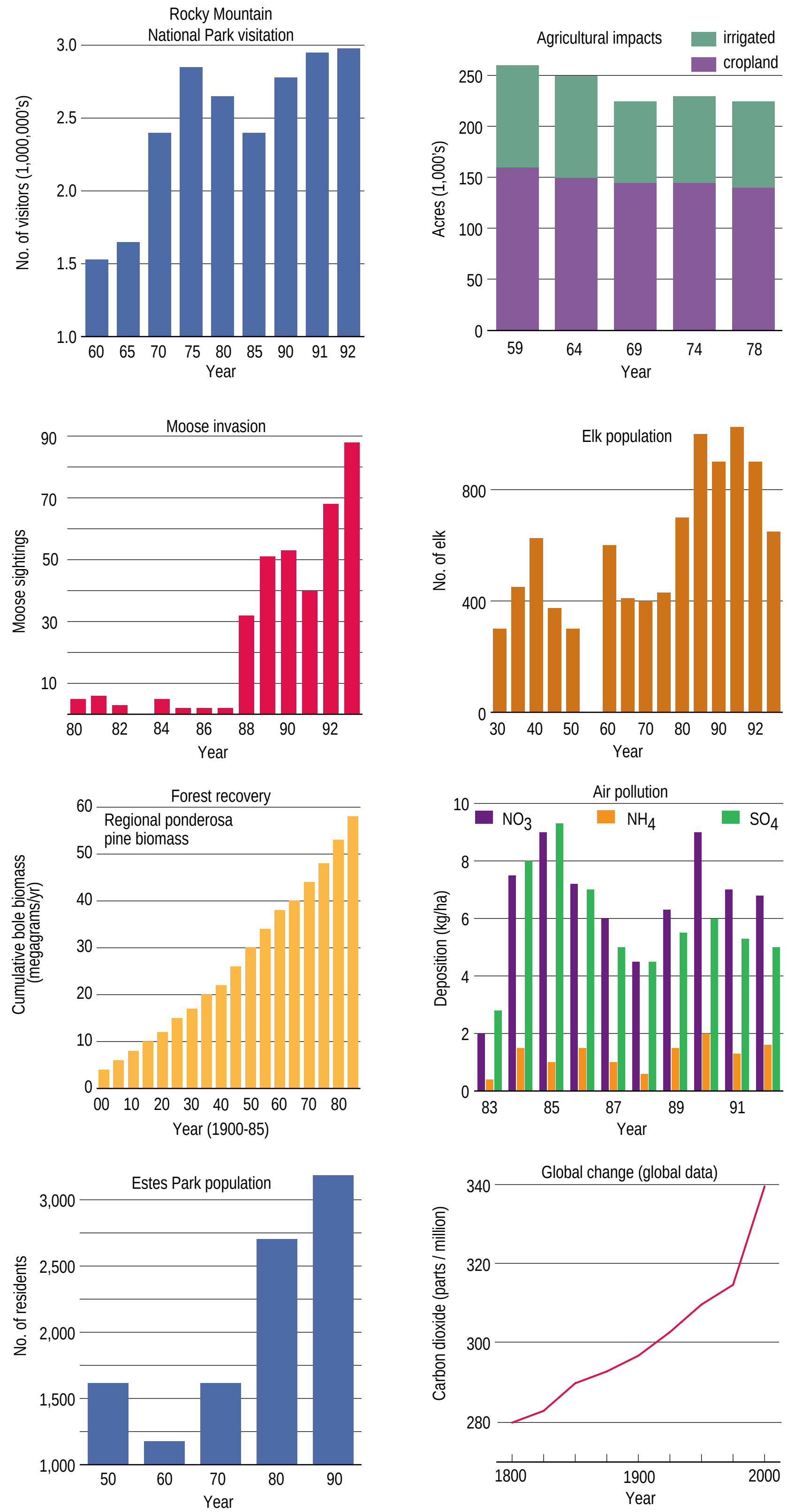


Fig. 2. Trends in Rocky Mountain National Park visitors, agricultural impacts, moose invasion, elk population, forest recovery, air pollution, Estes Park population, and global change in carbon dioxide.

challenged by fire suppression, nonindigenous species' invasions, weather modification (i.e., cloud seeding), and global climate change (Stohlgren et al. 1993).

Just as a species-by-species approach to conservation biology is prohibitively expensive, a complex of ecosystem threats cannot be addressed one by one. Our interdisciplinary approach in the Colorado Rockies is based on developing partnerships, consolidating and evaluating the status and trends in existing data, and developing a biogeographical, long-term, multiple spatial-scale monitoring program that fills information gaps and provides a scientific basis for sound ecosystem management. Preliminary results from the National Biological Service global climate change research program show significant interactions of climate, hydrological, and vegetation systems.

Mesoscale (1- to 100-km grids) climate modeling in the Front Range of the Colorado Rockies demonstrated that changes in land cover (e.g,. wild prairie to irrigated agricultural land) can lead to significant and perhaps unexpected changes in mesoscale climate. Computer modeling results indicate that the severity of summer thunderstorms in Rocky Mountain National Park is influenced by spatial patterns in albedo (*see* glossary) and surface roughness of farmlands several kilometers away (Pielke et al. 1993).

Quantifying trends in mountain hydrology and vegetation change caused by global climate change and assessing the effects of nearby cloud seeding require the development of new predictive models (Baron et al. 1994). Hydrological models are proving effective at estimating stream discharge and regional water supply.

In our long-term forest plots, we found the old-growth spruce and fir forests of the central Rocky Mountains range in biomass from

150,000 to more than 320,000 kg/ha (133,828-285,500 lb/acre) in standing biomass, and annual tree growth remains relatively high in these ancient forests. We are finding that ecotones are sensitive indicators of forest change; the forest-tundra ecotone (transitional area between distinct habitats or ecosystems) in Rocky Mountain National Park has been undergoing substantial directional change for some time (Baker et al. 1994). There is substantial evidence of seedling and sapling invasion within some previously unforested areas within the ecotone, particularly in wet areas in the patch forest zone. This filling in of the ecotone could substantially alter the ecotone environment (Baker et al. 1994). There is little evidence, however, of upward establishment of trees into tundra. To synthesize the vegetation change data, we are developing predictive vegetation change models by using geographic information systems. Our long-term study plots and transects will validate future models.

Implications

This interdisciplinary approach can be widely applied to most U.S. Department of the Interior land units and most ecosystems and will be an essential link to large-scale inventory and monitoring programs (e.g., Gap Analysis Program and EMAP). Ecosystem science is the most logical approach to determine the status and long-term trends of selected resources, populations, and ecosystems. This approach fosters discovery, standardization, linkages, and partnerships as well as coordinated inventory, monitoring, and research. New, standardized sampling protocols are being developed to accurately assess vascular plant species richness, an index of biodiversity (Stohlgren 1994).

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The Greater Yellowstone Ecosystem

by
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Greater Yellowstone is described as the last large, nearly intact ecosystem in the northern temperate zone of the earth (Reese 1984; Keiter and Boyce 1991). Conflict over management has been controversial, and the area is a flagship site among conservation groups that aggressively promote ecosystem management (Greater Yellowstone Coalition 1992). The Greater Yellow Ecosystem (GYE) is one of the world's foremost natural laboratories in landscape ecology and geology and is a world-renowned recreational site (Knight 1994).

History

Yellowstone National Park (YNP) boundaries were arbitrarily drawn in 1872 in hopes of including all regional geothermal basins. No other landscape considerations were incorporated. By the 1970's, however, the grizzly bear's (*Ursus arctos*) range in and near YNP became the first informal minimum boundary of a theoretical Greater Yellowstone Ecosystem that included at least 1,600,000 ha (4,000,000 acres; Schullery 1992). Since then, definitions of the GYE have steadily grown larger (Fig. 1). A 1994 study listed the GYE size as 7,689,000 ha (19,000,000 acres; Clark and Minta 1994), while a 1994 speech by a Greater Yellowstone Coalition leader enlarged that to 8,000,000 ha (20,000,000 acres; Wilcox 1994).

In 1985 the House Subcommittees on Public Lands and National Parks and Recreation held a joint subcommittee hearing on Greater Yellowstone, resulting in a report by the Congressional Research Service (1986) outlining shortcomings in interagency coordination and concluding that the area's essential values were at risk.

Ecosystem Management by Species

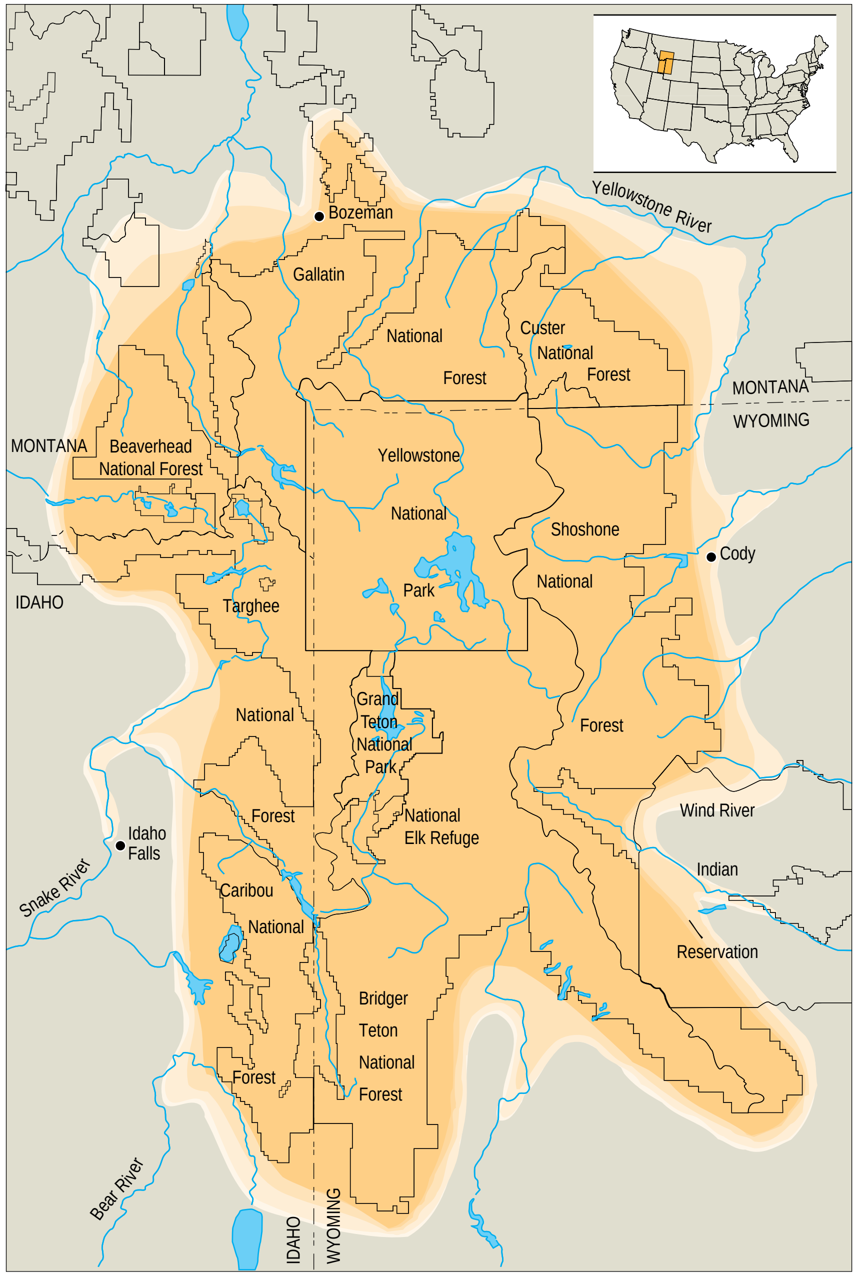
The GYE concept has been most often advanced through concerns over individual species rather than over broader ecological principles. GYE managers must keep at least two types of "long-term" status in mind. One is the known, or at least probable, trend of a species based on historical and prehistorical information. The second type is that which has existed since the beginning of formal scientific study. Though 20 or 30 or even 50 years of information on a population may be considered long-term by some, one of the important lessons of GYE management is that even half a century is not long enough to give us a full idea of how a species may vary in its occupation of a wild ecosystem.

For example, anecdotal information on grizzly bear abundance dates to the mid-1800's (Schullery and Whittlesey 1992), and administrators have made informal population estimates for more than 70 years (Schullery 1992). From these sources, we know the species was common in the GYE when Europeans arrived, and we know that the population was not isolated before the 1930's, but is now. We do not know if bears were more or less common than now.

A 1959-70 bear study suggested a grizzly bear population size of about 175, later revised to about 229 (Craighead et al. 1974). Later estimates have ranged as low as 136 and as high as 540 (Schullery 1992); the most recent is a minimum estimate of 236 (Servheen 1993). Although the GYE population is relatively close to recovery goals, the plan's definition of recovery is controversial (Mattson and Reid 1991; Schullery 1992). Thus, even though the population may be stable or possibly increasing in the short term, in the longer term, continued habitat loss and increasing human activities may well reverse the trend.

Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*) have suffered considerable declines since European settlement, but recently began flourishing (Varley and Schullery 1983) in some areas. Especially in Yellowstone Lake itself, long-term records indicate an almost remarkable restoration of robust populations from only three decades ago when the numbers of this fish were depleted because of excessive harvest (Gresswell and Varley 1988). Its current recovery, though a significant management achievement, does not begin to restore the species' historical abundance.

Early accounts of pronghorn (*Antilocapra americana*) in the GYE described herds of hundreds seen ranging through most major river valleys (Schullery and Whittlesey 1992). These populations were decimated by 1900, and declines continued among remaining herds. On the park's northern range, pronghorn declined from 500-700 in the 1930's to about 122 in 1968 (Houston 1982). By 1992 the herd had increased to 536 (J. Mack, National Park Service, personal communication).



Courtesy Desktop Assistance and Greater Yellowstone Coalition.

Fig. 1. Progressively lighter shading is used around the edges of a recent map of the Greater Yellowstone Ecosystem to illustrate the uncertainty that still plagues definitions of the ecosystem.

Among plants, whitebark pine (*Pinus albicaulis*) is a species of special interest, in large part because of its seasonal importance to grizzly bears, but also because its distribution could be dramatically reduced by relatively minor global warming (Blanchard and Knight 1991; Romme and Turner 1991; Fig. 2). In this case, we do not have a good long-term data set on the species, but we understand its ecology well enough to project declining future status.

Estimates of the decline of quaking aspen (*Populus tremuloides*) on YNP's northern range since 1872 range from 50% to 95% (Houston 1982; Kay 1993), and perhaps no controversy underway in the GYE more clearly reveals the need for comprehensive interdisciplinary research. Several factors are suspected in the aspen's changing status, including Native American influences on numerous mammal species and on fire-return intervals before the creation of the park in 1872; European influences on fire frequency since 1886; regional climate warming; human harvests of beaver and ungulates in the first 15 years of the park's history and of wolves and other predators before 1930; human settlement of traditional ungulate migration routes north of the park since 1872; ungulate (especially elk) effects on all other parts of the ecosystem since 1900; and human influences on elk distribution in the park (Houston 1982; Schullery and Whittlesey 1992; Kay 1993).

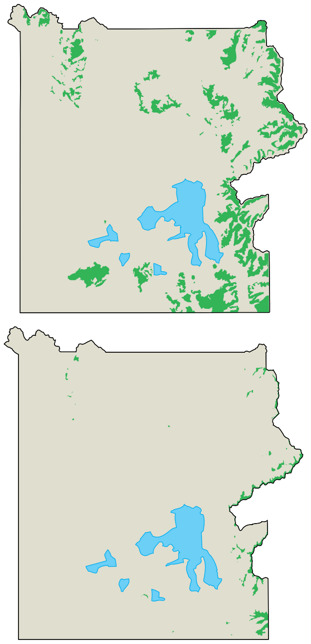


Fig. 2. Top: Current distribution of whitebark pine portrayed by a computerized geographic information system (GIS). Bottom: Distribution of whitebark pine projected by GIS analysis under a modest increase in warmth and dryness, showing a decrease of approximately 90%. (Derived from Romme and Turner [1991] by the Yellowstone GIS Laboratory, Yellowstone National Park.)

Conclusions

Research is but one component of land-management decisions (Varley 1993). While in some respects the GYE has fulfilled the promise of early scientists who described it as one of the foremost natural laboratories on earth, both managers and researchers need more information to deal with the increasing demands on the region's resources, either in terms of raw information or in terms of an ecosystem-level understanding. In YNP, a landscape model is being developed based on a computerized geographic information system that will integrate, analyze, and display information from many disciplines (Shovic et al. 1993). Through this level of synthesis we may be able to better understand trends in the GYE.

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Subalpine Forests of Western North America

by
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Subalpine forest and meadow ecosystems are important, climatically sensitive components of mountainous regions of western North America (Peterson 1991). Changes in temperature, precipitation, snowpack, storm frequency, and fire all could affect the growth and productivity of these systems, resulting in substantial shifts in the location of ecotones (*see* glossary) between subalpine and alpine zones and montane and subalpine zones (Canaday and Fonda 1974).

Subalpine forests of western North America provide an excellent opportunity to examine response to past climate variation. Trees in the subalpine zone are frequently more than 500 years old and respond to climatic variations over annual to centuries-long time scales. The magnitude of climatic variation these forests have experienced may be compared with projections of future climate resulting from increased concentration of greenhouse gases. The population dynamics of subalpine tree species can be used to interpret climatic conditions under which trees have regenerated and can indicate how subalpine forest and meadow ecotones changed in the past. Preserved pollen and plant fossils can be used to examine subalpine vegetation distribution during different climatic periods of the Holocene (since the last ice age).

Recent literature on the potential effects of climate change has focused on changes in the growth and distribution of low-elevation forests (e.g., Woodman 1987; Davis 1989). In western North America, most low-elevation forests are sensitive to soil moisture deficits during relatively dry summers (Peterson et al. 1991; Graybill et al. 1992). Although subalpine forests have been the subject of considerably less study, it appears that snowpack is an important limiting factor to growth, with respect to length of growing season (Graumlich 1991; Peterson 1993). Duration of snowpack also limits seedling establishment in subalpine meadows (Fonda 1976) and after disturbance by fire (Little et al. 1994). Summer temperature also positively affects the growth of mature subalpine conifers (Graumlich 1991; Peterson 1993) and negatively affects the seedlings' survival (Little et al. 1994).

Several reports document recent increases in the growth of subalpine conifer species in western North America (Innes 1991) as well as recent increases in the abundance of subalpine conifer populations at several locations. This article reviews recent reports of changes in the growth and distribution of subalpine conifers in western North America and discusses some possible causes.

Courtesy D.L. Peterson, NBS

Many subalpine forests in western North America, such as this site in the Olympic Mountains, are currently protected in national parks and wilderness areas. Some of these areas have been experiencing increased tree growth and rapid establishment of young trees during the past century.

Tree Growth

The first prominent report of a recent increase in growth of subalpine coniferous species was published by LaMarche et al. (1984), who reported dramatic increases in the growth rate of bristlecone pine (*Pinus longaeva*, *P. aristata*) and limber pine (*P. flexilis*) in California and Nevada. The extreme age of these trees, combined with the fact that radial growth has increased since 1850, makes this a particularly interesting result. The authors suggested that elevated levels of carbon dioxide associated with fossil fuel combustion may enhance the growth and productivity of these trees, perhaps through increased water-use efficiency. A more recent examination of these data corroborates the growth increase and restates that carbon dioxide fertilization is the hypothesized cause of the increase (Graybill and Idso 1993). Some disagreement exists about the factors causing the growth increase and whether the increases in growth found in these studies (which included sampling of strip-bark trees) are representative of the populations as a whole (Cooper and Gale 1986).

A subsequent study of basal area growth trends of lodgepole pine (*P. contorta*) and whitebark pine (*P. albicaulis*) at sites above 3,000-m elevation in the east-central Sierra Nevada of California also revealed that a high proportion of trees has had recent growth increases (Peterson et al. 1990), with the onset of the increase normally between 1850 and 1900, as found by LaMarche et al. (1984). Growth was particularly rapid during the past 30 years or so.

There are other reports of recent growth increases in subalpine conifers of western North America (Innes 1991). Jacoby (1986) found radial growth increases in lodgepole pine in the San Jacinto Mountains of southern California, but did not identify a strong causal factor despite detailed climatic analysis. Graumlich et al. (1989) found increases in the growth and productivity of Pacific silver fir (*Abies amabilis*) and mountain hemlock (*Tsuga mertensiana*) in the Cascade Mountains of Washington State, and suggested that these trends were related to increased temperature.

Recent growth increases have also been reported in European conifers (Innes 1991), such as Norway spruce (*Picea abies*; Kienast and Luxmoore 1988; Briffa 1992) and silver fir (*Abies alba*; Becker 1989), although these species are generally found below the subalpine zone. Both increased carbon dioxide (Kienast and Luxmoore 1988) and temperature (Becker 1989; Briffa 1992) have been suggested as potential causes for increased growth.

Not all studies of subalpine conifers have found recent increased growth, however. Graumlich (1991), for example, did not find increased radial growth in foxtail pine (*Pinus balfouriana*), limber pine, and western juniper (*Juniperus occidentalis*) in the Sierra Nevada. It is difficult to compare the various studies of tree growth discussed here because the studies employed a diversity of sampling and analytical techniques to evaluate growth patterns.

As noted previously, there are several potential explanations for recent increased growth in subalpine conifers. The possibility of carbon dioxide fertilization has been supported by experimental studies (Graybill and Idso 1993), but is extremely difficult to demonstrate for mature trees in the field. Increased temperature is another potential cause, but its relationship with growth is correlative and also difficult to demonstrate for mature trees. Changes in snowpack duration, which affects length of growing season, are a more likely cause of growth increases. Unfortunately, the long-term relationship of snowpack to tree growth has not been adequately investigated because snowpack data are often difficult to obtain.

Fertilization through nitrogen deposition could be another cause of growth increases. Although nitrogen deposition is relatively low in western North America, it is probably somewhat higher now than in the past because of the combustion of fossil fuels. Many subalpine forests are in sites with shallow soils and relatively low fertility, so even a small increase in nitrogen input could have some effect over several decades. Finally, the growth increases may simply be the result of normal forest stand dynamics because relatively little is known about the growth and ecological characteristics of subalpine forest ecosystems. Although the observed increases appear abnormal compared to lower elevation species, they may in fact be a normal phenomenon that reflects the natural range of variation in growth of subalpine species. Growth response to climate or other factors likely varies considerably by region (e.g., the Rocky Mountains have a continental climate, the Sierra Nevadas a Mediterranean climate) and by microsite (north aspect versus south aspect).

Patterns of Establishment

Recent increases in tree establishment in subalpine meadows have been documented in mountainous regions throughout western North America (Rochefort et al. 1994). Most locations show an expansion of the forest margin after 1890, with establishment peaks between 1920 and 1950. Additional establishment peaks have been identified on a local basis. Most investigators have concluded that increases in tree establishment are the result of a warmer climate following the Little Ice Age (Franklin et al. 1971; Kearney 1982; Heikkinen 1984; Butler 1986). It is unclear if establishment patterns signify a long-term directional change or short-term variation in relatively stable ecotones, regardless of the potential causes.

Most studies on subalpine tree establishment have been conducted in the Pacific Northwest in British Columbia in Canada and Washington and Oregon (Woodward et al. 1991; Rochefort et al. 1994) where tree invasion in subalpine meadows is widespread. Trees in this area are rapidly becoming established (Rochefort and Peterson 1991; Woodward et al. 1991), especially in meadows dominated by ericaceous species (species in the heath family such as heather and huckleberries). Much of this establishment is occurring in concavities and other places where snow would normally accumulate and inhibit germination and survival (personal observation). As trees become established, tree clumps act as black bodies to increase the absorption of radiation, snowmelt occurs progressively earlier, tree canopies intercept (and allow sublimation of) snow, and tree survival adjacent to the tree clump is further enhanced. This progression of events is termed “contagious dispersion” (Payette and Filion 1985).

Eight separate studies in the Pacific Northwest have documented large increases in populations of subalpine fir (*Abies lasiocarpa*), Pacific silver fir, mountain hemlock, subalpine larch (*Larix lyallii*), and Alaska yellow-cedar (*Chamaecyparis nootkatensis*). All these species experienced increases in establishment between 1920 and 1950. This was generally a period of lower snowpacks, which probably allowed seedlings to become established during a longer growing season. Winter precipitation limits subalpine tree growth and establishment in the Pacific Northwest, which has a maritime climate with wet winters and dry summers; high summer temperature can also limit tree establishment because shallow-rooted seedlings are subject to soil moisture stress (Little et al. 1994).

Increases in establishment of three species have been documented in the Sierra Nevada and White Mountains of California: foxtail pine, lodgepole pine, and bristlecone pine. Soil moisture stress is clearly a limiting factor in this area, which is dominated by a Mediterranean climate with very dry summers. Temporal patterns of establishment are inconsistent among the different locations in this region, and there has been little documented establishment during the past 20 years.

Studies conducted in the Rocky Mountains have documented increases in subalpine tree establishment for subalpine fir, lodgepole pine, and Engelmann spruce (*Picea engelmannii*). This region is dominated by a continental climate, with low precipitation and cold winters. Temporal patterns of establishment were more consistent in the Rocky Mountains, especially during 1940-50, a period with a warmer, wetter climate.

It is unclear whether observations of subalpine tree invasions are isolated events or part of a broad pattern in western North America. There are insufficient data from locations other than the Pacific Northwest to speculate about the geographic extent of this phenomenon.

Future Changes

Data on subalpine tree growth for western North America are too sparse to infer that growth increases are a broad regional phenomenon. Additional data from other sites are needed to quantify growth trends in subalpine species. Furthermore, consistent sampling and analytical methods should be applied so that different data sets can be compared.

Sufficient information exists, however, about long-term growth trends and shorter-term response of growth to climate to make some general predictions about potential growth under future climate scenarios. If the climate becomes warmer and drier, as predicted by general circulation models, growth rates of subalpine conifers will probably increase. This growth increase would depend on the seasonality of precipitation. A decrease in snowfall would be particularly beneficial to species such as subalpine fir and Engelmann spruce (Ettl and Peterson 1991; Peterson 1993), although warmer summer temperatures could cause summer soil moisture deficits that would be detrimental to growth. It is unknown how future growth patterns will be influenced by increased concentrations of carbon dioxide. Any potential growth changes must, of course, be considered with respect to the effects of climate change on interspecific competition and disturbance, as well as deposition of nitrogen or other nutrients.

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Southwestern Sky Island Ecosystems

by
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The "sky islands" of Arizona and New Mexico in the southwestern United States form a unique complex of about 27 mountain ranges whose boundaries, at their lowest elevation, are desert scrub, grasslands, or oak woodlands (Figs. 1 and 2; Table 1). Since the last glaciation, these forested mountain ranges have become relatively isolated from each other. Expanding desert grasslands and desert scrub in the valleys ("the sea" between the sky islands) have limited genetic interchange between populations and created environments with high evolutionary potential. The resulting sky island ecosystems support many perennial streams in an arid climate, have a high number of endemic species, and harbor most game species as well as most threatened and endangered species in the Southwest.

The southwestern sky island "archipelago" is unique on the planet. It is the only sky-island complex extending from subtropical to temperate latitudes (compared to the Great Basin, the Venezuelan, and the African sky islands) with an exceptionally complex pattern of species of northern and southern origins. The "continents" that have been the main sources of species for the archipelago are the Sierra Madre of Mexico and the Rocky Mountains of the United States, although the flora has been influenced by the Californian, Sonoran, Intermountain, Cordilleran, and Sierra Madrean Floristic Provinces (S. McLaughlin, University of Arizona, unpublished data).

The ecosystems of each mountain range are of major interest to resource managers concerned with preserving each sky island's unique biogeography and biological diversity as well as to the public for recreation. Land uses sometimes conflict on the sky islands: camping, rock climbing, car-based tourism, military maneuvers, hunting, fishing, exotic grass and fish stocking, grazing, water-supply withdrawals, timber and fuelwood extraction, bird watching, critical habitat for threatened and endangered species, skiing, summer homes, mining, scientific research, sacred Native American ceremonies, and archaeological sites.

Most American sky islands are within the Gila River basin. About 15 additional sky islands are in Mexico and will not be discussed here. Nevertheless, the cross-border management of sky islands is important for such tasks as reintroduction of the Mexican wolf (*Canis lupus baileyi*), maintenance of disjunct populations of rare plant species, and migration of the Mexican pronghorn (*Antilocarpa americana mexicana*), if it still occurs.

Fig. 1. Sky island mountain ranges of Arizona, New Mexico, and adjacent Sonora and Chihuahua (Marshall 1957). All of the labeled mountain ranges have pine-oak woodland.

Status of Information

The floras of the largest sky islands of Arizona have been inventoried (S. McLaughlin, University of Arizona, unpublished data), including most insular, endemic, and rare species. Certain inventory gaps (e.g., the Baboquivari, Galiuro, Santa Rita, Whetstone, and Patagonia mountains) exist. In addition, the

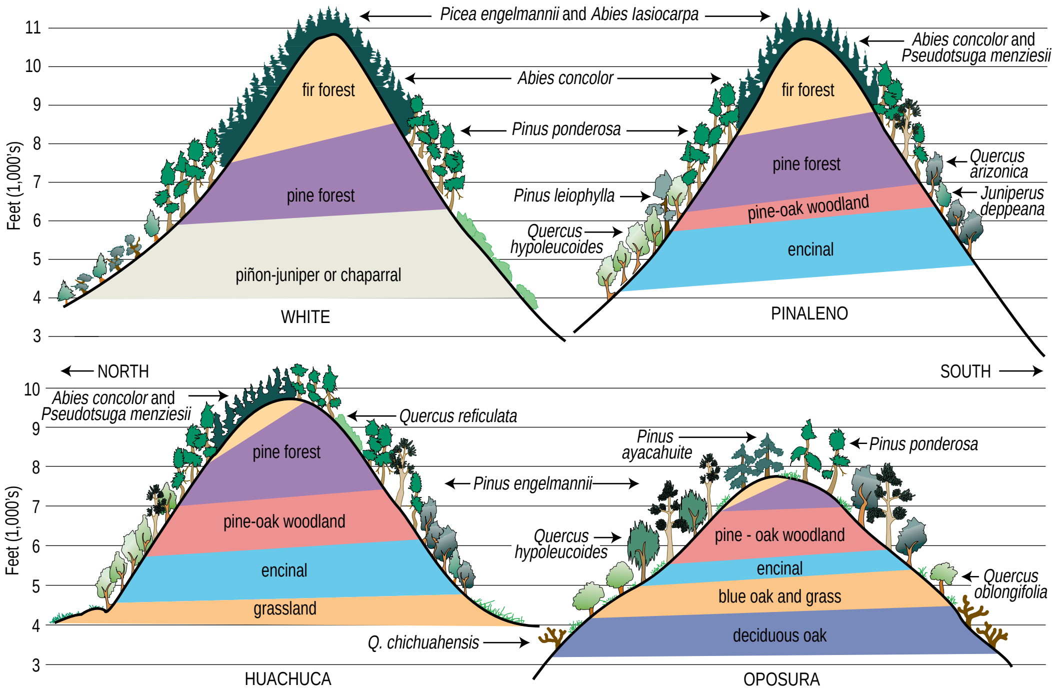

Fig. 2. Cross-sections of three sky islands showing the "stacked" biotic communities varying with latitude. The White Mountains are close to the Rocky Mountain flora and fauna. The Oposura Mountains begin to show the full development of the Sierra Madre communities (Marshall 1957).

number of sites for rare or insular plants, their abundance at each site, and other species diversity indices are lacking for many species of concern. The areal extent, age class, structural characters, and regeneration rates of the five or six biotic communities on the sky islands are poorly known, especially for oak woodlands (McPherson 1992).

Fungi have been intensely inventoried on the Chiricahuas, though only partial inventories exist for ranges of mycorrhizal hypogeous fungi, truffles, and false truffles (States 1990; Nishida et al. 1992). The lichen flora, one of the most diverse and complex in western North America, is poorly inventoried for almost all the sky islands.

The highest sky islands, except the Peloncillos and the Animas, have been intensely inventoried for all groups of insects (C. Olson, University of Arizona, personal communication). Spider and pseudoscorpion distribution is poorly understood. The larger millipedes and scorpions have been extensively collected, but the micro-millipedes, the insular flightless beetles, and the flightless grasshoppers in the upper elevations are poorly known. For instance, a 6-week survey on top of the Pinalenos yielded three new species of flightless beetles (Warshall 1986).

The land mollusks have been inventoried (Bequaert and Miller 1973), though their range extensions and taxa need review. The cienaga (wetland) mollusks are being studied by the Smithsonian. Fish, birds, amphibians, reptiles, and mammals are well-inventoried and yield continuing surprises such as the recent discovery of the Ramsey Canyon leopard frog (*Rana subaquavocalis*). Specific inventory and monitoring gaps in frequency and abundance of sensitive species remain.

Flora and Fauna

A major dividing line between the flora and fauna of southern and northern origins occurs in the sky island ecoregion. The sky island complex harbors more than 2,000 plant species. Of the more than 190 snail species in the Southwest, the sky islands support 3 endemic genera, and over 60 endemic species, including the genus *Sonorella* and the monotypic genus

Table 1. Sky island forested ecosystems of Arizona and New Mexico.

Mountains	High point	Elevation (ft)	Base (ft)	Range (ft)
Pinaleno	Mount Graham	10,720	4,000	6,720
Santa Catalina	Mount Lemmon	9,157	3,000	6,157
Rincon	Mica Mountain	8,666	3,000	5,666
Santa Rita	Mount Wrightson	9,453	4,000	5,453
Chiricahua	Chiricahua Peak	9,796	4,500	5,296
Huachucas	Miller Peak	9,466	5,000	4,466
Dos Cabezas	Dos Cabezas	8,369	4,000	4,369
Animas	Animas Peak	8,519	4,500	4,019
Baboquivari	Baboquivari Peak	7,730	3,500	4,230
Galiuro	Bassett Peak	7,650	4,000	3,650
Santa Teresa	Cottonwood Mountain	7,489	4,000	3,489
Winchester	Reiley Peak	7,631	4,400	3,231
Whetstone	Apache Peak	7,684	4,800	2,884
Dragoon	Mt. Glenn	7,519	4,700	2,819
Patagonia	Mt. Washington	7,221	4,500	2,721
Mule	Mt. Ballard	7,370	4,800	2,570
Tucson	Wasson Peak	4,687	2,200	2,487
Atascosa	Atascosa Peak	6,440	4,000	2,440
Swisshelm	Swisshelm Mountain	7,185	4,800	2,385
Peloncillo	Owl Peak	6,625	4,500	2,125
Tumacacori	(Unnamed)	5,634	3,500	2,134
Sierrita	Samaniego Peak	5,991	4,000	1,991
Mustang	(Unnamed)	6,469	4,750	1,719
Empire	(Unnamed)	5,588	4,000	1,588
Pajarito	Pajarito Peak	5,236	4,000	1,236

Table 2. Selected sensitive, rare, and endangered plants of the sky islands.

Species		Location	Status
Common name	Scientific name		
Agave	*Agave parviflora* ssp. *parviflora*	Patagonia/Atascosa/Santa Rita/S. Cibuta	Rare
Wild onion	*Allium gooddingii*	Catalina	Disjunct
Amsonia	*Amsonia kearneyana*	Baboquivari	Endemic/rare
Aster	*Aster potosinus*	Huachuca	Single population
Milk-vetch	*Astragalus cobrensis* var. *maguirei*	Chiricahua/Peloncillo	Insular
Milk-vetch	*A. hypoxylus*	Patagonia/Huachuca	Very rare
Zorillo	*Choisya mollis*	Atascosa	Endemic
Pincushion cactus	*Coryphantha robbinsorum*	Peloncillo	Endemic
Climbing milkweed	*Cynanchum wigginsii*	Atascosa/Patagonia/Baboquivari/Mule	Very rare
Fleabane	*Erigeron kuschei*	Chiricahua	Endemic
Fleabane	*E. heliographis*	Pinaleno	Endemic
Fleabane	*E. lemmonii*	Huachuca	Endemic
Lemon lily	*Lillium parryi*	Huachuca/Santa Rita/Chiricahua	Rare
(No common name)	*Lilaeopsis schaffneriana recurva*	Huachuca	Rare
(No common name)	*Perityle cochisensis*	Chiricahua	Endemic
Dock, sorrel	*Rumex orthoneurus*	Chiricahua/Sierra Ancha/Pinaleno/Huachuca	Unique/rare
Groundsel	*Senecio huachucanus*	Santa Rita/Huachuca	Rare
Sophora	*Sophora arizonica*	Pinaleno /Swisshelm/Whetstone/Hualapai	Vulnerable
Lady's-tresses	*Spiranthes delitescens*	Canelo	Endemic

Chaenaxis. More than 75 reptile species inhabit the sky islands, one of the most diverse herpetological regions in North America with several endemic races.

About 265 bird species occur within the sky island complex, including valley and riparian species. About 30 are of subtropical origin and have their northern limits within the sky island complex. The sky islands are the most diverse U.S. area for mammals; some 90 native mammals inhabit the area from the chaparral community to higher elevations; at least 6 are endemic subspecies.

Trends and Management

Researchers have begun measuring biological trends in six major categories concerning inventory, monitoring, preservation, and restoration that are most pertinent to sky island forested ecosystems. A discussion of these categories follows.

Endemics and Insular Species

With new investigative techniques, there has been growing respect for the genetic diversity of this area, especially late-Cenozoic and Pleistocene relict faunal populations (see examples, Tables 2, 3). For instance, recent genetic analyses on the Mt. Graham red squirrel (*Tamasciuris hudsonicus grahamensis*) and the lemon lily (*Lillium parryi*) showed that both populations are more highly divergent from closely related populations than previously thought. Increasingly, however, local and insular species have hybridized with introduced races and species; hybridization is particularly evident among fish (e.g., hybrids of the Apache trout, *Oncorhynchus apache*, with the rainbow trout, *O. mykiss*), but can also be found among white-tail deer (*Odocoileus virginianus* versus *hemionus*), pronghorn (*Antilocapra americana* versus *mexicana*), turkeys (*Meleagris merriami* versus *mexicana*), and bighorn sheep (*Ovis mexicana* versus *nelsoni*).

Selected rare, unique, threatened, and endangered species and subspecies whose critical habitat includes the sky islands are listed in Tables 2 and 3. Figure 3 compares the Coronado National Forest to other forests. The number of sensitive plant and animal species from the sky island ecoregion has increased over the last 20 years. (Sensitive means that the population's viability is of concern and requires monitoring or active protection.) The increase is, in part, the product of more detailed knowledge. For instance, a recent review of *Erigeron pringlei* split this fleabane into four species, creating a new endemic, *E. heliographis*, on the Pinalenos. Nevertheless, the Coronado Forest reports 56 sensitive plants, among the largest number reported from any national forest, including 1 on the federal endangered list and 3 candidate species. McLaughlin (University of Arizona, unpublished data) suggested that the local extirpation of six plant species in the last century was related to either global warming, habitat alteration, or both.

Seven insects are listed by the Coronado National Forest as species of concern (U.S. Forest Service, unpublished memo). About 12

Table 3. Various threatened, endangered, and candidate species.*

Common name	Scientific name	Location	Status
Yaqui catfish	*Ictalurus pricei*	San Bernardino Creek	E
Yaqui topminnow	*Poeciliopsis occidentalis sonoriensis*	Swisshelm	E
Apache trout	*Oncorhynchus apache*	Various reintroductions	T
Gila chub	*Gila intermedia*	Gila drainages	(T)
Rosetail chub	*G. robusta*	Galiuro	(T)
Sonora chub	*G. ditaenia*	Atascoca	E
Spikedace	*Meda fulgida*	Galiuro/Pinaleno	T
Loach minnow	*Tiaroga cobitis*	Galiuro/Pinaleno	T
Mexican stoneroller	*Campostoma ornatum*	Chiricahua	T
Barking frog	*Hylactophryne augusti*	Pajarito/Santa Rita	(E)
Chiricahua leopard frog	*Rana chiricahuensis*	Chiricahua (+)	(T)
Great Plains narrow-mouthed toad	*Gastrophryne olivacea*	Various	(C)
Mexican garter snake	*Thamnophis eques*	Various	(C)
Ridge-nosed rattlesnake	*Crotalus willardi willardi*	Huachuca, Patagonia, Santa Rita	(C)
Ridge-nosed rattlesnake	*C. willardi obscurus*	Animas; San Luis	(C)
Thick-billed parrot	*Rhynchopsitta pachyrhyncha*	Various (Chiricahua)	E
Buff-breasted flycatcher	*Empidonax fulvifrons*	Huachuca	(E)
Gray hawk	*Buteo nitidus*	Santa Cruz/San Pedro drainages	(T)
Yellow-billed cuckoo	*Coccyzus americanus*	Various gallery forests	(T)
Mexican spotted owl	*Strix occidentalis lucida*	Various	E
Northern goshawk	*Accipiter gentilis*	Various	C/T
Peregrine falcon	*Falco peregrinus*	Various	E
Sanborn's long-nosed bat	*Leptonycteris sanborni*	Various	E
Mt. Graham red squirrel	*Tamasciuris hudsonicus grahamensis*	Pinaleno	E
Mexican wolf	*Canis lupus baileyi*	Extirpated	E
Grizzly bear	*Ursus arctos*	Extirpated	T
Jaguar	*Felis onca*	Extirpated	E
Ocelot	*F. pardalis*	Extirpated	E
Mexican long-tongued bat	*Choeronycteris mexicana*	Various	T
Arizona shrew	*Sorex arizonae*	Chiricahua, Huachuca, Santa Rita	C
Red bat	*Lasiurus borealis*	Various	(C)

*This list is incomplete because of revisions since publication in 1988. Only animals inhabiting sky islands within or above the oak-pine woodlands are included. Some wetland species have not been included. The species must have been on federal lists. Various — more than two sky-island ecosystems; T — threatened; E — endangered; C — candidate in Arizona or New Mexico; ( ) — listing by Arizona Fish and Game only; federal status not yet determined.

fish species are considered vulnerable, including 9 federally listed and 7 living in sky island drainages (Table 3). Within this national forest, there are 11 amphibians whose population viability is of concern, though none is on the federal list; 8 dwell in the valleys or lower drainages of the sky islands. They are sensitive to upstream watershed alterations. Fourteen sky island reptiles are considered sensitive but not federally listed. There are about 55 bird species of concern, 5 federally listed, and about 20 whose population viability is of significant concern (Table 3). About 30 mammals are of concern, 4 federally listed. The grizzly bear, jaguar, ocelot, Tarahumara frog, and gray wolf have been extirpated from the sky island archipelago. Not counting the extirpated species and the 11 bats of concern, there are 13 mammal species and subspecies dwelling on the sky islands that have low populations of concern (Table 3).

Distribution

Some of the most interesting aspects of sky island ecosystems and history are why some mountains lack a particular species ("holes"), why some species skip mountain ranges or appear as an exception in an otherwise species-poor flora or fauna ("outliers"), and why some species, even mobile animals such as birds, end their distribution on a particular sky island (Warshall 1986). For example, why are there no chipmunks on the Huachucas? Why does the Mexican chickadee (*Parus sclateri*) stop on the Chiricahuas, but only 35 mi away the mountain chickadee (*P. gambeli*) inhabits the Pinalenos? Why are there no voles on the Catalinas?

Colonization of the sky islands by exotic species is increasing with over 60 non-native plants having established regenerative populations in the Arizona sky islands. Major issues include limiting introductions of buffel grass (*Pennisetum cilaris*) and exotic lovegrasses (*Eragrostis* spp.) by the U.S. Department of Agriculture, as well as controlling and restoring habitats swamped by exotic forbs such as *Euryops multifida* on the Pinalenos and Catalinas. Fifteen non-native fish species have been added to the five or six native freshwater fish families, with consequent hybridization, predation, and competition throughout springs and drainages. The Central Arizona Project has become a new corridor for exotic fish, some of which are invading the last strongholds of natives.

Three feral exotic mammals may have colonized the sky island complex. The opossum (*Didelphis* spp.) colonization is believed to be a mix of released Virginia opposum (*D. virginiana* and range-expanding Mexican opossum (*D. marsupialis*). It has been reported but not confirmed that the European ferret (*Mustela putorius*) has become feral in the Huachucas. Over the last 50 years, about 12 birds and mammals of southern origin have been recorded colonizing more northern sky islands. No animal species is known to have retreated south except the extirpated jaguar, ocelot, and thick-billed parrot. A few species such as the Abert's squirrel (*Sciuris aberti*) have been introduced for hunting and have then expanded their range. The monitoring of these changes will be an important barometer to how habitat changes, species introductions, and climate interact with ecosystem management practices.

Vertical Migration

Each sky island has a unique ecosystem with a stack of life zones ranging from arid to boreal (Fig. 2). Many species migrate vertically to feed and breed at various elevations. The Pinalenos contain the most stacked life zones in the shortest vertical distance of any mountain in North America. By traversing five biotic communities in a few hours, bears can feed on *Opuntia* (prickly pear cactus) fruit in the morning and grass roots growing in semi-alpine meadows in the afternoon. Assuring minimal viable habitat size and the appropriate forest age-class structure to support animal populations with vertical migration is an unstudied aspect of forest ecosystem management.

In addition, various biotic communities are remnants of colder climates with small relict acreage. For instance, only about 243 ha (600 acres) of spruce-fir (*Picea engelmannii-Abies lasiocarpa*) forest are left within the sky island complex. This forest type, found only on the Pinalenos, is critical habitat for the endemic and federally listed endangered Mt. Graham red squirrel, which also inhabits the transition to mixed conifer forests (Douglas fir-white fir; *Pseudotsuga menziesii-Abies concolor*) at lower elevations. These two plant associations, heavily logged and cleared, will not become mature enough to supply the minimum viable habitat to ensure the squirrel population's survival for 2 centuries. Annual growth rates, seeding rates, and regeneration cycles have become less predictable with the unknown effect of global warming and fire risk, requiring rethinking of the minimum size required for viable habitats.

Special Habitats

Special habitats and plant associations (e.g., high-elevation cienagas, limestone outcrops, perennial streams, talus slopes) create islands of habitat within the sky island ecosystem, increasing biological richness. For instance, talus slopes support a series of endemic land snails; the rock cliffs and outcrops support plant species such as the fleabanes *Erigeron lemmonii*, *E. heliographis*, and *E. pringlei*, found

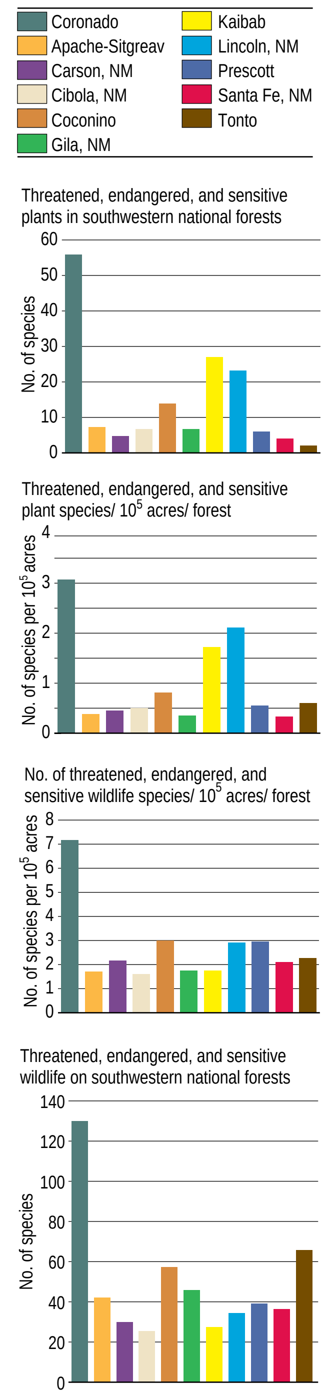

Fig. 3. Comparison of the Coronado National Forest, which administers all or part of 16 sky islands, with national forests with no sky island flora and fauna.

nowhere else in the world. The perennial streams support seven rare native fish species. Of the special habitats, the cienagas (swampy, marshy cover) and perennial streams require the most monitoring, protection, and restoration.

Special Interest Game

The densest populations of most game species are found on the sky islands. For instance, the densest populations of black bear (*Ursus americanus*) and mountain lion (*Felis concolor*) south of the Mogollon Rim are on the Pinalenos. In general, over a 20-year period, both species increased with population troughs occurring from rancher depredation and drought. White-tailed deer (*Odocoileus virginianus*) populations have increased, while mule deer are less stable. Javelina (*Tayassu tajacu*) are stable or declining, having suffered from canine distemper after the drought of 1989. Band-tailed pigeon (*Columba fasciata*), a species dependent on sky island forests, has had a long-term decline as have two subspecies of turkey.

Corridors

For many land animals, corridors of animal movement between sky islands have been through riparian zones. Increasing habitat fragmentation from increased subdivisions around the base of the sky islands is further isolating some populations, especially in the Tucson area, which separates the Santa Catalina and Rincon mountains from the Tucsons and the Santa Ritas. This structural change in migration patterns has not been studied but is believed to be the most significant threat to "safe passage" corridors between sky islands.

In summary, the single best indicator of ecosystem management has been the increasing number of threatened and endangered populations (USFS 1993). This trend requires increasing acreage of critical or otherwise protected habitat; increased monitoring and control over the introduction and spread of exotic grasses, fish, and gamebirds, and the reintroduction of locally extirpated mammals and tree species in restoration projects.

Other Issues

Fire management is planned to reduce catastrophic fires from fuel build ups, to allow natural burns required by certain species, and to increase fire suppression to maintain remnant old-growth forest biodiversity. Experimentation and debate about fire management are widespread, however. Another trend is toward the restriction of cattle to prevent overgrazing and trampling, to protect sensitive plant species, and to protect and promote recovery of wetland and riparian habitats. A third trend is toward upstream rehabilitation in specific watersheds where flooding endangers sensitive plants.

In addition, there is increasing urban pressure on the Forest Service to clear more habitat for recreation such as camping and skiing (on the Catalinas) and to expand roads into the sky islands for greater access and uses that can conflict with habitat protection (USFS 1993). Managing land use on private inholdings, on properties adjacent to public land, and on properties bordering intermountain corridors will be increasingly important.

The final trend is the unknown impact of global warming on the biseasonal (winter and summer) rainfall pattern of the southwestern sky islands. This trend is of special importance because of the large number of relict and insular species and subspecies in the region.

Because of the geological, topographic, and biological uniqueness of each sky island, the policies for each mountain range will need to be custom-designed on a watershed by watershed basis.

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Endangered Cui-ui of Pyramid Lake, Nevada

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Cui-ui (*Chasmistes cujus*) is a large plankton-feeding fish that only occurs in Pyramid Lake, Nevada. It was put on the federal endangered list in 1967 based on declining population and absence of reproduction. A lake dweller, cui-ui is a stream spawner. Most of this century, this sucker species was unable to access the Truckee River, Pyramid Lake's only perennial tributary, to reproduce. Water diversion from the Truckee River, as a result of the nation's first Bureau of Reclamation project (Newlands Project), reduced the lake elevation and, in most years, caused an impassable delta to form at the mouth of the Truckee River. Cui-ui live more than 40 years; it is this longevity that has allowed the species to persist for as many as 19 years with virtually no recruitment (*see* glossary) to the adult population (Scoppettone 1988).

Cui-ui is one of three remaining species of the genus *Chasmistes*. Of the three, its habitat is most intact, and it thus has the best opportunity for recovery (Scoppettone and Vinyard 1991). Each spring, cui-ui adults, most of which mature at 8-12 years of age, migrate to the mouth of the Truckee River at the south end of Pyramid Lake, where they aggregate, awaiting environmental cues and sufficient stream flow to enter the river (Scoppettone et al. 1986). This behavior provides an excellent opportunity to capture the adults for estimating population numbers and year-class (year hatched) structure. In this article we report changes in adult cui-ui population number and year-class structure from spring 1983 to spring 1993.

Status and Trends

Each spring, cui-ui are captured, anchor-tagged, and released for recapture. The proportion of tagged to untagged fish is used to estimate population number. Virtually all mature adults enter the prespawning aggregate each year (Scoppettone, unpublished data); thus an estimate of the number of adults entering the aggregate is an estimate of the entire adult population. We provide data of 4 select years (1983, 1991, 1992, and 1993) to illustrate trends between 1983 to 1994.

Courtesy G.G. Scoppettone, NBS

Cui-ui (*Chasmistes cujus*) with dorsal radio tag.

Captures of cui-ui from the prespawning aggregate have been successful enough to give us reliable estimates of the adult population. In 1982 and 1983, 3,000 adults were captured and tagged. From 1989 through 1993, captures increased markedly because of a change in capture gear and increased population. More than 100,000 cui-ui have been captured, and tags were applied to 60% of these. By spring 1993, tag returns were close to 4% of the fish captured.

The adult cui-ui population has increased 10-fold from 1983 to 1993 (Fig. 1), an increase attributed in part to unusually wet years from 1980 to 1986. During these years more than 65,000 adults entered the lower Truckee River to spawn, and produced more than 250 million cui-ui larvae for Pyramid Lake. In contrast, virtually no spawning occurred in the Truckee River from 1988 through 1992, a fact that will probably be reflected later in this decade as a downward trend in the number of adults.

Adult Year-class Structure

To understand cui-ui demographics and why the species is still considered endangered, it is necessary to understand its year-class structure. In 1983 when there were about 100,000 adult cui-ui in Pyramid Lake, almost 90% were from a single year class produced in 1969; the second predominate class represented about 5% of the population and was hatched in 1950 (Fig. 2). From 1950 to 1968 and from 1970 to 1979, very little recruitment occurred. The situation has improved; in 1991, the 1981 year class replaced the 1969 in predominance, and it remained so through 1993. In 1993, 400,000 of the estimated 1 million adults were fish that had been hatched in 1981. The dramatic increase in the spawning population from 1991 to 1992 is assumed to be those fish that hatched in 1981, 1982, and 1983 and finally reached adulthood.

These improvements in population numbers and year-class structure are partly attributed to several extraordinarily wet years; similar conditions may not occur with sufficient frequency to assure species recovery or preclude extinction.

In addition to the prespawning aggregate, the adult and juvenile populations have been sampled around Pyramid Lake throughout the year. Our results suggest that few juveniles hatched after 1986, and thereby provide testimony to inconsistency in cui-ui recruitment.

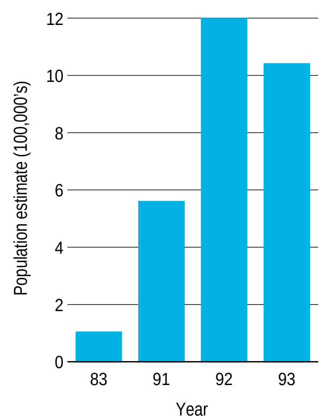

Fig. 1. Estimated population of adult cui-ui in spring 1983, 1991, 1992, and 1993.

Spawning cui-ui.

Future Outlook

The cui-ui has an excellent prognosis for recovery. It has an approved recovery plan and supporting legislation (P.L.101-616), which provide for acquisition of water and water rights to elevate Pyramid Lake, improve fish passage over the delta, and enhance spawning flows. Plans to acquire water for Pyramid Lake and cui-ui are being developed. Cui-ui population trends over the past 10 years demonstrate the rebound potential of the species when it is provided with passage and sufficient water for reproduction. Because limited water is available for acquisition, however, Truckee River flows required for cui-ui recovery need to be precisely determined. Our monitoring of the adult cui-ui population is part of a cui-ui population dynamics study aimed at calibrating an existing Truckee River water-management model being used for cui-ui recovery. Monitoring will continue through the 1998 spawning season, at which time sufficient information should have been generated to calibrate the model.

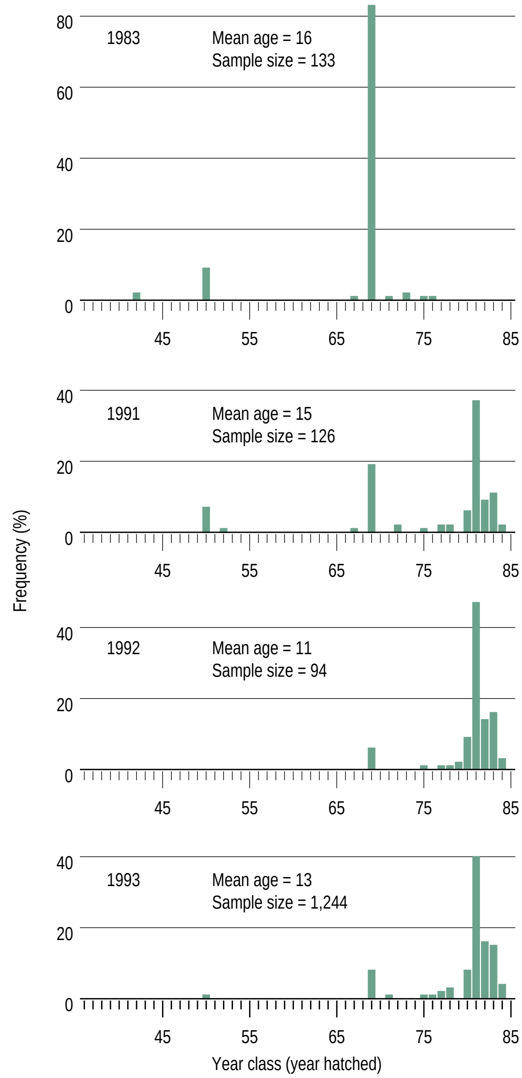

Fig. 2. Year-class structure of adult cui-ui in spring 1983, 1991, 1992, and 1993.

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Bonytail and Razorback Sucker in the Colorado River Basin

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Bonytail (*Gila elegans*) and razorback sucker (*Xyrauchen texanus*) are large river fish found only in western North America's Colorado River basin. The bonytail is nearly extinct and the razorback sucker is becoming rare.

The bonytail (Fig. 1) is a large, streamlined minnow (family Cyprinidae) that may reach 50 cm (18 in) in length and weigh up to 0.5 kg (1 lb). The razorback sucker (Catostomidae; Fig. 2) may grow to 75 cm (2.5 ft) in length and weigh up to 5 kg (10 lb). Both species have evolved a unique dorsal keel or hump, a characteristic shared by few other fish. Individual life spans approach 50 years.

Historically, both species were common and were used by Native Americans and early settlers as food and fertilizer. Physical and biological changes to their habitat and direct competition and predation from non-native fishes are responsible for their decline. Young fish no longer survive to replace adults as they die of old age.

Status

Information about these fish is found in sources ranging from scattered personal journals of early travelers to more recent biological reports and scientific literature. Bonytail and razorback sucker were first described by scientists in the late 1850's. Comprehensive studies were not conducted in the lower Colorado River until 1930, while similar investigations upstream were delayed until the 1960's because the area is rugged and remote.

Dill (1941) reported an alarming decline of endemic fish in the lower river; Miller et al. (1982) reported similar trends farther upstream. Three years after the 1973 passage of the Endangered Species Act, the Colorado River

Fishes Recovery Team was formed. The Colorado River Fishery Project was established in 1978 to recover threatened and endangered fish in 965 km (600 mi) of the upper Colorado and Green rivers. Recovery efforts intensified in 1987 with the Recovery Implementation Program. These and other projects have funded major research on the biology and habitat needs of these species.

Bonytails were historically common in the mainstem Colorado, Green, Gunnison, Yampa, Gila, and Salt rivers before the construction of large dams. Bonytail became rare in the lower river system by 1935 and suffered similar declines farther upstream by the mid-1960's. The last confirmed bonytail taken from any river was in 1985. Bonytail continue to be captured in low numbers from Lake Mohave in Arizona and Nevada, a reservoir on the Colorado River downstream of Hoover Dam.

Razorback suckers were historically common to abundant in the Colorado mainstem and portions of the Green, San Juan, Animas, Duchesne, Gila, Salt, and Verde rivers. Razorback suckers also had begun declining in the lower river by 1935, but were commercially harvested near Grand Junction, Colorado, and Phoenix, Arizona, until 1950. Numbers dramatically declined in the upper Colorado River during the 1970's and 1980's, and today the fish is very rare. The largest river population is in the Green River, Utah, and is estimated (1993) at fewer than 500 adults.

Large populations of razorback sucker developed in some newly created reservoirs in the lower river before fish predators became abundant. For example, populations that numbered into the hundreds of thousands became established in the Salton Sea, Roosevelt Lake, Saguaro Lake, Lake Havasu, Lake Mead, Lake Mohave, and Senator Wash Reservoir. Predation by non-native fishes eventually proved overwhelming, and, without recruitment (addition of individuals to a population through reproduction and immigration), populations disappeared after 40 to 50 years.

Razorback suckers are now rare except in Lake Mohave, which supports the last large population. Spawning is successful there, but as was true at older reservoirs, young razorback suckers are eaten by sunfish, bass, and other fish. The reservoir population declined by 60% between 1988 (59,500) and 1991 (23,300). Remaining suckers are expected to die by the end of the decade.

It is unlikely that the bonytail and razorback will survive in the wild. No measurable recruitment is evident in any part of the drainage and old individuals are reaching the end of their life span. Bonytail are found in less than 2% of their former range, and razorback sucker in less than 25% of their former range (Fig. 3).

Reasons for Decline

The Colorado River ecosystem has been dramatically altered by water development that transformed an erratic and turbulent river system into a series of calm reservoirs and channelized river reaches. Eight dams were built across the lower 563 km (350 mi) of the river by 1950. The historical habitat of these fish is now controlled by 44 large dams and is being drained by hundreds of miles of diversion canals. Nursery areas, critical for early life stages, have been flooded by reservoirs, and upstream migration is physically blocked by dams. Seasonally warm and turbid flows of the natural hydrology of the basin were replaced by cold, diminished reservoir releases governed by hydroelectric and downstream water demands.

Although physical habitat changes have been dramatic, subtle ecological changes may have been even more damaging. Reservoirs and cold tailwaters presented favorable conditions to develop recreational fisheries. Although the bonytail and razorback sucker were once valuable food sources, they became viewed as trash fish when more desirable sportfish (e.g., trout, catfish, and bass) became established. Resource agencies stocked and promoted recreational fisheries, often at the expense of native fishes. For example, in 1962, 723 km (450 mi) of the upper Green River was poisoned to improve trout production. Today, over 90% of all fish found in the river system are species introduced for recreational fishing. Uncounted other aquatic plants and animals, pathogens, parasites, and

Fig. 1. Bonytail (*Gila elegans*).

Fig. 2. Razorback sucker (*Xyrauchen texanus*).

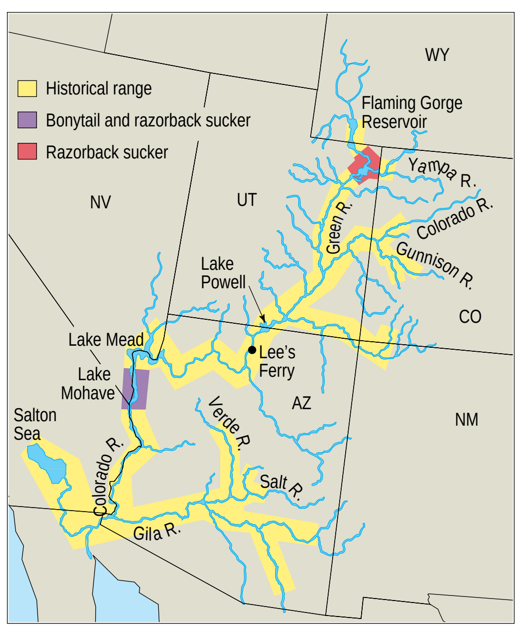

Fig. 3. Historical range and current concentrations of bonytail and razorback sucker (Minckley and Deacon 1991).

chemical contaminants were introduced and have changed the river's delicate ecosystem.

The dramatic decline prompted the listing of the bonytail as endangered in 1980, and a similar listing for the razorback sucker followed in 1991. Although both fishes are federally protected and recovery programs began over 15 years ago, these species continue to edge toward extinction. The problem lies in the complexity of the environmental and legal issues, combined with possible conflicts in land-, water-, and fishery-management philosophies. Controversy and debate have slowed, stalled, and complicated recovery effort. While sociopolitical issues of recovery are debated, old relict populations are not being aggressively protected through management and they continue to die off.

Recovery and Management

The goal of recovery is to reestablish species or enhance their ability to maintain self-perpetuating populations in native habitat, which may require both physical and biological habitat restoration. Many scientists believe recovery of bonytail and razorback sucker will take an aggressive and long-term commitment. Recovery efforts in the upper river are being intensified to restore adequate spring flows and develop nursery habitat. Stocking of bonytail and razorback sucker is being postponed until these habitat changes are made, and guidelines for stocking recreational species and possibly reducing their populations are being negotiated. Whether these actions will be sufficient to recover these fish is unknown.

While bonytail and razorback sucker are not being stocked in the upstream recovery program, they are being stocked farther downstream. A 10-year stocking program reintroduced razorback sucker into Arizona streams, but although nearly 15 million razorbacks were stocked between 1981 and 1990, the effort failed because most small suckers were believed to have been eaten by catfish and other non-native fishes. This emphasizes the need for predator removal or the stocking of larger fish.

Removal of non-native species is virtually impossible and sometimes undesirable. Larger bonytails and razorback suckers are being stocked by the Native Fish Work Group to attempt to maintain the Lake Mohave population by replacing the old population with young adults that exhibit the genetic characteristics of the remnant population. Bonytail and razorback suckers are being raised in isolated coves where other fish have been removed. Fish grow to about 30 cm (12 in) in length in a year and are then released into the reservoir. At this size, many should escape predation and could potentially survive 40 to 50 years.

Stocking is not an alternative to recovery, but if done properly, it can be used to maintain, expand, or reestablish long-lived endangered fish populations. Lake Mohave is not pristine habitat; however, maintenance of its population can help preserve genetic diversity, enhance species diversity in the reservoir, help ensure against catastrophic loss of hatchery brood stocks, and provide opportunities to study these fish in the wild.

Aggressive management of remaining populations is essential to provide the time to complete and test habitat restoration programs. If remnant populations are not saved, we stand to lose important pieces of a very complex ecological puzzle.

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Amphibian and Reptile Diversity on the Colorado Plateau

by
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The Colorado Plateau region is an area of high uplands, cut by the dramatic canyons of the Colorado River system in northern Arizona, northeastern New Mexico, eastern Utah, and western Colorado (Figure). Habitats within the region range from upland desert in the lower stretches of the Colorado River to small areas of alpine tundra on the highest peaks. The amphibian fauna is relatively small and dominated by species adapted to dry conditions such as toads (genus *Bufo*) and spadefoot toads (genus *Scaphiopus*). Reptile species are more numerous and varied, with the spiny lizards (*Sceloporus*), whiptail lizards (*Cnemidophorus*), and garter snakes (*Thamnophis*) well-represented. The reptiles and amphibians of the area have not been well-studied, although several species are known or suspected to have suffered recent declines.

As part of an overall project to assess the completeness of biological inventory information on National Park Service lands (Stohlgren and Quinn 1992), we compiled information from species lists, literature reports, and limited field work to prepare a preliminary data base of amphibian and reptile occurrence on 25 park areas in National Park Service lands on the Colorado Plateau.

Status and Trends

The quality and completeness of amphibian and reptile inventory information for Colorado Plateau parks vary. Grand Canyon National Park has received moderately thorough survey effort along the Colorado River corridor (Miller et al. 1982), but the canyon rim areas have received relatively little study. Parts of Glen Canyon National Recreation Area were surveyed by the University of Utah (Woodbury 1959), but this information is now 35 years old. Most other parks have had little or no thorough survey work and a few, such as Rainbow Bridge National Monument, have no inventory information at all. The information for many park species lists is based on large-scale range maps or unverified records. We found incorrect identifications or outdated taxonomy on about 10% of the species recorded. The generally poor and sometimes unreliable state of inventories in many parks echoes the results of Stohlgren and Quinn (1992) in a larger study.

Sixty-two reptile and 18 amphibian species are known from the Colorado Plateau as a whole. Most occur in one or more of the national park areas. The few species that apparently do not live in any parks, such as the mountain treefrog (*Hyla eximia*), Chiricahua leopard frog (*Rana chiricahuensis*), and narrow-headed garter snake (*Thamnophis rufipunctatus*), are primarily found in the area of the precipitous Mogollon Rim in north-central Arizona, which forms the southern boundary of the plateau. The most widespread species in the region include Woodhouse's toad (*Bufo woodhousii*; 20 areas), tiger salamander (*Ambystoma tigrinum*; 16 areas), eastern fence lizard (*Sceloporus undulatus*; 22 areas), tree lizard (*Urosaurus ornatus*; 21 areas), side-blotched lizard (*Uta stansburiana*; 22 areas), striped whipsnake (*Masticophis taeniatus*; 21 areas), pine snake (*Pituophis melanoleucus*; 22 areas), and western rattlesnake (*Crotalus viridis*; 22 areas). Some other species are much more limited; for example, the Jemez Mountains salamander (*Plethodon neomexicanus*) is known only from a small area of north-central New Mexico, including Bandelier National Monument. The painted turtle (*Chrysemys picta*) is apparently rare in the region and may have declined further; there have been no recent reports of this species from Glen Canyon National Recreation Area, where it formerly occurred. Other species, like the desert iguana (*Dipsosaurus dorsalis*) and the Gila monster (*Heloderma suspectum*), only have a small portion of their range on the Colorado Plateau (although they occur within the geographic boundaries of the area, some of these species are restricted to habitats not representative of the plateau, such as the Sonoran Desert).

Four of the 62 reptile species (7%) are listed as threatened or endangered by either individual states, the Department of the Interior, or both (Table). In contrast, 5 of the 18 amphibian species (27%) in the region are considered threatened or endangered. The high proportion of amphibians listed is due to several frog and toad species that have experienced serious population declines. One of these, the relict leopard frog (*Rana onca*) of southern Nevada, was thought extinct but has recently been rediscovered (D. Bradford, Environmental Protection Agency, Las Vegas, Nevada, personal communication). The western toad (*Bufo boreas*) has suffered drastic declines in other parts of its range (e.g., Carey 1993); its status on the Colorado Plateau is not known. The northern leopard frog (*Rana pipiens*), although not yet listed by state or federal governments, has disappeared from large areas of its range in western North America (Hayes and Jennings 1986). There are recent reports of healthy populations in a number of the perennial streams on the Colorado Plateau, but this species, in particular, needs further survey.

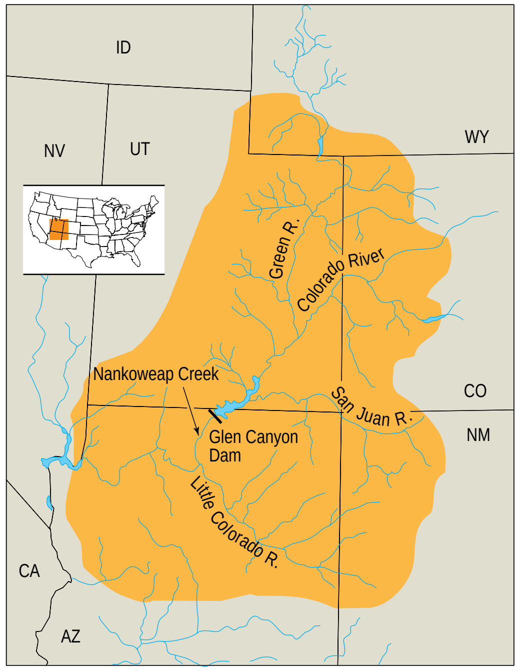

Figure. The Colorado Plateau region is cut by dramatic canyons of the Colorado River system.

Northern leopard frog (*Rana pipiens*).

Table. Threatened and endangered species of the Colorado Plateau. An X indicates that a species is listed as threatened or endangered by particular states or by the U.S. Department of the Interior.

Common name	Scientific name	Threatened or endangered status				
		AZ	CO	NM	UT	Federal
Amphibians						
Jemez Mountains salamander	*Plethodon neomexicanus*			X		
Western toad	*Bufo boreas*		X	X		
Chiricahua leopard frog	*Rana chiricahuensis*	X				
Relict leopard frog	*Rana onca*	X			X	X
Spotted frog	*Rana pretiosa*				X	
Reptiles						
Desert tortoise	*Gopherus agassizii*				X	X
Chuckwalla	*Sauromalus obesus*				X	
Gila monster	*Heloderma suspectum*			X	X	
Narrow-headed garter snake	*Thamnophis rufipunctatus*			X		

This ongoing project provides an assessment of our current knowledge, baseline information on distribution, and a starting point for intensive studies of rare and declining species. The development of an adequate inventory, coupled with long-term population studies of particular species of concern, forms the basis for informed protection and management of local natural communities.

Future Needs

In the arid Southwest, plant and animal communities depend on the same scarce water resources as human populations, agriculture, and industry. Amphibians and some reptiles, such as garter snakes, are directly dependent on free-flowing water and aquatic habitats. Amphibians are of further concern because of recent, unexplained losses in many areas (Barinaga 1990; Blaustein and Wake 1990). A thorough understanding of the present status, population trends, and requirements of native species is essential to avoid or lessen conflicts among competing natural resource demands.

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Wintering Bald Eagles Along the Colorado River Corridor

by
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The construction and operation of reservoirs have had a dramatic influence on wintering and migrant bald eagles (*Haliaeetus leucocephalus*; Southern 1963; Spencer 1976; Steenhof 1978; Stalmaster 1987). In contrast to reservoir-induced destruction of riverine habitat upon which many wintering bald eagles have traditionally relied, reservoirs may harbor, in some instances, new or alternative food sources (Spencer 1976; Jenkins 1992). In addition to hunting the shorelines and surface waters of reservoirs, eagles congregate below some dams in winter to feed on fish that are killed or stunned while passing through the turbines or to hunt in ice-free water (Steenhof 1978).

Commercial river guides on the Colorado River first noted winter bald eagle concentrations on the southern Colorado Plateau below Glen Canyon Dam at Nankoweap Creek in the early 1980's (Fig. 1). Before this, bald eagles were considered uncommon along the Colorado River in Grand Canyon National Park (Brown et al. 1987). A preliminary study by Brown et al. (1989) concluded that wintering bald eagles had increased in numbers, particularly below Glen Canyon Dam, because of a combination of regulated discharge of cold water from the dam and the introduction of rainbow trout (*Oncorhynchus mykiss*). Although trout were introduced by the National Park Service into tributaries of the Colorado River in the 1920's, it was not until after the dam was completed (1963) that trout numbers increased in the Colorado River. By 1988 the mouth of Nankoweap Creek had become a concentration

Fig. 1. The Colorado Plateau.

point for foraging because of the ease with which spawning trout could be obtained by eagles.

The concentrations of wintering and migrant bald eagles at Nankoweap are analogous to how eagles formerly concentrated at McDonald Creek in Glacier National Park, Montana (McClelland 1973). There, the introduction of non-native kokanee salmon (*Oncorhynchus nerka*) eventually attracted hundreds of migrant bald eagles (McClelland et al. 1982). The subsequent introduction of exotic zooplankton into Flathead Lake recently caused the collapse of this salmon population and ended the concentration of wintering eagles. In Grand Canyon National Park, it was felt that if the number of spawning trout remained high in the Colorado River tributaries, bald eagles might continue to concentrate there for food, as happened along McDonald Creek at Glacier National Park.

This article outlines the 1989-94 status of wintering bald eagles along the Colorado River corridor, from the Glen Canyon Dam through Grand Canyon National Park. We also discuss the trends of bald eagle numbers as determined from monitoring eagle and fish populations throughout the river corridor.

We determined the annual status of bald eagles from 1990 to 1994 by direct ground observations from the river bottom at the confluence of Nankoweap Creek and the Colorado River, and from aerial censusing flights from January through April.

Trends

Aerial Surveys

Wintering bald eagles were present each year along the Colorado River corridor from late fall (October-November) through early spring (March-April). During the 1990-91 aerial censusing surveys, peak numbers occurred in January and February, so aerial surveys in subsequent years were confined to December through March (Fig. 2). Eagles were observed on every flight, with numbers ranging from 2 (in March 1993) to 23 (in February 1991). Bald eagles were generally distributed evenly along the river corridor except in January and February, when conditions were suitable and rainbow trout were spawning in tributaries (Leibfried and Montgomery 1993). During these 2 months birds concentrated at the small tributaries.

Ground Surveys at Nankoweap

The trend of bald eagle numbers at Nankoweap Creek was for birds to closely parallel spawning trout numbers (Fig. 3). During 1990-91 we recorded the highest known bald eagle concentration at Nankoweap Creek with up to 26 eagles present on a peak day (Fig. 4). About 70-100 individuals were documented during the eagle concentration (when at least 10 eagles were present each day) from 8 February to 8 March 1990. The previous high of 18 wintering eagles was recorded at Nankoweap in February 1988 (Brown et al. 1989). The trend was for fewer numbers of trout and birds in following years (Fig 3). For example, in 1993 when spawning was extremely low in Nankoweap Creek, there were concomitantly low numbers of eagles. In 1994 spawning trout numbers were also low in the creek and few bald eagles were found in the area.

Other Areas of the Plateau

During 1992-94 when the numbers of wintering bald eagles along the Colorado River were low (Fig. 3), concentrations of bald eagles were reported at other locations on the southern Colorado Plateau. For example, in 1992, Bureau of Reclamation pilots (personal communication) noted eagle concentrations at the junction of the Green and Colorado rivers (Fig. 1). During 1993 the Arizona Game and Fish (personal communication) reported up to 20 eagles at Lake Mary, just east of Flagstaff, Arizona. These birds were feeding on some of the thousands of rainbow trout the agency had stocked into the lake during the winter. In 1994, another year of low bald eagle numbers along the Colorado River corridor, we received numerous reports from state and federal agency biologists of small eagle concentrations at elk and deer carcasses over the southern Colorado Plateau.

Status

The status of bald eagles along the Colorado River, especially in portions of Grand Canyon National Park and Glen Canyon National Recreation Area, has been improved by an increase in numbers of introduced rainbow trout. For example, at Nankoweap Creek, the trend went upward from a few birds starting in the mid-1980's to peak numbers in 1990-91. In following years (1992-94), poor rainbow trout spawning resulted in low numbers of bald eagles in this region. Creek morphology and flow conditions varied among years and influenced the availability of trout, and thus eagle numbers.

Bald eagles at Nankoweap, however, can be the largest such concentration in the southwestern United States. The 70-100 individual eagles recorded during 1990 represent what is believed to be one-fourth of the entire population of bald eagles wintering to the south of the Grand Canyon (in Arizona and northern Mexico). We expect that wintering eagles will continue to

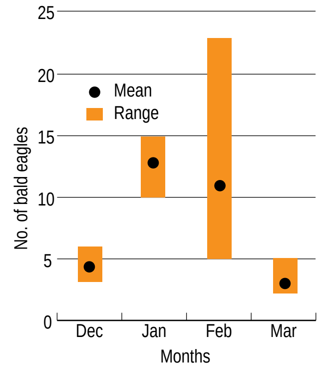

Fig. 2. Average number of bald eagles detected each month (1990-94) during aerial surveys along the Colorado River from the junction of the Little Colorado River north to Glen Canyon Dam.

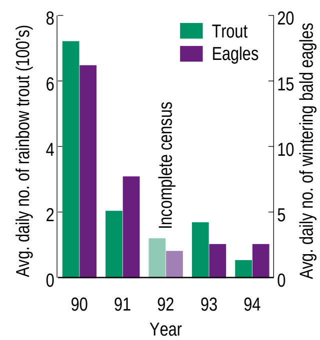

Fig. 3. Average daily numbers of rainbow trout and bald eagles at Nankoweap Creek in Grand Canyon National Park, AZ, 1990-94.

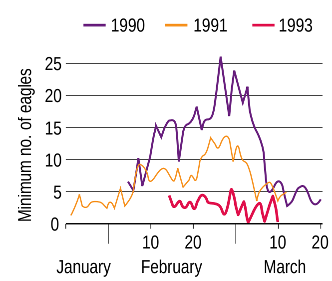

Fig. 4. Bald eagle abundance at Nankoweap in 1990, 91, and 93.

frequent this region if annual spawning trout are present.

Bald eagle counts along the Colorado River corridor during the winters of 1990-94 mirrored the bald eagle numbers at Nankoweap Creek. Their numbers peaked during late February and early March and varied greatly among years. Higher concentrations of bald eagles noted in other areas of the southern Colorado Plateau, when lower numbers were recorded along the Colorado River, suggest widespread eagle movements over the region. Bald eagles appear to concentrate in areas that have the most abundant and available food resources, and these locations change annually.

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Mexican Spotted Owls in Canyonlands of the Colorado Plateau

by
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In response to perceived threats to critical nesting habitat and lack of adequate protective regulations, the Mexican spotted owl (*Strix occidentalis lucida*) was officially listed as a threatened species under the Endangered Species Act in 1993 (Federal Register 1993). Limited information is available on the distribution of Mexican spotted owls inhabiting arid canyonlands throughout the southwestern United States (Ganey and Balda 1989). Though widely distributed, the Mexican spotted owl apparently occurs in isolated populations restricted to habitat islands (Fig. 1). Here I report findings from spotted owl surveys conducted throughout the northwest portion of the Colorado Plateau in Utah.

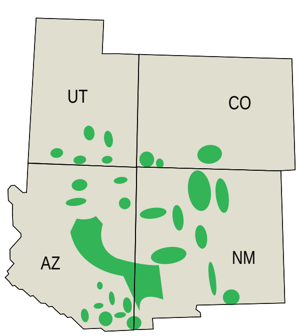

Fig. 1. Distribution of Mexican spotted owls in the southwestern United States.

The Colorado Plateau Physiographic Province consists of extensive sandstone canyons interspersed by eroded valleys, upwarped plateaus, and isolated mountain ranges (Thornbury 1965). Prolonged erosional dissection produced a maze of complex watersheds within the Colorado Plateau region (Youngblood and Mauk 1985). Agency lands encompassed by the Colorado Plateau include extensive U.S. Department of Agriculture national forests and U.S. Department of the Interior Bureau of Land Management (BLM) areas, seven National Park Service national parks, two national recreation areas, several national monuments, and state-administered lands, all in the Four Corners region (Arizona, New Mexico, Colorado, and Utah) of the southwestern United States.

These lands may function as biological refugia, providing dispersal corridors and habitat islands joining occupied and potentially suitable spotted owl habitat. In the Four Corners region, spotted owls are associated with rocky canyon terrain (i.e., canyonlands) and could be negatively affected by such activities as timber harvesting, mining, and recreation (Ganey 1988). Long-term study of spotted owl distribution and habitat use is necessary to provide information on the potential effects of human activities and to develop ecologically based conservation plans (Gutiérrez 1989).

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Mexican spotted owls (*Strix occidentalis lucida*) roosted in canyonlands in southern Utah.

Surveys

Information on Mexican spotted owl distribution within canyonlands of the Colorado Plateau was gathered by using published species accounts and conducting field surveys.

During the field surveys, individuals and pairs of owls were located by imitating their calls with the human voice or using taped broadcasts of their calls to elicit a response from the owls (Forsman 1983). The surveys were conducted during each breeding season (1 April-31 August) from 1989 through 1993. Target areas were visited four times during the breeding season to search for owls. Spotted owl callers ("hooters") conducted searches by "hooting" at stations located on night-time survey routes placed within search areas. Hooters conducted daytime visits to sites where spotted owls were heard at night in order to find nests and count young.

Historical Records

Historical records of Mexican spotted owls on the Colorado Plateau date back to the 1920's (McDonald et al. 1990). The earliest record in the canyonlands was from Zion National Park in June 1928. A single owl was reported in August 1957, in Davis Gulch, a tributary of the Escalante River in southern Utah. Three birds were seen in July 1958, in a small side canyon of Glen Canyon National Recreation Area, and another was observed at the mouth of the Escalante River. The most northerly occurrence of a spotted owl on the Colorado Plateau was recorded in September 1958, in the Book Cliff Mountains. Spotted owls have been observed occasionally since the early 1970's throughout the canyonlands of southern Utah. Kertell (1977) detected spotted owls at six locations in Zion National Park in the early 1970's. The species accounts suggest that spotted owls were widely dispersed throughout the canyonlands of the Colorado Plateau, especially in deeply eroded sandstone gorges.

Field Survey Results

About 202,500 ha (500,000 acres) were surveyed from 1990 to 1993 on U.S. Forest Service lands, and more than 483 km (300 mi) of BLM canyons were surveyed from 1991 to 1993. Surveys were also conducted in portions of Grand Canyon, Capitol Reef, Canyonlands, and Zion national parks, as well as Natural Bridges and Navajo national monuments. Seventy-six spotted owls (26 pairs and 24 single adults) were detected at 50 locations: 6 on U.S. Forest Service lands, 12 on BLM lands, 1 on state lands, and 31 on National Park Service lands (Fig. 2).

Groups or subpopulations of owls were distributed among several landscape areas spread across the northwest portion of the Colorado Plateau including the greater Zion National Park area; the greater Capitol Reef area; the Dirty Devil River watershed; Canyonlands National Park; and near Elk Ridge and Dark Canyon on the Manti LaSal National Forest.

Mexican spotted owls were widely distributed and appeared coincident with canyon habitat. Canyon habitats on the Colorado Plateau are discontinuous and reflect the naturally fragmented topographic conditions of the plateau region. This patchy landscape could explain the patchy locations of surveyed spotted owls. A study conducted in Zion National Park found owls nesting and roosting in humid, narrow canyons with dense understories (Rinkevich 1991). Since many owls on the Colorado Plateau were found in similar habitat, the owls may be selecting these canyons because of their unique habitat features: large cliffs that provide escape cover to avoid predation, shaded roost sites to avoid high summer temperatures, patches of forest vegetation, and availability of suitable prey.

Relatively few owls were found in the canyonlands area compared with forest sites in Arizona and New Mexico; thus, canyonland owl sites may need special protection. Further surveys should be conducted across USDI lands to more accurately assess distribution and habitat of spotted owls.

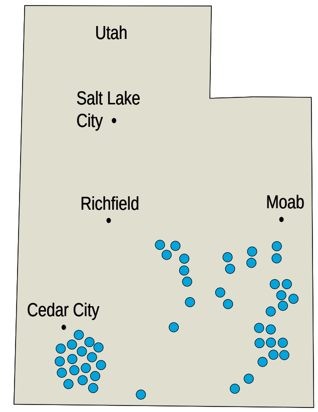

Fig. 2. Distribution of Mexican spotted owls in canyonlands of southern Utah, representing the northwest portion of the Colorado Plateau Physiographic Province.

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Bighorn Sheep in the Rocky Mountain National Parks

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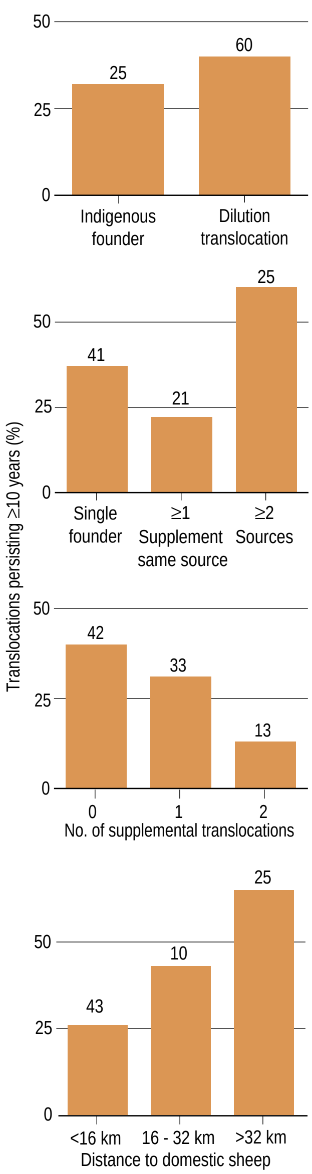

Figure. Factors contributing to persistence of bighorn sheep translocations. The numbers above the bars refer to the total number, or sample size, of populations in that category.

Current numbers of bighorn sheep (*Ovis canadensis*) are probably only 2%-8% of their numbers at the time of European settlement. The Rocky Mountain subspecies (*O.c. canadensis*) and the California subspecies (*O.c. californiana*) combined may have numbered roughly 1 million, and the desert subspecies (*O.c. nelsoni*) of the southwestern United States and Mexico also likely numbered about 1 million (Buechner 1960; Wishart 1978; Bleich et al. 1990). Unregulated harvesting, habitat destruction, overgrazing of rangelands, and diseases contracted from domestic livestock all contributed to drastic declines, the most drastic occurring from about 1870 through 1950.

Bighorn exist mostly in small, isolated populations within their former vast range. Thorne et al. (1985) found that 64% of 166 populations in the western United States contained fewer than 100 individuals. In Arizona, 88% of the populations (52 of 59) contained fewer than 100 individuals (Krausman and Leopold 1986).

Small populations of animals may be at higher risk of extirpation (Gilpin and Soulé 1986). The negative effects of small population size on bighorn were documented by Berger (1990), who reported that no indigenous populations of fewer than 50 animals survived for 5 decades, whereas all populations numbering more than 100 animals survived for the same period. Berger's (1990) published review did not consider national park populations of bighorns.

Restoration of bighorn sheep has been pursued actively by many state and federal agencies since the 1940's, although these efforts have met with only limited success, and most of the historical range of bighorns remains unoccupied. Human encroachments near bighorn populations are severe enough in some areas that the peninsular population of desert bighorns in California has been proposed for federal threatened status.

This article reviews the status of bighorns of three subspecies, the desert, Rocky Mountain, and badlands (*O.c. audobonii*), in 17 national park service units in the Rocky Mountains. Factors contributing to the success of 115 transplants of bighorn sheep that occurred over the past five decades in six Rocky Mountain states are also reviewed.

Information on the status and restoration of bighorns in the National Park Service units came from published accounts, university theses, unpublished park records, and a questionnaire mailed to state wildlife agencies and land managers in Colorado, Montana, North and South Dakota, Wyoming, and Utah. Only populations that had been translocated at least 10 years were included in the analysis.

Status in National Parks

Eighteen national park units historically contained populations of bighorn sheep. Native populations of bighorns were extirpated in all but five of the units, and populations were greatly reduced in four of these five. Only the Yellowstone's ranges remained fully occupied by bighorn during this period. Native bighorns survived but were greatly reduced in Grand Teton, Canyonlands, Glacier, and Rocky Mountain parks. The Badlands subspecies was eliminated about 1921. This subspecies originally inhabited clay badlands and low river breaks in the Dakotas, including Badlands and Theodore Roosevelt national parks.

Restoration efforts of bighorns into park units began in the late 1940's in 11 national park units where bighorns had been extirpated. Augmentations or translocations of additional subpopulations occurred in three of five national park units where bighorns had not been completely extirpated. Bighorn ranges are now considered fully or very fully occupied in two of these units, Rocky Mountain and Canyonlands parks.

Restoration of bighorns into other national park units has had only limited success. Ten national park units support persisting populations (numbering 100 or more sheep the previous 4 years); five park units have populations estimated to exceed 500 animals; and five other park units have populations of 100-200. Five other park units have fewer than 100 individuals, and two of these units support populations on the verge of extirpation (only 6-14 animals).

Translocations

Only 39% of 115 bighorn transplants in six Rocky Mountain states were rated as persisting (Figure). Sixty-four percent of transplants located more than 32 km (20 mi) from domestic sheep were persistent, but only 44% of those bighorn populations located 16 to 32 km from domestic sheep were persistent (Figure). In addition, nearly twice as many transplanted populations that were sedentary failed than populations that migrated to separate winter and summer ranges. Most translocated populations do not regain the historical migration patterns of the extirpated native population; instead, many spend the summer and winter on the same small ranges.

Limited evidence suggests that threshold population size or genetic diversity is related to persistence of bighorn transplants. Transplant persistence and genetic diversity were positively correlated to initial founder group size (the number of animals moved in the initial translocation), to multiple (versus single) source populations represented in the initial founder group, to the use of native populations as a source, and to sheep interaction with other nearby subpopulations (Fitzsimmons 1992).

Implications

Restorations into national park units are, as yet, incomplete. At present, bighorn sheep occur in small, widely scattered populations, with the smallest groups (fewer than 50 animals) seemingly at highest risk of extirpation. Thus, to achieve larger, more secure populations, restoration is necessary. To improve the chances for successful translocations, greater care must be taken; only about one-third of past translocations were persistent. Our analysis suggests that a population distant from domestic sheep improved the probability of its persistence more than any other factor. Larger founder sizes, multiple versus single sources of founders, native source groups, interactions with nearby subpopulations, and migratory tendencies also may contribute to continued persistence of translocated sheep and should be considered during translocations. Habitat suitability assessments before translocations would also probably contribute to sheep restoration success and are recommended as an integral part of any restoration.

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Desert Bighorn Sheep

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Desert bighorn sheep (*Ovis canadensis* spp.) are subspecies of concern in the continental United States. Populations declined drastically with European colonization of the American Southwest beginning in the 1500's (Buechner 1960). At present, desert bighorn numbers are extremely low, although the overall population trend has increased since 1960.

Desert bighorn are considered good indicators of land health because the species is sensitive to many human-induced environmental problems (McCutchen 1981). In addition to their aesthetic value, desert bighorn are considered desirable animals by hunters.

The Rocky Mountain and California races of bighorn occupy the cooler western and northwestern regions of the United States. In contrast, the desert sheep races are indigenous to the hot desert ecosystems of the Southwest.

Population Trends

The number of desert bighorn in North America in pristine times is unknown but most likely was in the tens of thousands (Buechner 1960). Seton (1929) estimated the pre-Columbian numbers of all subspecies of bighorn in North America (United States, Canada, and Mexico) at 1.5-2 million. By 1960, however, the overall bighorn population in the United States, including desert bighorns, had dwindled to 15,000-18,200 (Buechner 1960). Buechner documented major declines from the 1850's to the early 1900's. These declines were attributed to excessive hunting; competition and diseases from domestic livestock, particularly domestic sheep; usurpation of watering areas and critical range by human activities; and human-induced habitat changes (Buechner 1960; Graham 1980; McCutchen 1981).

These declines were followed by a period of population stabilization that Buechner ascribed to conservation measures. The decline of desert bighorn probably mirrored the pattern of decline of the overall bighorn population. Desert bighorn population trends have been upward since the 1960's when Buechner (1960) estimated their population at 6,700-8,100. In

Courtesy H.E. McCutchen

Desert bighorn sheep (*Ovis canadensis nelsoni*).

1980 desert bighorn populations were estimated at 8,415-9,040 (Wishart 1978). Weaver (1985) conducted a state-by-state survey a few years later and estimated the U.S. desert bighorn population at 15,980. The 1993 estimate of the population is 18,965-19,040 (Table).

Table. Status and trends of desert bighorn sheep in the United States 1960-93. Estimate for 1960 by Buechner (1960). Estimates for 1993 from state wildlife agency status reports presented to the Desert Bighorn Council, Moab, Utah, 1994.

	Population estimate by year	
State	1960	1993
Arizona	3,000-3,500	6,000
California	2,140-2,450	4,300-4,325*
Colorado	0	475
Nevada	1,500-2,000	5,294
New Mexico	400-500	295
Texas	25	401
Utah	Remnant	2,200-2,250
Total	**7,065-8,475**	**18,965-19,040**

*In California, Nelson's bighorn (*Ovis canadensis nelsoni*) population trends are upward. Peninsular bighorn (*O.c. cremnobates*) populations are lower and are of concern.

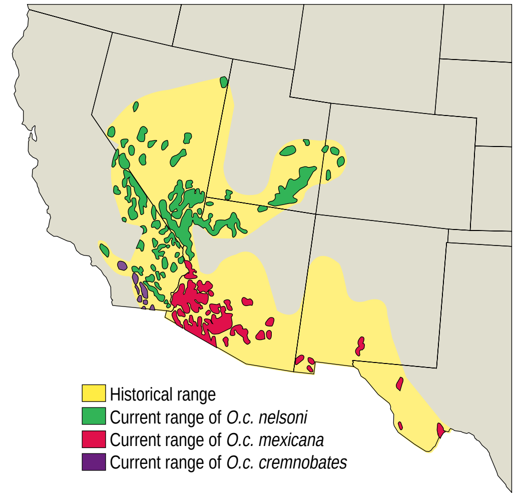

Figure. Historical range and current distribution of the three subspecies of desert bighorn in the United States (redrawn from Trefethen 1975 and Weaver 1985).

Subspecies

Cowan (1940) used morphological characters and measurements to identify three subspecies of desert bighorns (*O.c. nelsoni*, *O.c. mexicana*, and *O.c. cremnobates*) occurring in the United States. A recent reevaluation of mountain sheep races in the United States, however, suggested significant differences between the northern and southern (desert) sheep (Ramey 1993). Differences among the three desert bighorn races, however, did not support separate subspecies designations.

The distribution of desert bighorn races is uncertain, although the distribution maps of Trefethen (1975) and Weaver (1985) are accepted by mountain sheep biologists (Figure).

Status and Trends by State

Arizona

Historically, desert bighorn occurred on all mountain ranges and plateau slopes in the southern, northern, and western sections of Arizona (Russo 1956). In spite of early protection (beginning in the 1880's), researchers believed that bighorn populations declined until the 1950's (Russo 1956).

Arizona began a limited hunting program in 1953 and reintroduction programs in 1958. The Arizona Game and Fish Department conducts annual helicopter surveys. Buechner (1960) estimated the 1960 population at about 3,000-3,500. In 1993 the population had increased to an estimated 6,000 (R. Lee, Arizona Game and Fish Department, unpublished data).

California

Desert bighorn occupied desert mountains in southeast California in historical times. California protected bighorn in 1883, and by 1960 Buechner (1960) estimated the population at about 2,150-2,450 (1,800-2,100 *O.c. nelsoni* and 350 *O.c. cremnobates*). The state began transplanting in 1971 and permitted hunting beginning in 1986 (Bleich et al. 1990). In 1993 the populations were estimated at 4,300-4,325, with the breeds occupying about 50 mountain ranges (S. Torres, California Department of Fish and Game, unpublished data).

The less common peninsular bighorn (*O.c. cremnobates*) occurs in the desert mountains of southeast California from Palm Springs south to the Mexican border. From 1977 to 1993 this population declined from an estimated 1,171 to 400-425 individuals because of excessive lamb mortality (Weaver 1989; S. Torres, California Department of Fish and Game, unpublished data). In 1992 the U.S. Fish and Wildlife Service proposed listing the peninsular bighorn as endangered (Torres et al. 1993). This subspecies also occurs southward into Mexico; populations there are larger. One survey estimated a population of 780-1,170 adult bighorn in northern Baja California, Mexico (DeForge et al. 1993).

Colorado

There is no scientific evidence that desert bighorn occurred historically in Colorado, although there is habitat in the state contiguous with desert bighorn habitat in Utah. Thus, desert bighorn probably occurred in the state, and became extirpated before subspecies' determinations could be made.

The Colorado Division of Wildlife began transplanting desert bighorn in 1979. By 1993 populations containing approximately 475 bighorn had been established from the release of animals originally from Arizona and Nevada (Wolfe 1990; V. Graham, Colorado Division of Wildlife, unpublished data).

Nevada

Desert bighorn (*O.c. nelsoni*) historically occupied the central and southern portions of Nevada (McQuivey 1978). Hunting the animals was prohibited from 1901 to 1952. Transplanting programs have been successful: between 1968 and 1988 more than 800 desert bighorn were transplanted. From these animals, 21 transplanted herds have been established (Delaney 1989).

Buechner (1960) estimated the Nevada population at 1,500-2,000 in 1960. The state began annual population trend counts in 1969. In 1993 the population was estimated at 5,294 animals, occupying 45 mountain ranges (P. Cummings, Nevada Division of Wildlife, unpublished data).

New Mexico

Although desert bighorns (*O.c. mexicana*) historically occupied mountain ranges and canyons in the southern part of New Mexico, by 1930 the animals were restricted to only four mountain ranges, and by the late 1940's were found in only two (Weaver 1985).

In 1972 the state constructed the 300-ha (741 acres) Red Rock propagating enclosure and added brood stock. Transplants from the captive herd were subsequently made into the Big Hatchet, Peloncillo, and Alamo Hueco mountains (Sandoval 1979).

The San Andres Mountain population was formerly the state's largest, but declined from 200 to fewer than 25 by 1991 (Clark and Jessup 1992) because of psoroptic scabies (*Psoroptes* spp.).

Buechner estimated the New Mexican population at 400-500 in 1960. In 1993 the estimated population was 295, of which 100 were at Red Rock (A. Fisher, New Mexico Department of Game and Fish, unpublished data).

Texas

Desert bighorn (*O.c. mexicana*) appear to have occupied all the mountains in southwest Texas west of the Pecos River (Carson 1941). In 1880 the population was estimated at 1,500 animals (Kilpatric 1982); some populations still existed in the late 1930's. By the mid-1950's all bighorns had become extirpated except for a small herd of 25; excessive hunting and competition with domestic livestock are believed to have been major factors in the final decline (Buechner 1960).

In 1957 the Texas Game and Fish Department began a highly successful captive breeding and release program. By 1993 the free-ranging population was estimated at 310; 91 other sheep were in captivity (G. Calkins, Texas Parks and Wildlife Department, unpublished data).

Utah

Historically, desert bighorn (*O.c. nelsoni*) occupied canyons and ranges in southern and eastern Utah. Significant population declines occurred in the 1870's (Buechner 1960), and the state did not permit hunting of bighorn from 1899 to 1967.

In 1967 limited hunting began, and in 1973 the state started an active transplant program. Between 1973 and 1990, over 250 desert bighorn sheep were transplanted, establishing at least nine populations that augment four additional areas containing native populations (Cresto et al. 1990).

Buechner (1960) believed that only remnant populations persisted in the state. Utah, which has conducted aerial trend counts on bighorn since 1969 (Cresto et al. 1990), documented increasing populations statewide. Individual populations, however, have exhibited large increases and sudden declines. In 1993 the desert bighorn population was estimated at 2,200-2,250 (N. McKee and J. Karpowitz, Utah Division of Wildlife, unpublished data).

Future of Desert Bighorn

Since 1960 bighorn have increased in numbers, but their population levels are still low when compared with the estimates of pre-European numbers and the amount of available unoccupied habitat. The number of sheep in individual populations has fluctuated greatly. Population monitoring and efforts to restore desert bighorn must continue to ensure viable future populations.

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