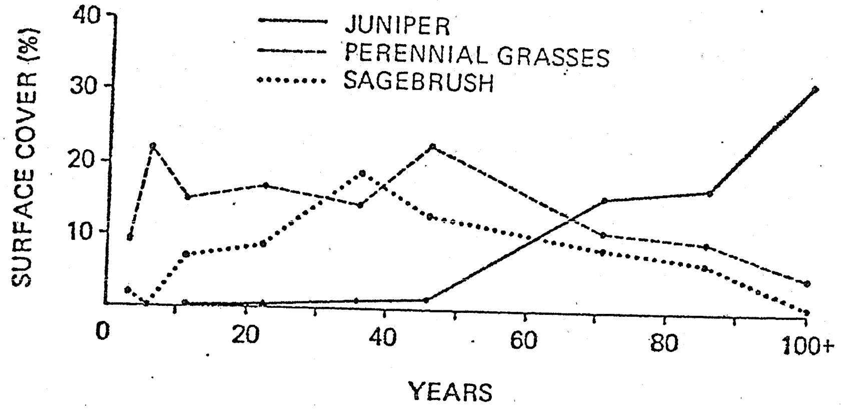

Figure 2. Change in vegetation with time following burning.

palatable species being bitterbrush, curlleaf mountain mahogany (*C. ledifolius*), true mountain mahogany (*C. montanus*), and cliffrose. Other studies also emphasize the importance of a variety of species, including shrubs, grasses, and forbs, in the diet of deer (Smith and Doell 1968, Wood et al. 1970, Trout and Thiessen 1973, Leach 1956, Dealy 1971, Wight and Fisser 1968, Reynolds 1964). In southeastern Utah, where sagebrush, juniper, and pinyon were the only species present, stomach analyses revealed that sagebrush constituted as much as 80 percent of the deer diet in October (Julander 1952). Thus deer were making heavy inroads on their winter forage supply as early as late October.

The report by Kufeld et al. (1973) revealed that during winter, shrubs and trees average 74 percent of the mule deer diet, forbs 15 percent, grasses, sedges, and rushes 11 percent. However, the percentage of grasses and grasslike plants in the winter diet ranged from 0 to 53 percent among the 99 selected studies. During spring, the average consumption of forbs rose to 25 percent of the diet, grasses to 26 percent, and woody species dropped to 49 percent. Again, use of grasses varied greatly, ranging from 4 to 64 percent of the diet for the studies summarized. During summer, the percentage of shrubs and trees in the diet remained at an average 49 percent, forbs increased to an average 48 percent, and grasses dropped to 3 percent. The composition of forbs varied from 3 to 77 percent among the various studies; grasses and grasslike plants ranged from 0 to 22 percent.

Mule deer tend to seek the shelter of trees during heavy winter storms, not unlike the white-tailed deer (*Odocoileus virginianus*) observed by Gladfelter (1966) in Idaho. He found that activity increased before storms or high winds, but decreased during storms. The deer chose paths of least resistance and avoided snowdrifts that accumulated in open areas and on some ridges. In winter, those deer bedded down much earlier than in summer, usually under a tree or other heavy cover. Increased length of bedding was accompanied by low temperatures and high humidity.

Published information on use of pinyon-juniper woodland by elk (*Cervus canadensis*) is scarce compared to that for deer. Wight and Fisser (1968) reported some use by elk of the pinyon-juniper woodland in northwest Wyoming as a source of food, shelter, and cover; in addition to extensive use of the area by deer as a winter and spring range. Kufeld (1973) also reported that juniper is moderately palatable to elk in winter and is slightly used by these animals in fall and spring. In southern Colorado, juniper comprised 8 percent of the diet of elk in spring, 4 percent in autumn and winter, and less than 1 percent in summer (Hansen and Reid 1975).

In Utah, elk make considerable use of the pinyon-juniper type in winter, especially where these trees are associated with mountain mahogany (*Cercocarpus ledifolius*) and other palatable shrubs (Homer Stapley, Utah State Dep. of Natural Resources, Salt Lake City, personal communication). A small herd on the Indian Peaks area in southwestern Utah spends considerable time in pinyon-juniper year round. The capacity of this area is limited to a herd of about 20 to 30 head.

On the Dutton Range east of Panguitch, the pinyon-juniper is not particularly good summer range for elk; so they spend most of the summer in lower canyon bottoms where there are small meadows (Floyd Coles, Utah State Dept. of Natural Resources, Cedar City, Utah, personal communication). In September as the rut begins, elk move higher up the mountain slopes and often winter on south-facing slopes to 9,000 feet elevation. Pinyon-juniper trees are intermixed with mahogany and other shrubby species on the intermediate slopes. As elsewhere, the trees serve more as cover and shelter for elk than as food. Elk usually are found where there is a mixture of shrub species upon which they browse in winter.

From 1970 to 1974, elk use in winter averaged 20 days per acre for six different pinyon-juniper areas on the Manti-LaSal National Forest (Coy Jemmett, U. S. Forest Service, Price, Utah, personal communication). Average use was highest on

Biddlecomb Ridge (43 elk days per acre) and South Trail Mountain (42 elk days per acre). Heaviest use on South Trail Mountain was 117 elk days use per acre the winter of 1970-71. Heaviest use on Biddlecomb Ridge was 80 elk days use per acre the winter of 1973-74.

Desert bighorn sheep (*Ovis nelsoni*) utilize pinyon-juniper range in rough areas otherwise suitable for bighorn habitat. Yoakum (1971) quoted Buechner to the effect that bighorns inhabited most of the lands in the Southwest when the white man first came to North America. Hunting, competition with domestic livestock, and decrease of available habitat contributed to the drastic reduction of these animals from the 1850s to the 1950s.

Wilson (1968) reported on the distribution of desert bighorn sheep in the White Canyon area of southeastern Utah where a part of their range is occupied by pinyon-juniper woodland. Between 124 and 144 bighorns in scattered bunches in the study area made up the total population of these animals. Adult rams tend to utilize the higher, more remote, rocky areas, but ewes, lambs, and immature rams 1 to 3 years old use steep, talus slopes, and lower mesas and canyons. High rims and buttes afford most protection for the animals. On Wingate Mesa, which is utilized by rams in summer, use increases proportionately as tree density thins out toward the southwest end of the mesa. Over the entire study area, lack of available free water was the most limiting factor to the bighorn populations. But internal parasites and competition for forage and water by deer and cattle in certain areas also limited population growth.

A herd of about 130 buffalo (*Bison bison*) roams through the pinyon-juniper woodland in the Henry Mountain area of southern Utah (Homer Stapley, Utah State Department of Natural Resources, Salt Lake City). When disturbed by man, the herd heads into the pinyon-juniper woodland where trees offer protection from view. These animals forage mostly on understory grasses.

The pronghorn antelope (*Antilocapra americana*) also uses the pygmy forest to some extent. The animal is found mostly on the fringes of the type where the pygmy forest intergrades with more open vegetation. A feeding trial with fenced antelope showed these animals preferred big sagebrush, black sagebrush, and juniper in that order over 13 other species. Single-leaf pinyon was fifth in order of preference, behind cliffrose (Smith et al. 1965).

Wild horses use the pinyon-juniper woodland extensively in various parts of the West; the Cedar City District of the Bureau of Land Management is typical. A total of 256 horses were counted from the air in January 1975, about 80 percent of the total number of horses in that district (Stanley Adams, Bureau of Land Management, Cedar City, Utah). The average annual increase is estimated to be about 18 percent, but varies from 0 to 42 percent among different bands. Most bands are small, 2 to 11 head, with a dominant stallion in each. Some bands were found that consist of 5 to 6 stallions only. Apparently, these animals were driven away from other bands by dominant stallions. Most colts, which are born from March through October, come in May. Horses use south-facing slopes in winter and often eat such shrubs as cliffrose, bitterbrush, sagebrush, rabbitbrush, and oak leaves in fall. Certain rehabilitated areas that have been seeded to grasses are used heavily by horses year round. One of these, approximately 1,200 acres in size (Bible Springs), is well utilized by about 55 horses when cattle enter the area in late spring. This usage causes great anxiety among stockmen who run permitted cattle on the area.

Predators

Rasmussen (1941) reported that the mountain lion (*Felix concolor*) was a major predator on the Kaibab Forest deer herd and that as many as two deer per week might be taken per lion. He reported further that the coyote (*Canis latrans*) was a factor in deer predation and that coyote numbers on the Kaibab Plateau during winter were two or three times greater than in summer. Rasmussen is quoted as follows: ". . . many dead deer were found and a great number had not been disturbed by carnivores, while numerous others showed definite evidence of having been killed by coyotes. A preference for meat of its own killing or freshly killed meat was clearly indicated . . ." He stated further that small rodents were the coyote's main year-long food.

Rasmussen (1941) reported that bobcats (*Lynx rufus*) were found wherever cliffs and ledges were present on the Kaibab pinyon-juniper woodland and that their food consisted of small rodents, some birds, and an occasional fawn.

A study by Beale and Smith (1973) on antelope herd productivity on the Desert Experimental Range in southwestern Utah showed that 61.5 percent of all mortality among antelope fawns, excluding fawns that were abandoned, was due to bobcats. Most kills occurred in a general area where dry washes were present and juniper trees growing in the washes formed stringers leading out into the valley. Although antelope were confined to some extent, it is doubtful that fences were a direct factor in fawn mortality.

The badger (*Taxidea taxus*) and weasel (*Mustela frenata*) inhabit certain areas where pinyon and junipers grow and where small mammals are available as prey.

Porcupines, Rabbits, and Small Mammals

Porcupines (*Erethizon epixanthum*) can be permanent residents in the pinyon-juniper ecosystem in such areas as Mesa Verde National Park where the mesa ranges from 6,500 to 8,500 ft. in elevation. The rock cliffs and talus slopes in relatively inaccessible canyons on the edges of the mesa provide ample den sites for porcupines and offer protection from predators (Spencer 1964). Spencer used dendrochronological techniques on pinyon pine in Mesa Verde National Park to determine the abundance of porcupines over the past several centuries. Four of the population eruptions in the last 120 years were centered in 1845, 1885, 1905, and 1935. The 1905 eruption covered a period of about 12 years, whereas evidence of the other three eruptions covered about 20 years. From scars on a few older trees, he determined that four earlier eruptions had centered in the years 1716, 1745, 1785, and 1815.

The range of desert cottontail rabbits (*Silvilagus audubonii*) extends over 60 million acres of pinyon-juniper woodland in the Rocky Mountain and Intermountain regions (Kundaeli and Reynolds 1972). In undisturbed pinyon-juniper areas, use by cottontails was greatest where vegetation composition averaged 85 trees, 85 shrubs, and 270 pounds of grass per acre. However, it was determined that if sufficient shrubs are present the habitat will be occupied by cottontails whether living trees are present or not. Cottontail use grew as shrubs increased from 53 to 85 per acre, but decreased as shrubs increased from 85 to 125 per acre. Tree density above 160 trees per acre suppresses shrub abundance. In tree-removal programs, cottontail use was depressed unless a mixture of 70 to 90 downed trees and shrubs remained. Where shrubs alone were that numerous, the habitat was not improved by leaving downed trees. A range of 150 to 320 pounds per acre of herbaceous vegetation didn't seem to affect cottontail use. Black-tailed jackrabbits (*Lepus californicus*) are scattered throughout the pinyon-juniper ecosystem, and the white-tailed jackrabbit (*Lepus townsendii*) occurs in some areas. In years of heavy snows, jackrabbits often concentrate in the pinyon-juniper woodland.

A study of small mammals in the pinyon-juniper woodland of westcentral Utah showed that of 13 species taken, deer mice (*Permyscus maniculatus*) comprised 83 percent of the catch (Baker and Frischknecht 1973). Other species included Great Basin pocket mice (*Perognathus parvus*); long-tailed voles (*Microtus longicaudus*); western harvest mice (*Reithrodontomys megalotis*); chisel-toothed kangaroo rats (*Dipodomys microps*); Nuttall's cottontails (*Sylvilagus nuttallii*); sagebrush voles (*Lagurus curtatus*); desert woodrats (*Neotoma lepida*); pinyon mice (*Peromyscus truei*); least chipmunks (*Eutamias minimus*); cliff chipmunks (*Eutamias dorsalis*); rock squirrels (*Spermophilus variegatus*); and northern grasshopper mice (*Onychomys leucogaster*). In another study of small mammals in nearby sagebrush-grass communities, eight of these same species were taken; the two species of chipmunks, sagebrush vole and rock squirrel, were not taken; and the traps used (Museum Special) were not adapted to taking cottontails (Black and Frischknecht 1971). In this general area, mainly above 6,500 feet, long-tailed voles in a peak population year girdled and killed big sagebrush and other shrub species on open areas adjacent to pinyon-juniper woodland (Frischknecht and Baker 1972).

The pinyon-juniper woodland on the LaSal Mountains in southeastern Utah is the habitat of the pinyon mouse and three kinds of woodrats: Mexican woodrat (*Neotoma mexicana*), white-throated woodrat (*N. albigula*), and the bushy-tailed woodrat (*N. cinerea*) (Bradley 1971). Other species found on the lower edge of the community are the Colorado chipmunk (*Eutamius quadrivittatus*), the apache mouse (*Perognathus apache*), ord's kangaroo rat, canyon mouse (*Peromyscus crinitus*), the western harvest mouse, the northern grasshopper mouse, the white-tailed antelope squirrel (*Ammospermophilus leucurus*), the rock squirrel, the Zuni prairie dog (*Cynomys gunnisoni*), and the desert cottontail. The least chipmunk and Nuttall's cottontail were found in the upper part of the pinyon-juniper community and as high as the spruce-fir community. The deer mouse is found within every vegetation zone of the LaSal Mountains and the porcupine occurs in all communities except the alpine tundra.

During an 8-year study of small mammals on pinyon-juniper deer range in western Colorado, deer mice and pinyon mice were the most abundant of the seven species taken (Shepherd 1972). The author concluded that these mice were probably more beneficial than detrimental on a pinyon-juniper-deer range because they consume large quantities of insects. Population highs for pinyon mice may be more than 25 times population lows. Combined populations of pinyon mice and deer mice were estimated to vary from 0 to 8.15 animals per acre. During population highs, mouse population density would be about 10 animals per acre. Other species trapped during the 8 years of study were the brush mouse (*Peromyscus boylii rowleyi*); the western harvest mouse (*Reithrodontomys megalotis aztecus*); the Mexican vole (*Microtus mexicanus mogollonensis*); the Colorado chipmunk (*Eutamius quadrivittatus hopiensis*); and the least chipmunk (*Eutamius minimus operarius*).

The desert woodrat (*Neotoma lepida*), identified with pinyon-juniper woodland and active year round, has received special study by Rasmussen (1941) and Stones (1960). Woodrat nests are located about the stump of a pinyon or juniper. Rasmussen reported that one of these nests contained approximately 10 bushels of materials, 85 percent of which was sticks and twigs of juniper, pinyon, and other available plant species; 5 percent empty pinyon cones; 4 percent bones of animals; 2 percent rocks,; 1 percent opuntia cactus; 1 percent mushrooms; and 2 percent deer hide and hair. The nest also contained about one-half pound of pinyon nuts and a small amount of juniper berries. In certain portions of the pinyon-juniper woodland, there was approximately one house per 1-1/4 acres. Other mammals of the community included the pinyon mouse, the deer mouse, the Utah cliff chipmunk, the jackrabbit (*Lepus californicus*), the cottontail, the rock squirrel, the porcupine, and the gray fox (*Urocyon littoralis*).

Of 233 woodrat houses visited near Jericho in west-central Utah, 90 percent were in direct association with live junipers, 6.4 percent were in dead trees or fallen limbs, 2.6 percent were on open ground, and 1.3 percent were in sagebrush (Stones 1960). Most houses in juniper were at the base of the tree, a few were in the middle portion of the tree, and one was in the top of a tree. Food caches were almost all of juniper berries and fresh juniper foliage; pinyon does not occur in this immediate area. Several other kinds of small mammals were captured at the woodrat nest, including kangaroo rats, pinyon mice, least chipmunk, deer mice, and pocket mice. The woodrat population ranged from 1 to 3 adults per acre to 8.5 adults per acre in a mixed juniper-sagebrush community. This area seems to have had an unusually high population of woodrats compared to most areas I have studied.

Reynolds (1966) observed Abert's squirrel (*Sciurus aberti* Woodhouse) feeding on pinyon pine near Silver City, New Mexico, in April 1965.

Numerous pinyon trees had terminal twigs clipped by squirrels, and twigs formed almost a complete mat beneath some trees. Portions of pinyon twigs without needles were stripped of bark as were ponderosa pine twigs, which were also utilized heavily by squirrels in this area. Several inches of inner bark behind the terminal needle portion was stripped from the ponderosa pine twigs found. Reynolds (1966) suggested that where small clumps of ponderosa pine are surrounded by pinyon trees, a border of pinyon should be maintained around the ponderosa pine clump to supplement the squirrels' food supply.

Birds

Many species of birds are associated with the pinyon-juniper ecosystem; some are permanent residents, some summer residents, and some winter residents, depending upon location. Sixty species of birds were listed recently by Behle and Perry (1975) as occurring in pinyon-juniper in Utah (appendix A). The following species not reported by Behle and Perry as being found in pinyon-juniper in Utah were listed in an earlier publication by Woodbury and Cottam (1962).

Rough-legged Hawk	*Buteo lagopus*
Bald Eagle	*Haliaeetus leucocephalus*
Lesser Nighthawk	*Chordeiles acutipennis*
Rufous Hummingbird	*Selasphorus rufus*
Red-shafted Flicker	*Colaptes cafer*
Clark's Nutcracker	*Nucifraga columbiana*
Winter Wren	*Troglodytes troglodytes*
Northern Shrike	*Lanius excubitor*
Common Redpoll	*Acanthis flammea*
Slate-colored Junco	*Junco hyemalis*
Tree Sparrow	*Spizella arborea*

In a 2-year study of nesting ecology of the ferruginous hawk in west-central Utah, Weston (1968) observed 27 occupied nests--11 in Utah junipers, 1 in cliffrose, 14 on the ground, and 1 on a cliff. Also, he observed 66 unoccupied nests of which 42 were on the ground, 20 were in juniper trees, 1 was in a cliffrose, and 3 were on cliffs. Nesting occurred from March to July and the ferruginous hawk was gone by September. By November, the bald eagle and rough-legged hawk moved into this area and stayed until February or early March. Other raptors permanent in this area included the golden eagle, Swainson's hawk, Kestrel (sparrow hawk), red-tailed hawk, and great horned owl.

In addition to the five permanent raptors listed by Weston, Smith (1971) observed nests of the prairie falcon, marsh hawk, Cooper's hawk, short-eared owl, burrowing owl, and raven in the same general area. Species nesting in juniper trees included the great horned owl, red-tailed hawk, Swainson hawk, Cooper's hawk, ferruginous hawk, and kestrels. Ground nesters included the marsh hawk, burrowing owl, and short-eared owl, in addition to some ferruginous hawks. Cliff sites were used by the golden eagle, great horned owl, red-tailed hawk, prairie falcon, and raven. These five species also used abandoned quarries with steep fronts for nesting as did the kestrel. Except for the nests of kestrels, which were in small crevices within tree trunks, tree nests were mainly in treetops. Nests of kestrels in quarries were also in crevices--8 to 32 feet above the quarry floor. The great horned own did not build nests. Instead, it used old nests of ravens, ferruginous or red-tailed hawks, or it laid eggs on the dirt or rock floors of crevices on ledges. Of these nesters, ravens chose the most remote and secluded areas for their nests.

In a study of birds in the Book Cliff Mountains of eastern Utah, Hardy (1945) considered 8 species to be permanent residents, but only 3 were obligate to pinyon-juniper: pinyon jay, gray titmouse, and the lead-colored bushtit, all of which nest in trees. The bushtit also nests in ponderosa pine trees at higher elevations. Five other species that are permanent residents but depend upon rocks and cliffs for nesting included: the western red-tailed hawk, golden eagle, kestrel, great horned owl, and American raven. Fourteen other birds, mostly passerines and primarily insect and seed eaters, are summer residents of this pygmy conifer area. Included among them are the broad-tailed hummingbird, ash-throated flycatcher, Say's phoebe, and canyon wren. Among the birds that nest in tree cavities is mountain bluebird. Hardy indicated that bluebirds are greatly disturbed when the sparrow hawk or horned owl appear, but do not seem to be alarmed by the red-tailed hawk.

Six species of birds that nest in the foliage of trees include the magpie, western gnatcatcher (*Polioptila caerulea amoenissima*), and black-throated gray warbler. The common house finch has been found in pygmy conifers in summer and is known to bring off two broods of young. The western chipping sparrow (*Spizella passerina arizonae*) nests in trees; Brewer's sparrow nests in lower greasewood (*Sarcobatus vermiculatus*) and sagebrush, but often appears in the pygmy forest.

The mourning dove nests in trees as well as upon the ground. Other ground nesting birds include the lark sparrow and lesser nighthawk (*Chordeiles minor howelli*). The latter merely deposits its eggs on the ground beneath a tree. Red-tailed hawks feed upon mourning doves, and the antelope ground squirrel often destroys their eggs.

Birds nesting in cavities beneath rocks and other semiunderground situations include the common rock wren, whose predators include the striped racer (*Masticophis taeniatus taeniatus*) and the desert woodrat. The antelope ground squirrel may also be destructive to these birds.

Woodbury and Russell (1945) reported that birds characteristic of the pygmy conifer type in southeastern Utah and adjacent areas include four permanent residents: pinyon jay, gray titmouse, lead-colored bushtit, and desert Bewick wren. Birds that breed in the pigmy conifer type include: poor will, ash-throated flycatcher, gray flycatcher, blue-gray flycatcher, and black-throated gray warbler. Other birds common to the pinyon-juniper woodland, but not necessarily limited to the area, are the following permanent residents: kestrel, horned owl, flicker, hairy woodpecker, mountain chicadee, white-breasted nuthatch, and house finch. Breeders not necessarily limited to the type include: mourning dove, black-chinned hummingbird, Cassin's kingbird, Say's phoebe, mountain and western bluebirds, plumbeous (solitary) vireo, and chipping sparrow.

Rasmussen (1941) observed that the most abundant resident birds on the Kaibab Plateau were the gray titmouse, Woodhouse's jay, western redtailed hawk, golden eagle, red-shafted flicker, pinyon jay, lead-colored bushtit, and rock wren. Summer resident birds include western chipping sparrow, black-throated gray warbler, Rocky Mountain grosbeak (*Guiraca caerulea*), western mourning dove, night hawk, northern cliff swallow, western lark sparrow, and desert sparrow. Most common winter resident birds were Shufeldt's junco, pink-sided junco, gray-headed junco, red-backed junco, Rocky Mountain nuthatch, mountain bluebird, western robin, and long-crested jay.

Balda (1969) reported that the oak-juniper woodland of southeastern Arizona supported 36 species of birds having a total nesting density of 267 pairs per 100 acres.

Each kind of bird found in the pygmy forest has one or more characteristic differing from other birds in the area that enables it to survive by partially eliminating competition. These habits include getting food, nesting incubation periods, and other adaptive characteristics.

Balda and Bateman (1971) studied a flock of about 250 pinyon jays northeast of Flagstaff, Arizona, for 2-1/2 years. From October through December, the flock fed as a unit. During January and February, pairs separated from the flock and courted in relative isolation from the main feeding flock. A special dining call was used by most members of the flock as cue to reassemble. Adult birds began nesting in late February to mid-March. All nests were within a traditional 230-acre breeding ground that had been used for at least 4 years. In any one year, nests were dispersed over about 120 acres. During incubation and brooding, nesting females were fed by their mates. Young birds were fed only by their parents for the first 6 days; after that time, helpers brought food to the young. Adults and young from nearby nests formed communal units for feeding young until late summer. In late summer, all feeding units gathered on the pinyon-laden slopes within the home range to gather and cache pinyon seeds, mostly on traditional breeding grounds, indicating strong reliance on pinyon nuts for reproduction. Individuals from one flock did not mingle with birds of other flocks and returned to their own flock even after a separation of 21 miles. This flock maintained a well-defined home range of about 8 square miles for 2 years, but left this home range the fall of 1970, probably because of lack of pinyon nuts.

Reptiles

Diurnal reptiles probably represent the group of animals best adapted to the hostile environment of the pinyon-juniper ecosystem (Woodbury 1933). In Zion National Park, the most important species are the ring-necked lizard (*Crotaphytus collaris baileyi* Stej.); the brown-shouldered lizard (*Uta stansburiana stansburiana* B. & G.); the desert whiptail lizard (*Cnemidophorus tessellatus tessellatus* Say); the red racer snake (*Masticophus flagellum frenatus* Stej.); and the Great Basin rattlesnake (*Crotalus confluentus lutosus* Klauber). The three lizards are insectivorous, but the ringneck is also cannibalistic and preys on other lizards. It is usually found on rocky slopes and hillsides and retreats under rocks for shelter. The brown-shouldered lizard occupies rocky, bushy or scrub forest types. It retreats under a bush or behind a rock for shelter. The desert whiptail lizard runs at great speed, is a ground dweller, and does not climb either rocks or trees. The red racer's food consists of lizards, bird eggs, young birds, and small rodents. It is found in foothills and valleys. The rattlesnake is a common inhabitant of the pygmy forest.

Reptiles found on the Kaibab pinyon-juniper woodland include: the short-horned toad (*Phrynosoma douglassi*); the sagebrush swift (*Sceloporus graciosus*); the blue-bellied lizard (*Sceloporus elongatus*); the brown-shouldered Uta; the western collared lizard; the Great Basin rattlesnake; the Grand Canyon rattlesnake; and the Great Basin gopher snake (*Pituophis catenifer*) (Rasmussen 1941). The horned toad is the most common reptile in the upper limits of pinyon-juniper, averaging 6 to 8 animals per acre in some areas. The four lizards occur in the lower and more open areas. The Great Basin gopher snake is found mainly in lower areas where sagebrush makes up a great part of the community.

Invertebrates

Rasmussen collected 83 species of invertebrates in the pinyon-juniper community, but 37 of these were taken only once. Various groups accounted for certain percentages of the total number taken as follows: spiders 25 percent, Chermidae 13 percent, Formicidae 12 percent, other Hymenoptera (mostly Ichneumonidae) 8 percent, Diptera 10 percent, Cicadellidae 9 percent, Hemiptera 6 percent, Coleoptera 5 percent, Orthoptera 3 percent, and others 9 percent.

Ants were predominate on the ground surface. The termite (*Reticulitermes tumiceps* Banks) made galleries in both pinyon and juniper wood where dead limbs were on the ground. Considering all invertebrates, the ground strata contained slightly more animals per acre than other strata and shrub and tree strata averaged more animals per acre than herb and grass strata. Two maximums in animal numbers occurred in late May and a lesser one in September. The total invertebrate population was much less than other researchers had found in deciduous forests of Illinois.

Woodbury (1933) indicated that because of sparse, dry, ground litter in Zion National Park, snails, millipeds, and ground beetles are uncommon. At certain periods, cicadas are conspicuous in the trees because of their noisy singing. Larval stages are spent underground for several years. Larvae feed on live roots sometimes as deep as 4 feet. Adult Tenebrionide beetles forage on the soil surface; larval stages forage in the soil.

Such predaceous insects as the robberflies (Asilidae) and bee flies (Bombyliidae) also complete larval stages in the soil. Scorpions and centipedes seek shelter under rocks during the daytime and search out their prey at night. Tarantulas usually use holes in the ground.

Insects often destroy 90 percent of the pinyon cone crop (Keen 1958). Staminate cones are often attacked by small, white-yellow larvae in maturing catkins. Subsequently, first-year cones are injured by gall midges (Itonidae). In the second year, cone moths and cone beetles often cause extensive damage. Grasshoppers have been reported to feed on the surface of juniper berries, causing them to wither and crack open. Juniper berries also are often heavily infested by a chalcid. In one instance, about 25 percent of the seeds of *Juniperus occidentalis* were infested.

Certain plants that grow in openings in the pinyon-juniper woodland are hosts for more species of insects than might be found on the trees. For example, Jorgensen and Tingey (1968) reported that over 300 species of insects were collected from three species of native shrubs (big sagebrush, rubber rabbitbrush, and bitterbrush) and one introduced grass (crested wheatgrass) in several shrub-grass communities interspersed among or near pinyon-juniper stands in west-central Utah. Special study of thrips (*Thysanoptera*) from this collection revealed that these four species of plants were hosts for 20 species of thrips, some of which damaged plants (Tingey et al. 1972).

Riffle (1972) reported that two species of nematodes, *Xiphenema americanum* and *Rotylenchus pumilis*, parasitized *J. monosperma* seedlings and reduced root weights and root collar diameters. Four of seven nematode species studied parasitized *Pinus edulis* seedlings, but did not significantly reduce seedling growth.

Seed Dispersal by Animals and Birds

After discussing certain faunal relationships, I will consider tree seed dispersal by some of the fauna, perhaps the most important mutualistic effect of pinyon-juniper-fauna relationships.

At Benmore, we have found dehulled juniper seeds in cheek pouches of Great Basin pocket mice, chisel-toothed kangaroo rats, and chipmunks. Also, we have found dehulled juniper seeds in rabbit pellets up to one-quarter mile away from the nearest juniper tree.

Johnsen (1962) reported that some seeds passed by animals germinate more readily than others. Also, that seeds remain viable for many years: 16 percent of alligator juniper (*J. deppeana*) seeds germinated after 9 years; 17 percent of Utah juniper seeds germinated after 45 years; and 54 percent of one-seed juniper germinated after 21 years. Invasion of grasslands by junipers has been attributed to distribution of seeds by animals (Parker 1945).

In Texas, racoons, foxes, wildcats, chipmunks, and other animals eat juniper fruits, and seeds can be found in their feces. Phillips (1910) observed that chipmunks and other squirrels in South Dakota had from 14 to 51 berries in their cheek pouches.

Smith (1948) referred to work by Wolff, who found that from October to February half of the food that jackrabbits eat in the redberry juniper area is the fruit of that tree, and that rabbits dispersed approximately 310,000 seeds per acre. Rabbits passed seeds as much as 1 mile from trees. Cottontails prefer Ashe (*J. ashei*) juniper seeds to redberry juniper seeds.

Phillips (1910) reported on use of juniper seeds by birds and indicated that birds are the most important means of seed distribution. Juniper seeds are available to birds and animals from the time seeds mature in fall until late the following spring, when many other seeds are covered by snow. Phillips noted that E. A. Mearns, experimenting with a caged waxwing (*Bombycilla garrulus*), found that more than 900 berries of *Juniperus scopulorum* passed through the bird in 5 hours. Phillips reported further that robins have been observed feeding in juniper trees in flocks of 20 or more per tree. He also mentioned that fence rows are often lined by junipers that have grown from seeds distributed by birds and which exist long after the fence has disappeared. Junipers often occur also under ponderosa pines and hardwoods where birds have roosted. On poor sites in New Mexico, 90 to 95 percent of all reproduction of *Juniperus monosperma* and alligator juniper was the result of bird dispersal.

Phillips (1910) stated that the Biological Survey had found juniper berries were eaten by the following birds: Canada grouse (*Canachites canadensis*), common crow, trail flycatcher (*Eitrailli*), plumed quail (*Oreortyx plumiferus*), sharp tailed grouse (*Pedioecetes phasianellus*), robin, black capped chickadee, kingbird, Bohemian waxwing, wild turkey (*Meleagris gallopavo*), cedar waxwing, purple finch, yellow-shafted flicker, Myrtle warbler, downey woodpecker, evening grosbeak, hermit thrush, white-tailed ptarmigan (*Lagopus leucurus*), mockingbird (*Mimus polyglottos*), fox sparrow (*Passerella illaca*), pine grosbeak (*Pinicola enucleator*), Say's phoebe, bluebird, yellow-bellied sapsucker (*Sphyrapicus varius*).

Summary and Conclusions

Many species of fauna use the pinyon-juniper ecosystem during the year; permanent residents are fewer in number. Perhaps the most characteristic small mammals are the woodrat and the pinyon mouse. Birds considered to be obligate to pinyon-juniper are the pinyon jay, the titmouse, and the lead-colored bushtit.

Deer is a dominant species in the pygmy forest and the most important game species in the ecosystem. Elk are often there in winter, at which time deer are also numerous. Pronghorn antelope are often found on the fringes of the pygmy forest where they have access to more open areas of sagebrush and salt-desert shrub communities. Desert bighorns are found in the pinyon-juniper woodland, but their habitat is determined more by the rough, steep topography, which gives them maximum protection from their enemies, than by the trees themselves. Where herbaceous species and shrubs are present for forage, wild horses use this forest type extensively year round. In southeastern Utah, a herd of buffalo also roams through the tree type, which provides cover as it does for other animals.

The main carnivores in the ecosystem are the mountain lion, coyote, bobcat, badger, and weasel. Numbers of mountain lions and coyotes increase during winter when deep snows force them and their main prey species down from higher ranges.

Porcupines, jackrabbits, and cottontails use pinyon-juniper areas extensively, but they are not necessarily permanent residents. The small mammals found in a given area of the pinyon-juniper woodland might belong to less than a dozen species, even where shrub and herbaceous plants provide a diversity of habitat. Usually not more than a half dozen reptile species are found in pinyon-juniper areas, except where rocks and cliffs provide additional habitat.

Except for invertebrates, birds constitute the majority of faunal species in the pinyon-juniper ecosystem. At least 75 species of birds are there some time during the year. This number includes about a dozen species of raptorial birds. Approximately half of these nest in trees; the other half depend more upon cliffs, quarries, and ground sites for nesting.

The presence of both pinyon and juniper trees on many areas provides food for many wildlife species that eat the highly nutritious seeds. A mutualistic relationship exists between the trees that produce food and the animals the disperse seeds, thereby insuring perpetuation of trees. Large crops of seeds might well stimulate reproduction in birds and certain animal species; at least, this is known to be true of pinyon pine cones and the pinyon jay. The mere sight of green cones stimulated reproduction in experimental birds in New Mexico (Ligon 1974).

Juniper berries are more consistently available than pinyon nuts. The berries remain on trees a large part of the year, and they are not so completely destroyed by insects as pinyon nuts. This last could account for the fact that junipers often invade adjacent plant communities first, followed by pinyon. However, the reverse situation exists in the Pine Nuts Mountains of western Nevada where pinyon trees comprise over 90 percent of the tree communities. These observations suggest that the primary invasion by one or the other species is related to the total volume of seeds produced and dispersed.

Because many species of birds and animals disperse seeds of these pygmy conifers, it can be expected that the tree community will continue to expand. One author concluded that the pinyon-juniper ecosystem has expanded sixfold since settlement in the mid-1800s.

The great challenge to resource managers is to achieve a proper balance among trees, shrubs, and herbaceous plants so as to benefit wildlife and other demands and uses. These last include grazing by domestic livestock, tree products, recreation, and watershed values.

The viewpoint expressed by Stearns and Creed (1964) would seem to apply to the pinyon-juniper pygmy forest as well as to major forest types: ". . . land management plans to benefit wildlife should provide maximum diversity of forest types and adequate harvest of game populations. Whereas nature once provided this diversity through catastrophic events, wind storms, fires, insect and disease epidemics, etc., the resource manger must now do for wildlife what catastrophic events once did. The outlook for the wildlife resources is far brighter with intensively managed forests than without."

Multiple use management with due considerations for all demands and uses for its products seems the best approach for managing the resources of the pinyon-juniper ecosystem.

Literature Cited

Arnold, Joseph F., Donald A. Jameson, and Elbert H. Reid. 1964. The pinyon-juniper type of Arizona: effects of grazing, fire and tree control. U.S. Dept. Agric. Prod. Res. Rep. 84.

Baker, Maurice F., and Neil C. Frischknecht. 1973. Small mammals increase on recently cleared and seeded juniper rangeland. J. Range Manage. 26(2):101-102.

Balda, R. P. 1969. Foliage use by birds of the oak-juniper woodland and ponderosa pine forest in southeastern Arizona. Condor 71:399-412.

Balda, R. P., and G. C. Bateman. 1971. Flocking and annual cycle of the pinyon jay. Condor 73: 287-302.

Barney, Milo A., and Neil C. Frischknecht. 1974. Vegetation changes following fire in the pinyon-juniper type of west central Utah. J. Range Manage. 27(2):91-96.

Beale, Donald M., and Arthur D. Smith. 1973. Mortality of pronghorn antelope fawns in western Utah. J. Wildl. Manage. 37(3):343-352.

Behle, William H., and Michael L. Perry. 1975. Utah Birds: Check-list, seasonal and ecological occurrence charts and guides to bird finding. Utah Museum of Natural History, Univ. Utah.

Black, Hal. L., and Neil C. Frischknecht. 1971. Relative abundance of mice on seeded sagebrush-grass range in relation to grazing. USDA For. Serv. Res. Note INT-147.

Bradley, Stephen Robert. 1971. The altitudinal distribution of mammals of the LaSal Mountains Utah. M.S. thesis, Brigham Young Univ., Provo.

Cottam, Walter P. 1961. The impact of man on the flora of the Bonneville Basin. Lecture, Feb. 20, 1961, Univ. Utah.

Dealy, J. Edward. 1971. Habitat characteristics of the Silver Lake mule deer range. U.S. Dep. Agric. For. Serv. Pac. Northwest For. & Range Exp. Stn. Res. Pap. PNW-125.

Frischknecht, Neil C., and Maurice F. Baker. 1972. Voles can improve sagebrush rangelands. J. Range Manage. 25(6):466-468.

Gladfelter, Harold Lee. 1966. Nocturnal behavior of white-tailed deer in the Hatter Creek enclosure. Res. Completion Rep., Idaho Fish & Game Dep. & Coop. Wild. Res. Unit, For., Wildl. & Range Exp. Stn., Univ. Idaho.

Hansen, R. M., and L. D. Reid. 1975. Diet overlap of deer, elk, and cattle in Southern Colorado. J. Range Manage. 28:43-47.

Hardy, R. 1945. Breeding birds of the pygmy conifers in the Book Cliff regions of eastern Utah. Auk. 62:523-542.

Interagency Committee. 1950. Range for big game and livestock in Utah. Utah State Dept. Fish and Game Publ. 2.

Jenny, Hans. 1941. Factors of soil formation. McGraw-Hill Book Co., Inc. New York and London.

Johnsen, T. N., Jr. 1962. One-seed juniper invasion of northern Arizona grasslands. Ecol. Monogr. 32:187-207.

Jorgensen, Clive D., and Ward M. Tingey. 1968. Insects of the Sagebrush-grass community in Utah. Prog. Rep. 19 p. in files Intermt. For. and Range Exp. Stn., Provo, Utah.

Julander, Odell. 1952. Forage habits of mule deer during the late fall as measured by stomach content analyses. U.S. Dep. Agric. For. Serv. Intermt. For. and Range Exp. Stn., Res. Note 2.

Julander, Odell. 1955. Deer and cattle range relations in Utah. For Sci. 1(2):130-139.

Julander, Odell. 1962. Range managment in relation to mule deer habitat and deer productivity in Utah. J. Range Manage. 15(5):278-281.

Keen, F. P. 1958. Cone and seed insects of western forest trees. USDA Tech. Bull. 1169.

Kufeld, R. C. 1973. Foods eaten by Rocky Mountain elk. J. Range Manage. 26:106-113.

Kufeld, Roland C., O. C. Wallmo, and Charles Feddema. 1973. Foods of the Rocky Mountain Mule Deer. US Dep. Agric. For. Serv. Res. Pap. RM-111.

Kundaeli, John N., and Hudson G. Reynolds. 1972. Desert cottontail use of natural and modified pinyon-juniper woodland. J. Range Manage. 25(2) 116-118.

Leach, Howard R. 1956. Food habits of the Great Basin deer herds of California. Calif. Dep. Fish and Game 42(4):244-362.

Ligon, J. David. 1974. Green cones of the piñon pine stimulate late summer breeding in the piñon jay. Nature 250(5461):80-82.

Major, Jack. 1951. A functional factorial approach to plant ecology. Ecology 32(3):392-412.

Parker, K. W. 1945. Juniper comes to the grassland: suggestions on control. Am. Cattle Prod. 27:12-14.

Phillips, Frank J. 1910. The dissemination of junipers by birds. For. Q. p. 60-73.

Rasmussen, D. I. 1941. Biotic communities of Kaibab Plateau. Ecol. Monogr. 11:229-275.

Reynolds, Hudson G. 1964. Elk and deer habitat use of a pinyon-juniper woodland in southern New Mexico. Trans. 29th North Am. Wildl. and Nat. Res. Conf. March 9-11, Publ. by Wildl. Manage. Inst. Wire Bldg. Washington, D. C.

Reynolds, Hudson G. 1966. Abert's squirrel feeding on pinyon pines. J. Mammal. 47:550-551.

Riffle, Jerry W. 1972. Effect of certain nematodes on the growth of *Pinus edulis* and *Juniperus monosperma* seedlings. J. Nematol. 4(2):91-94.

Shelford, Victor E. 1963. The Ecology of North America. 610 p. Univ. Ill. Press, Urbana.

Shepherd, H. R. 1972. Rodent effects on pinyon-juniper deer range. Colo. State Div. Wildl. Proj. W-101-R-14 Game Range Invest., Aug. 7, 1956-March 31, 1964.

Smith, Arthur D., and Richard L. Hubbard. 1954. Preference ratings for winter deer forages from northern Utah ranges based on browsing time and forage consumed. J. Range Manage. 7(6): 262-265.

Smith, Arthur D. 1959. Adequacy of some important browse species in overwintering of mule deer. J. Range Manage. 12:8-13.

Smith, Arthur D., Donald M. Beale, and Dean D. Doell. 1965. Browse preferences of pronghorn antelope in southwestern Utah. Trans. 30th North Am. Wildl. and Nat. Resour. Conf. March 8-10.

Smith, Arthur D., and Dean D. Doell. 1968. Guides to allocating forage between cattle and big game on big game winter range. Utah State Div. Fish and Game Publ. 68-11.

Smith, Dwight G. 1971. Population dynamics, habitat selection, and partitioning of breeding raptors in the Eastern Great Basin of Utah. PhD diss., Brigham Young Univ.

Smith, H. N. 1948. Rabbits spread cedar menace. The Cattleman 35(3):97-98.

Spencer, Donald A. 1964. Porcupine population fluctuations in past centuries revealed by dendrochronology. J. Appl. Ecol. 1:127-150.

Stearns, Forest W., and William A. Creed. 1964. Wildlife habitat and the managed forest. Wis. Acad. Trans. 53(Part A):123-129.

Stones, Robert C. 1960. Life history study of the desert woodrat. MS thesis, Brigham Young Univ.

Tingey, Ward M., Clive D. Jorgensen, and Neil C. Frischknecht. 1972. Thrips of the sagebrush-grass community in west-central Utah. J. Range Manage. 25:304-308.

Trout, Lester, E., and J. L. Thiessen. 1973. Physical condition and range relationships of the Owyhee deer herd. Job Completion Rep., Big Game Range Invest., Proj. W-141-R-2, Study II, Job. L, Idaho Fish and Game Dept.

U.S. Department of Agriculture. 1936. The western range. Sen. Doc. 199.

U.S. Forest Service. 1958. Timber resources for America's future. USDA For. Serv. Res. Rep. 14.

West, John Bradford. 1968. Nesting ecology of the ferrugioous hawk (*Buteo regalis*) in west-central Utah. MS thesis, Brigham Young Univ.

Wight, J. R., and H. G. Fisser. 1968. *Juniperus osteosperma* in northwestern Wyoming: their distribution and ecology. Wyo. Agric. Exp. Stn. Sci. Monogr. 7.

Wilson, Lanny. 1968. Distribution and ecology of the Desert Bighorn Sheep in southeastern Utah. State Utah, Dep. Nat. Resour., Div. of Fish and Game, Publ. 68-5.

Wood, J. E., T. S. Bickle, W. Evans, J. C. Germany, and V. W. Howard. 1970. The Fort Stanton mule deer herd. N. Mex. Agric. Exp. Stn. Bull. 567.

Woodbury, A. M. 1933. Biotic relationships in Zion Canyon, Utah with special reference to succession. A sur. of geol. bot. and geol. interrelationships within Zion Natl. Park, Utah. Ecol. Monogr. 3:147-246.

Woodbury, Angus, M., and Clarence Cottam. 1962. Ecological studies of birds in Utah. Univ. Utah Biol. Ser. Vol. XII, 7.

Woodbury, A. M., and H. N. Russell, Jr. 1945. Birds of Navajo Country. Univ. Utah Bull. 35, Biol. Ser. 9(1):1-160.

Yoakum, Jim. 1971. Habitat management for the Desert Bighorn. First Annu. North Am. Bighorn Conf. Proc. 1:158-162.

APPENDIX

Birds observed in pinyon juniper in Utah

Common Name	Scientific Name
✓ Turkey Vulture	*Cathartes aura teter*
✓ Red-tailed Hawk	*Buteo jamaicensis*
✓ Swainson's Hawk	*Buteo swainsoni*
✓ Ferruginous Hawk	*Buteo regalis*
✓ Golden Eagle	*Aquila chrysaetos canadensis*
✓ Marsh Hawk	*Circus cyaneus hudsonius*
✓ Prairie Falcon	*Falco mexicanus*
Peregrine Falcon	*Falco peregrinus*
✓ American Kestrel (Sparrow Hawk)	*Falco sparverius sparverius*
Sage Grouse	*Centrocercus urophasianus urophasianus*
Chukar	*Alectoris chukar*
✓ Mourning Dove	*Zenaida macroura marginella*
✓ Great Horned Owl	*Bubo virginianus*
Pygmy Owl	*Glaucidium gnoma californicum*
✓ Long-eared Owl	*Asio otus tuftsi*
✓ Saw-whet owl	*Aegolius acadicus acadicus*
✓ Poor-will	*Phalaenoptilus nuttalii nuttallii*
✓ Common Nighthawk	*Chordeiles minor*
✓ White-throated Swift	*Aeronautes saxatalis*
✓ Black-chinned Hummingbird	*Archilochus alexandri*
✓ Broad-tailed Hummingbird	*Selasphorus platycercus platycercus*
✓ Common Flicker	*Colaptes auratus*
✓ Hairy Woodpecker	*Dendrocopos villosus*
✓ Downy Woodpecker	*Dendrocopos pubescens leucurus*
✓ Western Kingbird	*Tyrannus verticalis*
Cassin's Kingbird	*Tyrannus vociferans vociferans*
✓ Ash-throated Flycatcher	*Myiarchus cinerascens cinerascens*
Say's Phoebe	*Sayronis saya*
Gray Flycatcher	*Empidonax wrightii*
Western Flycatcher	*Empidonax difficilis hellmayri*
✓ Violet-green Swallow	*Tachycineta thalassina lepida*
Scrub Jay	*Aphelocoma coerulescens*
✓ Pinyon Jay	*Gymnorhinus cyanocephalus*
✓ Black-billed Magpie	*Pica pica hudsonia*
✓ Common Raven	*Corvus corax sinuatus*
✓ Common Crow	*Corvus brachyrhynochos*
✓ Mountain Chickadee	*Parus gambeli*
Plain Titmouse	*Parus inornatus ridgwayi*
Bushtit	*Psaltriparus minimus plumbeus*
✓ White-brested Nuthatch	*Sitta carolinensis nelsoni*
Bewicks' Wren	*Thryomanes bewickii eremophilus*
✓ American Robin	*Turdus migratorius propinquus*
Western Bluebird	*Sialia mexicana*
✓ Mountain Bluebird	*Sialia currucoides*
Blue-gray Gnatcatcher	*Polioptila caeurulea amoenissima*
✓ Cedar Waxwing	*Bombycilla cedrorum*
✓ Loggerhead Shrike	*Lanius ludovicianus gambeli*
✓ Starling	*Sturnus vulgaris vulgaris*
Gray Vireo	*Vireo vicinior*
✓ Solitary Vireo	*Vireo solitarius*
Black-throated Gray Warbler	*Dendroica nigrescens*
Scott's Oriole	*Iceturs parisorum*
House Finch	*Carpodacus mexicanus frontalis*
✓ Pine Suskin	*Spinus pinus*
✓ Rufous-sided Towhee	*Pipilo erythrophathalmus*
Lake Sparrow	*Chondestes grammacus strigatus*
✓ Chipping Sparrow	*Spizella arborea orchracea*
✓ Brewer's Sparrow	*Spizella breweri breweri*
Black-chinned Sparrow	*Zonotrichia atricapilla*

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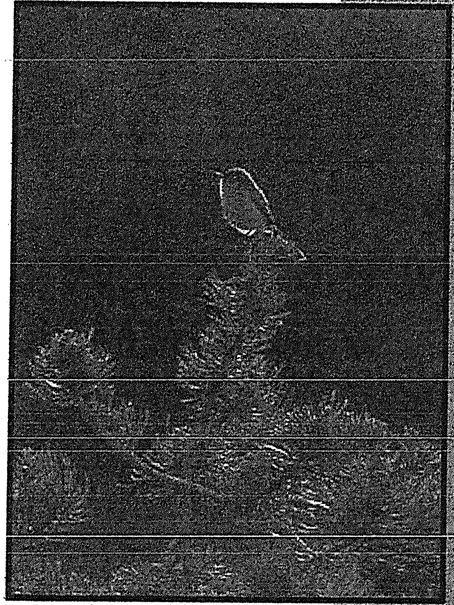

Sharing the Land with Pinyon-Juniper Birds

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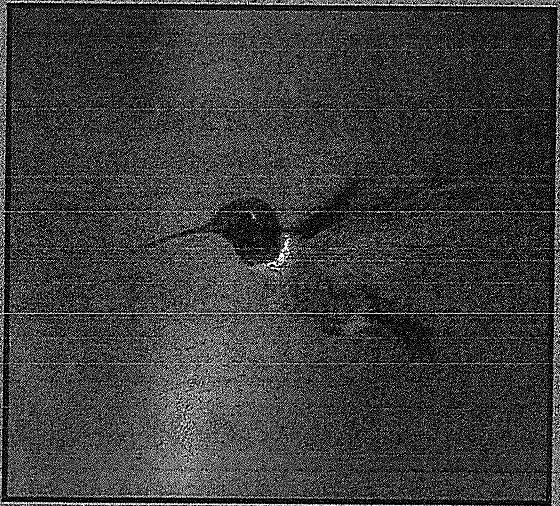


Scott W. Gillihan
Rocky Mountain
Bird Observatory

State of Utah
Department of Natural Resources
Division of Wildlife Resources

National Fish and Wildlife Foundation

Sharing the Land with Pinyon-Juniper Birds

Research and writing
Scott W. Gillihan, Rocky Mountain Bird Observatory,
14500 Lark Bunting Lane, Brighton, CO 80603

Design and layout
Sherrie York, Natural Visions,
310 Crestone Avenue, Suite 3, Salida, CO 81201

Printing
Utah Division of Wildlife Resources

Funding
U.S. Bureau of Land Management
U.S. Fish and Wildlife Service
Utah Division of Wildlife Resources
Colorado Division of Wildlife
Idaho Department of Fish & Game
Arizona Game & Fish Department
The Nature Conservancy, Migratory Bird Program
Rocky Mountain Bird Observatory

State of Utah
Department of Natural Resources
Division of Wildlife Resources

Acknowledgments

The National Fish and Wildlife Foundation provided the challenge grant (Project #2002-0141) from BLM and US Fish and Wildlife Service contributions. Idaho BLM State Office supplemented funds. Catherine Robertson, Manager of the Grand Junction (Colorado) Field Office provided the staff to oversee the project.

Development of this document was initiated and guided by a committee of the Partners in Flight Western Working Group: Carol Beardmore (U.S. Fish and Wildlife Service–Arizona), Ron Lambeth (Bureau of Land Management–Colorado), Sharon Nicholoff (Wyoming Game and Fish Department), Christine Paige (Ravenworks Ecology), Jim Parrish (Utah Division of Wildlife Resources), and Sharon Ritter (USDA Forest Service–Montana). Other reviewers included Russell Balda (Northern Arizona University), Marilyn Colyer (Mesa Verde National Park), Jim Davis (Utah Division of Wildlife Resources), John Fairchild (Utah Division of Wildlife Resources), Steve Kettler (The Nature Conservancy–Colorado), David Klute (Colorado Division of Wildlife), David Mehlman (The Nature Conservancy, Migratory Bird Program), Larry Neel (Nevada Department of Wildlife), Jim Ramakka (Bureau of Land Management–New Mexico), Terry Rich (U.S. Fish & Wildlife Service/Partners in Flight), George L. San Miguel (Mesa Verde National Park), Scott Walker (Utah Division of Wildlife Resources), and David Worthington (Capitol Reef National Park). Rocky Mountain Bird Observatory staff members Alison Cariveau, Glenn Giroir, David Hanni, Tony Leukering, Rich Levad, William Palmer, Arvind Panjabi, and Tammy VerCauteren provided editorial and logistical assistance. Brenda M. Martin (Fort Collins Museum) assisted with the material on indigenous peoples. However, any errors or omissions are the sole responsibility of the author.

Cover photos
Blue-gray Gnatcatcher (top)—Tony Leukering, RMBO
Landscape—Glenn Giroir, RMBO
Black-chinned Hummingbird (bottom)—© Brian Small / www.briansmallphoto.com

The mission of Rocky Mountain Bird Observatory is to conserve Rocky Mountain and Great Plains birds and their habitats through research, monitoring, education, and outreach.

Suggested citation
Gillihan, S. W. 2006. Sharing the land with pinyon-juniper birds. Partners in Flight Western Working Group. Salt Lake City, Utah.

Sharing the Land with Pinyon-Juniper Birds

WHY SHARE? 2
ECOLOGY OF PINYON AND JUNIPER WOODLANDS 4
    Pinyon-Juniper Animals 4
    The Changing Landscape 6
HUMAN HISTORY IN PINYON AND JUNIPER WOODLANDS 7
PINYON-JUNIPER MANAGEMENT ACTIVITIES AND BIRDS 9
    General Guidelines 9
    Brown-headed Cowbirds and Brood Parasitism 10
    Livestock Grazing 11
    Big Game Management 12
    Weed, Insect, and Mistletoe Management 13
    Tree Harvesting 14
    Mineral Exploration and Extraction 14
    Seed Crop Harvesting 15
    Recreation 15
    Prescribed Burning 16
    Wildfire Management 17
    Residential Development 18
    Sage Grouse and Pinyon Juniper 18
    Water Developments 19
    Farming 19
    Watershed Management 20
PINYON-JUNIPER BIRDS OF CONSERVATION CONCERN 21
    Black-chinned Hummingbird 21
    Ferruginous Hawk 22
    Ash-throated Flycatcher 23
    Gray Flycatcher 24
    Gray Vireo 25
    Pinyon Jay 26
    Juniper Titmouse 27
    Townsend's Solitaire 28
    Western Bluebird 29
    Virginia's Warbler 30
    Black-throated Gray Warbler 31
    Scott's Oriole 32
    Other Pinyon-Juniper Birds 33
REFERENCES AND ADDITIONAL READING 34
ADDITIONAL SOURCES OF ASSISTANCE 38
APPENDIX. SCIENTIFIC NAMES OF SPECIES IN THE TEXT 39

nest season - up thru July -

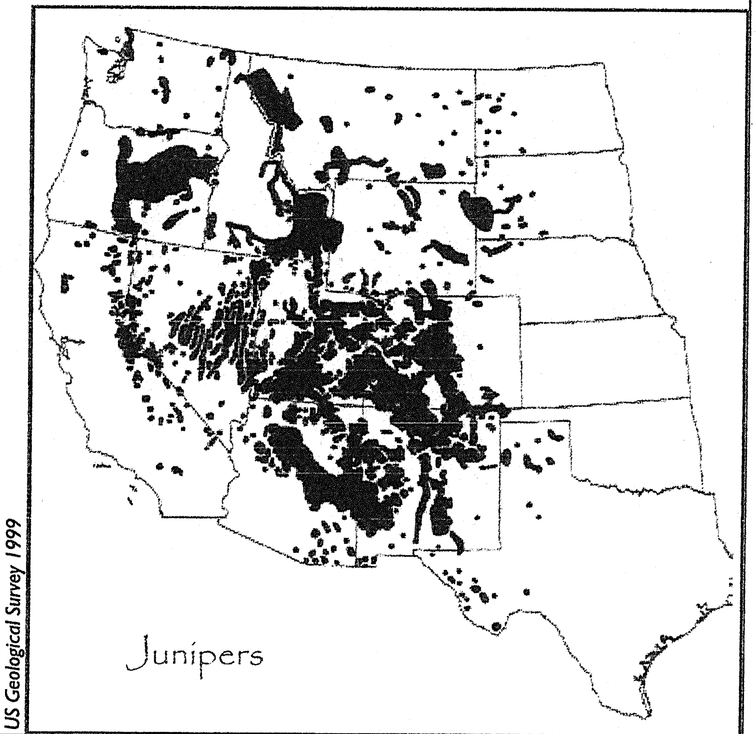
Why Share?

The pinyon-juniper woodland is a widespread ecosystem of the North American West. Estimates of its current extent vary widely, depending on how the habitat is defined and delineated. A recent estimate is 55.6 million acres (22.5 million ha; Mitchell and Roberts 1999). It is widely regarded that the extent of pinyon-juniper is increasing as some grasslands and shrublands are being transformed by pinyon-juniper encroachment. This expansion has been facilitated by a combination of climatic changes, fire suppression and, in some areas, overgrazing, which has removed the grassy understory that ordinarily would carry a fire. In some areas, pinyon-juniper woodlands are moving back into areas that formerly were woodlands but had been cleared in the late 1800s and early 1900s to meet the demand for pinyon-juniper wood products. In other areas, juniper is expanding into grasslands and shrublands where it had never been recorded before. Some researchers maintain that, at least in some areas, pinyon-juniper occurs in a dynamic equilibrium with adjacent vegetation types and that the expansion is part of a natural cycle independent of or, at best, facilitated by, human activities (Belsky 1996; Swetnam et al. 1999). These and other topics are still being debated; a number of conferences and symposia have been held to bring together researchers and managers who work in pinyon-juniper in order to share knowledge and identify information gaps that still exist (e.g., Gifford and Busby 1975; Aldon and Loring 1977; Everett 1987; Aldon and Shaw 1993; Monsen and Stevens 1999). What has come out of those gatherings is an appreciation for the habitat itself and a recognition of the need for more information.

While the body of knowledge regarding pinyon-juniper woodlands grows steadily, some synthesis of the existing knowledge is in order so that landowners and land managers can go about their business of caring for the land. One area where such a synthesis is needed, where an information vacuum exists, is in the area of managing pinyon-juniper woodlands to benefit birds.

Across a broad spectrum of habitat types, no wildlife group is as species-rich, as visible, or as vocal as birds. Pinyon-juniper woodlands are no exception. The pinyon-juniper bird community, especially in mature stands, contains a high number and variety of birds --- more than 70 species are known to breed in pinyon-juniper woodland --- although perhaps no more than 20-30 species may occur at any one site (Balda 1987; Balda and Masters 1980; Paulin et al. 1999). Pinyon-juniper woodlands support one of the highest proportions of obligate or semi-obligate bird species among forest types in the West (Paulin et al. 1999). Species closely tied to pinyon-juniper (scientific names of all species mentioned in the text are listed in the Appendix) include Black-chinned Hummingbird, Ash-throated Flycatcher, Cassin's Kingbird, Gray Flycatcher, Western Scrub-Jay, Pinyon Jay, Juniper Titmouse, Bushtit, Bewick's Wren, Northern Mockingbird, Blue-gray Gnatcatcher, Gray Vireo, Black-throated Gray Warbler, Lark Sparrow, and Black-chinned Sparrow (Balda and Masters 1980). However, little information is available on management practices that benefit bird communities in pinyon-juniper woodlands.

Birds can be useful indicators of biological integrity and ecosystem health (Hutto 1998). They fill this role because they comprise a diverse group of specialists that occupy a broad range of habitats, are sensitive to environmental changes, and reflect the abundance and diversity of other organisms with which they coexist. Responses by bird communities to environmental



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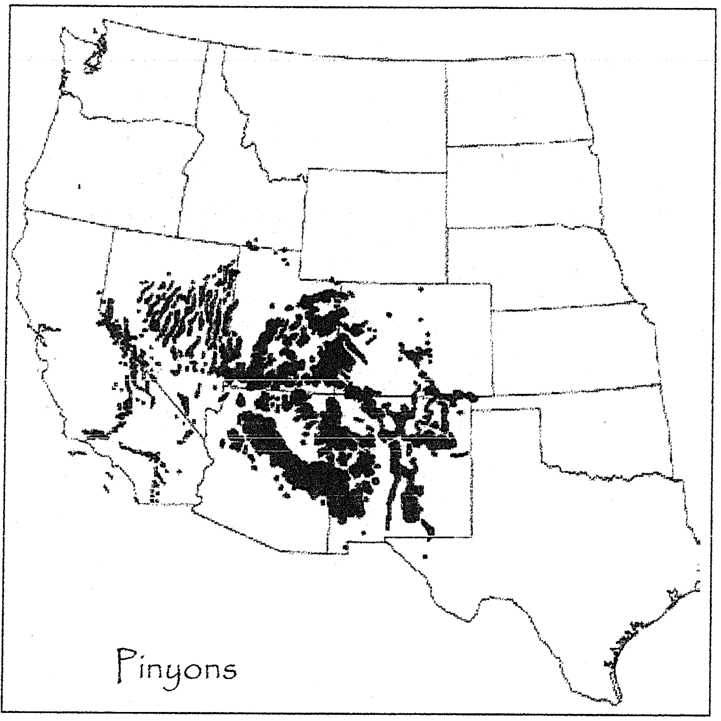

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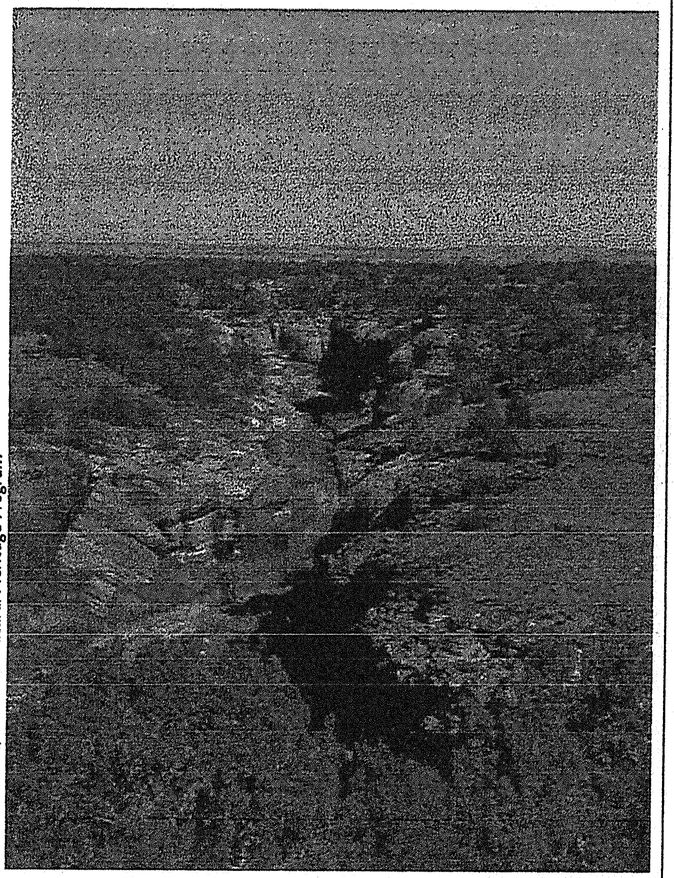
changes are well documented, and changes in bird communities often reflect the effects of resource management, conservation, restoration, and environmental degradation. There is a strong and growing interest, both nationally and internationally, to manage and conserve bird populations. Examples of this interest include the numerous bird conservation plans that have recently been completed (e.g., Brown et al. 2001, Donaldson et al. 2000, Kushlan et al. 2002, North American Waterfowl Management Plan Committee 1998; Rich et al. 2004).

Birds are a tremendous economic resource in and of themselves. Birds provide untold billions of dollars in ecosystem services as pollinators and seed dispersers of ecologically and economically important plants, and as predators on insect pests in forested and agricultural landscapes. Birds are also the basis for a recreation-related economic boon. A recent federal report found that 46 million birdwatchers across the United States spent $32 billion in 2001 on bird watching and related activities (U.S. Fish and Wildlife Service and U.S. Census Bureau 2002). This spending generated $85 billion in overall economic output and $13 billion in federal and state income taxes, and supported more than 863,000 jobs. Communities in pinyon-juniper country shared in this economic vitalization since certain bird species can only be seen in pinyon-juniper habitats and others reach their highest densities in pinyon-juniper.

Pinyons and junipers rely heavily on animals for seed distribution, a dependence that makes the pinyon-juniper woodland unique. Other forest types may require animals to disperse some seeds of some plant species, but in pinyon-juniper, animals are critical to the dispersal of the seeds of the dominant tree species. Birds, in particular, are responsible for this important role in the maintenance of pinyon-juniper woodlands. This unique ecological relationship is yet another reason for landowners and land managers to share pinyon-juniper woodlands with birds and other wildlife.

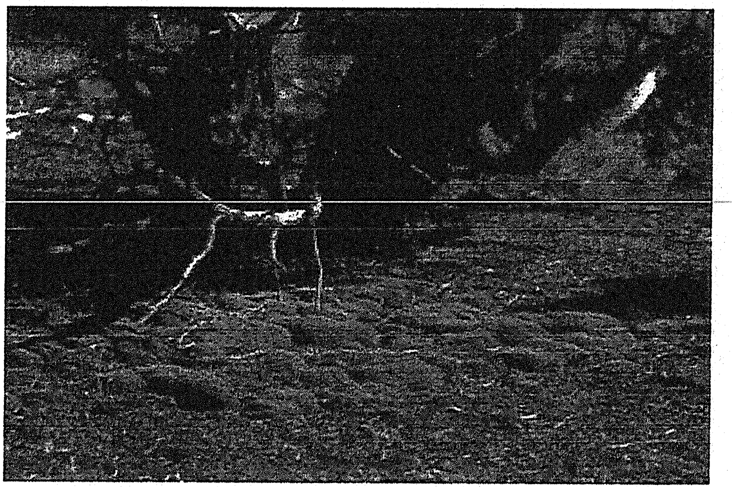
The purpose of this document is to provide information on the management of pinyon-juniper woodlands that will benefit individual bird species and bird communities while still using the woodlands for other purposes. This information will be useful to private landowners and natural resource managers on public lands in developing more comprehensive management strategies that benefit the long-term health and productivity of pinyon-juniper communities throughout the western U.S.


Renee Rondeau, Colorado Natural Heritage Program

Ecology of Pinyon and Juniper Woodlands

Pinyons are low-growing, rounded pines with one to five needles per bundle (fascicle), depending on species. The pinyon pine is represented by about 11 species, the two most widespread species in the West being the singleleaf pinyon and the Colorado pinyon. The other nine species are found primarily in Mexico, with some ranges extending into the southern portions of California, Arizona, New Mexico, and Texas (Lanner 1981).

Junipers are also conifers but with overlapping scales for leaves and seed-bearing cones that resemble small berries. Junipers associated with pinyon-juniper woodlands include about 17 species, including some in Mexico. The three species most commonly associated with pinyons in the West are alligator juniper, one-seed juniper, and Utah juniper. Other widespread species are Rocky Mountain juniper and western juniper (Lanner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Moss and biological soil crust in a pinyon-juniper woodland

Temperature and moisture play leading roles in the distribution of pinyon and juniper. Pinyon-juniper woodlands occur where annual precipitation is just 8-20 inches (200-500 mm) (West et al. 1975). That precipitation is delivered as winter snow, spring rain, and infrequent summer thunderstorms. Although mixed stands of pinyon and juniper abound, pure stands of either also occur. Junipers are more tolerant of drought and cold and therefore often dominate at the lower elevations and sometimes at the higher elevations, although the increased moisture usually associated with higher elevations tends to favor pinyons (West 1999).

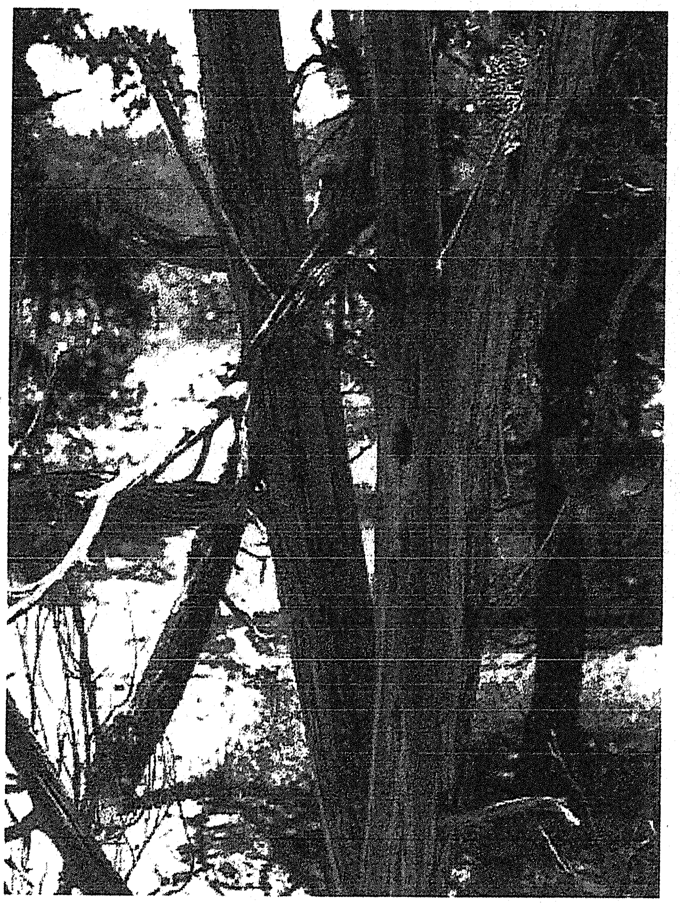
At higher elevations, junipers may drop out entirely, and pinyons grow larger and in denser stands (Pieper 1977). Both pinyons and junipers have wide elevational ranges, occurring from 2,700 to 11,000 feet (820-3,400 m) (California Partners in Flight 2002; Neel 1999; Parrish et al. 2002); species associated with pinyon-juniper range from Joshua tree and barrel cactus at low elevations to limber pine and bristlecone pine at high elevations (Tausch 1999a).

The soils in some pinyon-juniper woodlands are held in place by a biological soil crust --- a complex community of mosses, lichens, fungi, and algae forming a soil surface layer up to several inches thick. This gray-green crust is firm enough to hold soil in place if undisturbed, but easily breaks down under traffic from humans, vehicles, or livestock (Lanner 1981). The crust is predominantly moss within the dripline of trees, and algae beyond the dripline. The importance of biological soil crusts and their role in promoting soil productivity and preventing erosion has only recently been recognized and additional information is needed to more fully understand its role in pinyon-juniper communities (Belnap et al. 2001, Belnap and Lange 2001, Ladyman and Muldavin 1996).

Pinyon-Juniper Animals

Both pinyons and junipers provide food for wildlife species. Mammalian consumers of pinyon seeds include deer mouse, pinyon mouse, Abert's squirrel, rock squirrel, cliff chipmunk, Hopi chipmunk, Uinta chipmunk, Colorado chipmunk, desert woodrat, Stephen's woodrat, white-throated woodrat, Mexican woodrat, bushy-tailed woodrat, black bear, and desert bighorn. Avian consumers include Clark's Nutcracker, Pinyon Jay, Mexican Jay, Western Scrub-Jay, Steller's Jay, and Juniper Titmouse. Some insects consume pinyon needles, as do mule deer occasionally. Bark beetle larvae (mountain pine beetle and pinyon engraver beetle) and rodents (especially porcupines) consume pinyon pine phloem. Weakened or dead trees may also become host to woodborers or other insects. Other insect consumers include various sawfly species, pitch midges, and gall midges (Lanner 1981). In pinyon-juniper woodlands, the importance of pinyon pines to birds is clear --- the number of breeding bird species increases as the number of pinyon pines increases (Masters 1979).

While pinyons provide food for many animals, the animals in turn provide a service to the trees by dispersing their seeds. Most pine species have winged seeds to facilitate dispersal; the seeds are released from the cone and carried away from the parent tree by the wind. However, pinyon seeds are wingless and enclosed in pockets within the cone, which holds them in place. Rather than being dispersed by wind or gravity, pinyons rely on rodents and certain bird species to remove and disperse their seeds. Birds gather seeds in late summer and early fall and cache them in the soil for later consumption when few other food sources are available. Birds fail to relocate some of the cached seeds, which then may sprout and grow. Seeds dispersed away from the parent trees may give rise to a new stand of pinyons.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Mature juniper with nest cavity

Birds carry the seeds in their bill or esophagus. Some species are better adapted for carrying seeds than others; the Western Scrub-Jay can carry only 4 or 5 pinyon seeds, the Steller's Jay can carry up to 18, and the Pinyon Jay up to 50. The champion of seed-carrying capacity is the Clark's Nutcracker, which can carry up to 95 seeds thanks to a special pouch under its tongue (Lanner 1981). An individual jay or nutcracker may cache thousands or even tens of thousands of seeds each season, up to 13 miles (21 km) away from the parent tree (Lanner 1981; Vander Wall and Balda 1981). It has been estimated that a flock of 250 Pinyon Jays can cache 4.5 million pinyon seeds during a five-month period (Ligon 1978).

Junipers also provide food for wildlife. If more suitable browse is not available, mule deer will eat juniper foliage, sometimes browsing it extensively. Juniper berries, which consist of a hard-coated seed enclosed in a fleshy outer covering, are eaten by mammals such as rabbits, gray fox, black bear, coyote, striped skunk, and a variety of rodents, and by birds such as Western Bluebird, Mountain Bluebird, Townsend's Solitaire, American Robin, Bohemian Waxwing, and Cedar Waxwing (Lanner 1981; Chambers et al. 1999; White et al. 1999). These animals serve an important role in the future of local juniper populations by dispersing the seeds. When eaten, the hard-coated juniper seed often passes through the digestive system intact and is excreted, falling to the ground where it may germinate and become established. Removing the outer flesh increases by a factor of 10 the probability that a juniper seed will germinate. The junipers themselves facilitate this process by producing berries that are conspicuously colored blue or red and making them readily accessible on the outer layers of foliage.

Birds are the primary seed-dispersal mechanism for some juniper species (Chambers et al. 1999). Birds are effective because they deposit seeds under woody vegetation --- a suitable site for germination and seedling growth. Bird-facilitated dispersal is particularly important for reestablishing junipers within woodlands that have been burned or killed by insects or drought, provided some dead trees remain standing. Also, since seeds are usually deposited singly or in small clusters, the chances of density-dependent seed depredation are reduced. However, juniper seeds must be covered by soil to germinate, something not accomplished by the birds. Bird-dispersed seeds must be buried by some other means such as trampling, frost heaving, soil deposition, or rodent caching. Mammals distribute juniper seeds, too, sometimes traveling a mile or more before depositing their seed-ladened scats.

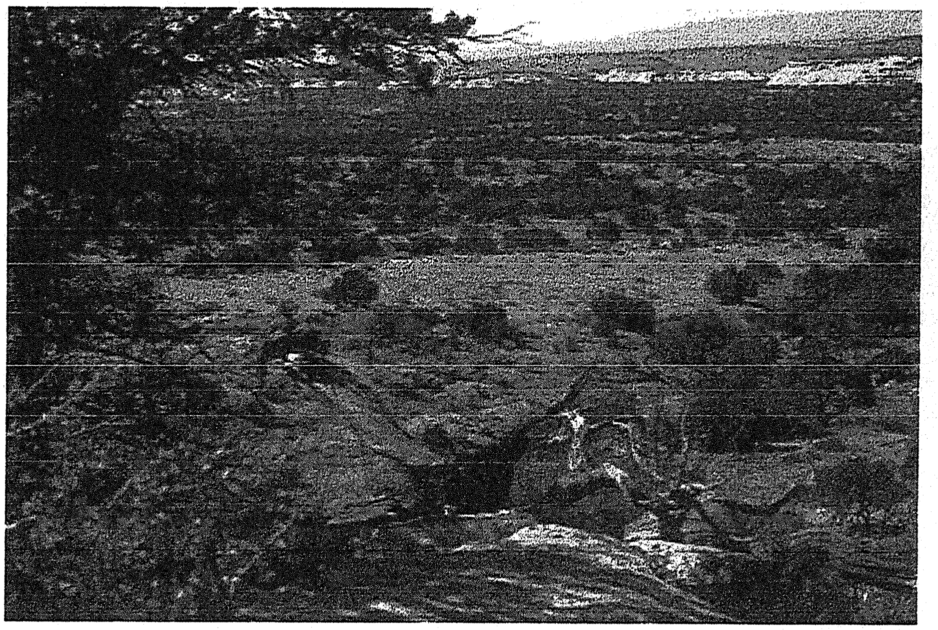
For animals that rely on these trees for food, mixed pinyon-juniper woodlands provide overlapping seed supplies. Singleleaf pinyons produce seeds every two to three years and Colorado pinyons produce seeds every five to seven years, although some seeds may be produced every year. Junipers typically produce seeds annually, although there is variation among individuals and years (Tueller and Clark 1975; Janetski 1999). Dry years seem to trigger heavy juniper berry crops, while big crops are rare in wet years.

Animals make use of the trees in other ways, too. Bats use cavities, loose bark, and broken limbs as maternity roosts. Woodrats make nests of shredded pinyon bark or juniper bark fibers (Lanner 1981). Rock squirrels use the tree cavities as caching sites of pinyon nuts and juniper berries. Birds build nests in the foliage or in tree cavities, in some cases using juniper bark fibers. Insectivorous birds search the bark and foliage of pinyon-juniper trees for insects, and raptors perch in the trees while hunting. Downed logs are used as cover by invertebrates, small mammals, and reptiles. Big game species use pinyon-juniper stands for thermal cover, especially where junipers are plentiful.

The Changing Landscape

Mature pinyons may grow to be 600 years old and mature junipers may surpass 1,000 years. Even so, in many areas the trees are less than 140 years old --- roughly the time that has passed since mining and other Euro-American settlement impacts have occurred. Little information is available about pinyon-juniper woodlands prior to Euro-American settlement. Many researchers believe that woodlands of juniper or juniper with pinyon were less common than now and in some areas a more savanna-like landscape prevailed in which the trees were scattered throughout a grass- or shrub-dominated landscape. Pinyon-juniper woodlands and even forests (where tree crowns touch) certainly existed, but they may have been largely confined to areas protected from fire, such as steep and/or south-facing slopes, rocky areas, areas with shallow soils, or other situations that precluded growth of the fine fuels needed to carry fires (Roundy and Vernon 1999; West 1999). Native Americans set fires to improve game habitat by clearing dense brush and possibly to protect their homes by reducing dangerous fuel loads; those fires and lightning-caused fires likely served to control tree expansion into grassy or shrubby areas.

Pinyon-juniper has expanded downward in elevation into areas with deeper soils and, historically, more fine fuels and higher fire frequency. This expansion was facilitated by the removal of fine fuels through heavy grazing and by the active suppression of wildfires (Roundy and Vernon 1999). In contrast, a study in a portion of the Colorado Plateau (Mesa Verde National Park) suggested that pinyon-juniper densities at higher elevations with naturally lower fire frequency have probably not increased greatly over the last 100 years and that the increases observed are probably the result of climatic changes rather than fire suppression (Romme et al. 2003). Fire intervals in much of the Mesa Verde area have likely always been on the order of hundreds of years and infrequent, severe, stand-replacing fires have likely always been the norm, rather than the exception (Romme et al. 2003). Fire history in pinyon-juniper woodlands is poorly understood, and much research remains before a clear picture of pinyon-juniper ecology can emerge (Baker and Shinneman 2004).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Human History in Pinyon and Juniper Woodlands

Carbon dating of pinyon charcoal and seed coats suggests that human use of pinyons dates back at least 6,000 years (Lanner 1981). Pinyons were prominent in the mythology of indigenous peoples living within the range of the pines and the trees undoubtedly also played an important role in the everyday lives of the people, which included the Anasazi, Zuni, Hopi, Pueblo, and Apache (Lanner 1981). Most were hunter-gatherer societies that also farmed corn and squash on a limited basis (Janetski 1999).

Indigenous peoples used pinyon and juniper woods as firewood for heat, cooking, and pottery firing, for housing material, and for carved tools, and they used pinyon pitch to make glue, dye, varnish, waterproof coating, a dressing for wounds, and even a type of nonstick coating for stone cooking surfaces. Pinyon needles or twigs were prepared as a drink to treat various illnesses, while pollen from both pinyons and junipers was used as a medicine. Pinyon charcoal was used to color ceremonial sand paintings. Shredded juniper bark was used for diapers, mats, sandal insoles, and was smoked in cigarettes (Lanner 1981). Juniper wood was used for cradle boards and spoons, and the ash was used as a leavening agent.

Pinyon nuts were also used as food. The nuts were collected in the early fall, around the time of the first frosts. Collection methods included picking them from the ground after they had naturally fallen, shaking the trees or thrashing them with poles to dislodge seeds from mature cones, or using hooked sticks to knock loose green cones. The green cones were then either stored until winter or placed on a fire to open the cones and roast the seeds. Pinyon nuts were eaten raw, roasted, boiled, ground into a flour or paste, mashed and cooked as soup or mush, or made into a spread to eat on corn cakes (Janetski 1999). Juniper berries were probably not as important a food source as pinyon nuts, but nevertheless the berries were eaten fresh or pounded for use in bread, stew, or tea.

Pinyon nuts could be stored, an important consideration in areas that experienced harsh weather or lean periods when other food was hard to acquire. The oils in raw nuts could turn rancid in one year, but roasted nuts kept much longer. Nuts or cones were stored in pits or other storage facilities and, when available, a two- or three-year supply was cached (Janetski 1999).

Pinyon nuts were an important dietary component because of their high nutritional value --- the nuts contain all 20 amino acids. The levels of seven of the nine essential amino acids are higher in pinyon nuts than in cornmeal, another staple in the indigenous diet (Lanner 1981). Not only were the nuts nutritious, but their occurrence in high concentrations and the relative ease of collection made for efficient gathering, important considerations for people living off the land.

As Euro-Americans arrived in the West, they too used pinyon and juniper trees. Early homesteaders took advantage of the strength and rot-resistance of juniper and used it for fence posts and corrals. Mining for silver in the central Great Basin was literally fueled by pinyons and junipers. The trees provided posts for shoring up mine shafts and constructing buildings, but their primary use was as firewood for smelters and heating and cooking in the mining camps (Young and Budy 1987). Unlike pinyon-juniper stands in other parts of its range, pinyon-juniper woodlands in the Great Basin were adjacent to sagebrush shrublands. In the Rockies and parts of the Southwest, ponderosa pine usually grows upslope from pinyon-juniper, while in other parts of the Southwest pinyon-juniper is associated with oak. Thus, human needs for wood products in the central Great Basin had to be met by pinyon-juniper alone (Young and Svejcar 1999).

The pace of the mining boom led to rapid deforestation of the pinyon-juniper hillsides surrounding active mines. The countryside for 10 miles around the town of Eureka, Nevada, for example, was deforested after just one year of major activity (1873); after two years, the ring extended 20 miles; after five years it was at least 35 miles (Young and Budy 1987). As workers clearcut the woods surrounding their towns, wood was brought by mule, wagon, and train from outlying areas. Much of the wood was used to produce charcoal for smelting ore. Cutting continued until the mining industry collapsed in

Pinyon-Juniper Birds of Conservation Concern

Birds can be good indicators of environmental quality --- a complete suite of bird species will occur only where habitat conditions are good. Birds are also good "umbrella" species --- other wildlife species benefit from healthy bird habitat. This section provides life history accounts for bird species of conservation concern in pinyon-juniper woodlands. These species were selected based on their ranking as priority species in pinyon-juniper in one or more of the Partners in Flight Bird Conservation Plans covering the core of the pinyon-juniper range, including Arizona (Latta et al. 1999), Colorado (Colorado Partners in Flight 2000), Idaho (Idaho Partners in Flight 2000), Nevada (Neel 1999), New Mexico (New Mexico Partners in Flight 2003), and Utah (Parrish et al. 2002). Each account includes a list of management actions you can implement to help conserve the species.

Black-chinned Hummingbird

Breeding Habitat:

This species typically breeds in riparian zones of arid regions, but also nests extensively in pinyon-juniper woodlands (Andrews and Righter 1992), especially when flowering shrubs and herbaceous plants are nearby.

Nest:

The Black-chinned Hummingbird constructs a tiny cup nest of plant down and spider webs, usually in a tree but sometimes in a shrub, often near or even over water. The nest is usually placed no higher than 10 feet (3 m). It may produce two or even three broods per season, resulting in an extended breeding season: early April to early August.

Wintering:

This species winters primarily in north-central and western Mexico in habitat that is structurally similar to the breeding habitat (Baltosser and Russell 2000).

Feeding:

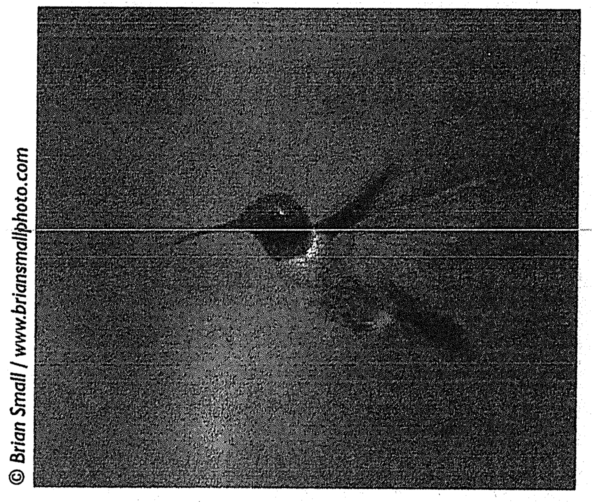
This species consumes nectar from flowers, also spiders and insects that it picks from vegetation. Dependent young are mostly fed invertebrates.

Status:

For the United States population, which is restricted to the western U.S., Breeding Bird Survey data show a significantly positive population trend of 1.4% per year for the period 1966-2005 (Sauer et al. 2006).

Conservation:

Threats to this species include loss of nectar-producing plants, habitat loss on the breeding or wintering grounds, and pesticide use. This species is fairly tolerant of human disturbance and will inhabit residential areas where the habitat is suitable.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What you can do:

- Use grazing systems that preserve flowering herbaceous plants, such as a rotation system that allows some pastures to rest without grazing pressure during the breeding season.
- Favor grazing by cattle over sheep, because sheep are more selective for flowering herbs which produce nectar needed by hummingbirds. However, it is easier to control the grazing impact of sheep and thus, if managed closely to leave ample forb stocks, sheep may be the preferred livestock.
- Avoid the use of insecticides; adopt IPM (Integrated Pest Management) practices; limit insecticide use to periods outside the breeding season.
- Plant flowering herbaceous plants, especially natives, in residential landscapes.

✓

Ferruginous Hawk

Breeding Habitat:

This species prefers relatively flat, open grasslands and shrublands and areas where these habitat types meet pinyon-juniper woodlands, especially outlier trees from main woodlots (Bechard and Schmutz 1995; Parrish et al. 2002).

Nest:

The bulky stick nest is usually placed on elevated sites, such as isolated trees (especially large, flat-topped junipers), shrubs, rock outcrops, and utility structures. Nesting occurs from March into July.

Wintering:

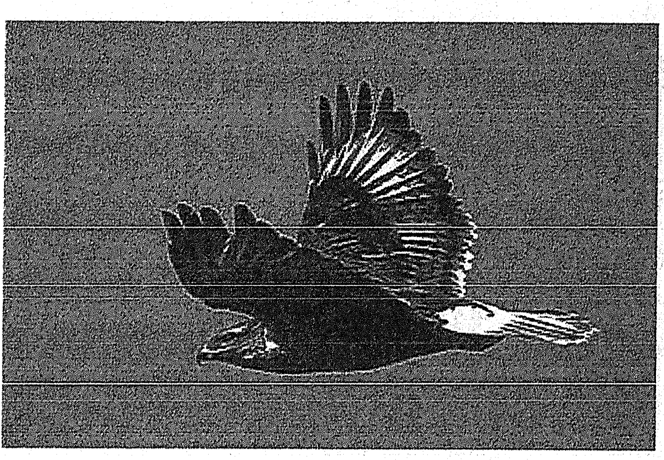
Wintering habitat is similar to breeding (i.e., open areas with few trees). This species is particularly attracted to prairie dog colonies in winter.

Feeding:

The Ferruginous Hawk hunts by scanning for prey from an elevated perch, while soaring, or while standing on the ground near burrows of prey species. In pinyon-juniper habitat this hawk hunts jackrabbits, cottontails, and small mammals.

Status:

This species has disappeared from some local areas used in times past. However, population trends over large areas are difficult to discern because the species is so rarely recorded by the Breeding Bird Survey, which records an average of just 0.25 Ferruginous Hawks/ survey route (Sauer er al. 2006), which equates to one bird for every 100 survey miles. This is a U.S. Fish and Wildlife Service *Species of Conservation Concern* in Bird Conservation Regions 9 (Great Basin) and 16 (Southern Rockies/Colorado Plateau) and at the national level (U.S. Fish and Wildlife Service 2002). It is also a Bureau of Land Management *Sensitive Species* in Colorado, Idaho, Nevada, and Wyoming, a *Sensitive Species* in U.S. Forest Service Region 2, a Colorado Division of Wildlife and Idaho Department of Fish and Game *Species of Special Concern*, and a Utah Division of Wildlife Resources *Species of Concern*. Throughout most of its breeding range, Breeding Bird Survey trends are not statistically significant for the period 1966-2005, a reflection of the rarity of the species. For the United States as a whole, the population trend is a positive 2.2% per year (Sauer et al. 2006).



Tony Leukering, RMBO

Conservation:

Threats to this species include loss of habitat due to invasion by non-native species or conversion of native land cover types to cropland (although this is more of a problem on the Great Plains than the Intermountain West) and disturbance to nest sites.

What you can do:

- Retain the open savanna structure of pinyon-juniper in areas known to support Ferruginous Hawks.
- Protect known nest trees; this may require fencing to preclude cattle from rubbing against the tree or compacting the soil around its roots.
- Maintain habitat for prey species by controlling invasive plant species such as cheatgrass.
- Maintain prey populations by limiting recreational "varmint" shooting.
- Maintain a buffer zone of 0.5 mile (0.8 km) around nests to reduce disturbance from mineral exploration/development, vehicles, or other human activities (Romin and Muck 2002). This species is extremely sensitive to human disturbance during the breeding season and will abandon nest sites if disturbance is excessive.
- Inventory lands under your control on an annual or biannual basis to locate nests or likely nest sites.

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Ash-throated Flycatcher

CN

Breeding Habitat:

This species breeds in mature pinyon-juniper or juniper woodlands, open riparian groves, or in juniper-yucca or juniper-oak ecotones (Cardiff and Dittmann 2002). It prefers low trees or shrubs for perching and foraging, often using the tips of snags or dead portions of trees, which provide unobstructed flight paths.

Nest:

This species nests in tree cavities created by woodpeckers or by rot, breakage, etc. It will use almost any suitable cavity and has been recorded using some unusual sites such as open metal pipes, mailboxes, an empty pail, even a pair of overalls hanging on a clothesline (Cardiff and Dittmann 2002). It produces a single brood during early April to early July.

Wintering:

Much of the wintering habitat in Mexico and the extreme southwestern U.S. is structurally similar to the breeding habitat (i.e., open woodlands).

Feeding:

This is an insectivore, and primarily captures flying insects by sallying out from a perch to capture them in the air. It will occasionally consume mistletoe and other small fleshy fruits.

Status:

The Breeding Bird Survey results for 1966-2005 reflect a gradual, but statistically significant, increasing trend of 1.0% per year for the United States population, which is restricted to the western U.S. (Sauer et al. 2006).

Conservation:

This species needs mature pinyon and juniper trees; younger trees are not large enough to provide nest cavities. The number of available nest cavities is probably a limiting factor. Threats include loss of habitat and pesticide use. This species adapts to human disturbance and will nest in residential areas, sometimes using nest boxes put up for bluebirds.

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What you can do:

- Retain mature pinyon and juniper trees, snags, and partially dead trees.
- Create small openings in woodlands to create foraging opportunities.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit the use of pesticides to the periods outside of the breeding season.
- Install nest boxes with floor dimensions of 6" x 6" and an entrance hole of 1.5" diameter 6-10" above the floor; mount the boxes at least 5' high.

✓

Gray Flycatcher

Breeding Habitat:

This species is closely tied to arid woodlands and shrublands, including pinyon-juniper with a sagebrush understory (Sterling 1999). In Wyoming, this species preferred mature Utah juniper stands with high juniper cover, senescent trees (as determined by presence of dead limbs, which are used as perch sites for singing or foraging when the branches extend beyond the canopy), high seedling and sapling density, and presence of pinyon pines; all factors except pinyon presence are typical for mature juniper stands (Pavlacky and Anderson 2001). In Arizona, the Gray Flycatcher is found in woodlands with tall, mature pinyon-juniper and open understory (Latta et al. 1999), although elsewhere it appears to prefer dense understory (New Mexico Partners in Flight 2003).

Nest:

The nest is an open cup of grasses, pine needles, shredded bark, and other plant materials. It is often placed in forks of branches of pinyon or juniper, on top of large branches against the trunk of ponderosa pine, or within shrubs (Sterling 1999). The female lays a clutch of three or four eggs in late April to early June, and a second clutch as late as July (Sterling 1999). This species is a host to the Brown-headed Cowbird (Sterling 1999).

Wintering:

This species winters in western and central Mexico in open habitat with scattered trees and shrubs (Sterling 1999).

Feeding:

The Gray Flycatcher captures flying insects by sallying from an exposed perch, to which it often returns. It also captures insects on the ground and on foliage, branches, and trunks (Sterling 1999).

Status:

This is a Partners in Flight continental *Stewardship Species* (Rich et al. 2004). Breeding Bird Survey (BBS) data for 1966-2005 show statistically significant increasing population trends of 2.2% per year in the BBS Pinyon-Juniper Ecoregion and 5.0% per year in the western U.S. (Sauer et al. 2006).

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Conservation:

Threats on the breeding and wintering range may include pesticide use, grazing that reduces shrub density or grass cover, and loss of mature shrubs and pinyon-juniper trees. This species is fairly intolerant of human disturbance, such as from residential development. The impact of Brown-headed Cowbird brood parasitism has been studied very little, but preliminary results suggest that the rate could be quite high in places (Sterling 1999).

What you can do:

- Retain old-growth shrublands (i.e., large shrubs).
- Retain mature trees.
- Create small openings within pinyon-juniper woodlands (Latta et al. 1999).
- Take steps to discourage Brown-headed Cowbird populations.
- Manage for a pinyon-to-juniper ratio of about 1:1, with at least 13% canopy cover (Latta et al. 1999; LaRue 1994).
- Retain snags and partially dead trees.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit insecticide use to periods outside the breeding season.

Gray Vireo

Breeding Habitat:

This species prefers mixed pinyon-juniper, oak scrub, and/or chaparral in hot, arid mountains and high plains shrublands (Barlow et al. 1999), and also occurs in juniper-dominated or pure juniper stands (Colorado Partners in Flight 2000) or open pinyon-juniper woodlands with widely spaced trees; it avoids dense stands (Dexter 1998; Neel 1999). Pairs often establish territories on steep slopes (Colorado Partners in Flight 2000).

Nest:

This species constructs its open cup nest in the fork of tree branches (Barlow et al. 1999), especially if the tree has a dead branch extending above the live foliage --- the dead branch may serve as a singing or foraging perch (Dexter 1998). A clutch of three or four eggs is incubated by both adults. This species is a Brown-headed Cowbird host and pairs typically abandon a parasitized nest (Barlow et al. 1999). Nesting takes place from early April through June with only a single brood produced.

Wintering:

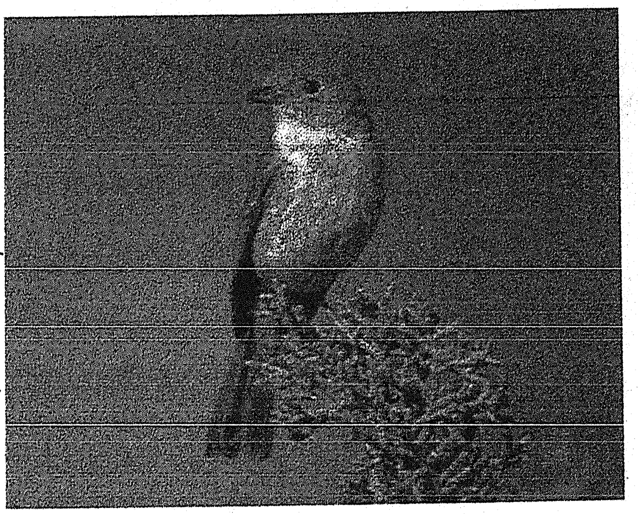
The Gray Vireo winters in southwestern Arizona and northwestern Mexico in desert scrub dominated by elephant trees, the fruits of which are the primary winter food of the species (Barlow et al. 1999).

Feeding:

This is an insectivorous species, gleaning insects from foliage and bark of trees and shrubs (Barlow et al. 1999) and catching insects in flight.

Status:

This is a Partners in Flight continental *Watch List Species* (Rich et al. 2004), and a U.S. Fish and Wildlife Service *Species of Conservation Concern* in Bird Conservation Regions 9 (Great Basin) and 16 (Southern Rockies/ Colorado Plateau) and at the national level (U.S. Fish and Wildlife Service 2002). Due to very low sample sizes, Breeding Bird Survey data for 1966-2005 reveal no statistically significant trends for any geographic area covered by BBS (Sauer et al. 2006).



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Conservation:

Threats to the species include habitat loss from conversion of pinyon-juniper to rangelands (Barlow et al. 1999) and comparatively high rates of depredation by jays and squirrels (Dexter 1998). However, the greatest threat may be the clearing of winter habitat, as the species is dependent upon a couple species of elephant trees in a restricted range.

What you can do:

- Maintain stands of mature pinyon-juniper.
- Take steps to discourage Brown-headed Cowbird populations.
- Maintain open pinyon-juniper woodlands with a shrubby understory, especially on moderate to steep rocky slopes.
- Use mechanical thinning to prevent stands from becoming too dense and to create small openings with shrubs and edge.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit the use of pesticides to the periods outside of the breeding season.

Pinyon Jay

Breeding Habitat:

This species is closely tied to pinyon-juniper woodlands, but also breeds in sagebrush, scrub oak, chaparral, ponderosa pine, and Jeffrey pine forests (Balda 2002). It prefers mature stages of pinyon (Short and McCulloch 1977), which produce more seeds. If habitat conditions are suitable, a flock may occupy the same home range for decades (Ryser 1985). However, because pinyon crops are unpredictable, flocks may wander in search of adequate seed supplies.

Nest:

The nest is a relatively large, bulky, open cup composed of sticks and lined with grasses, hair, feathers, and soft plant parts (Balda 2002); the bulkiness presumably provides insulation because this species begins nesting in winter, as early as February (Balda 2002). Some nesting attempts may start later in the spring with young still in the nest as late as August. The nest is placed in ponderosa, pinyon, or juniper trees, often on the south-facing side (Balda 2002). This species nests colonially.

Wintering:

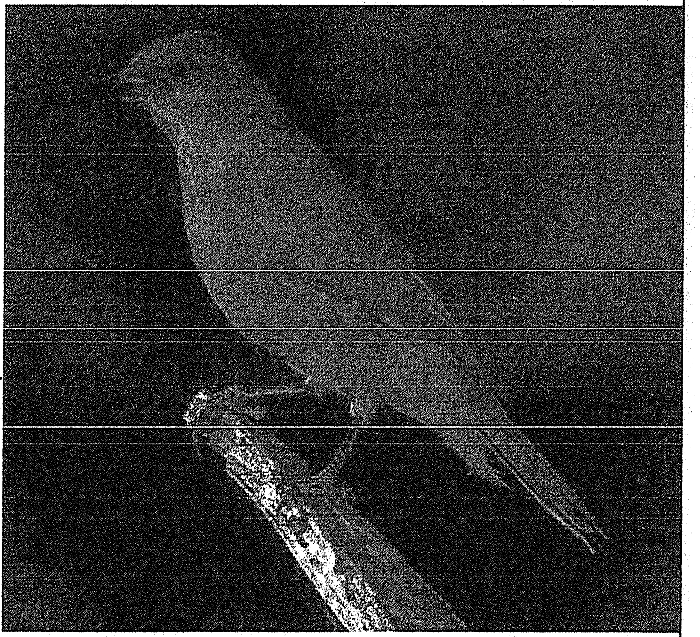
This species is a year-round resident, although individuals may disperse far from their normal range in years of cone crop failure.

Feeding:

The Pinyon Jay is an omnivore, consuming pine seeds, acorns, juniper berries, invertebrates, and small vertebrates; food items are collected on the ground, from vegetation, and occasionally by aerial capture (Balda 2002).

Status:

This is a Partners in Flight continental *Watch List Species* (Rich et al. 2004), a U.S. Fish and Wildlife Service *Species of Conservation Concern* in Bird Conservation Region 16 (Southern Rockies/Colorado Plateau) (U.S. Fish and Wildlife Service 2002), and an Idaho Department of Fish and Game *Species of Special Concern*. This species is undergoing a significant decline throughout its range; Breeding Bird Survey (BBS) data reveal statistically significant declines in the United States (-4.6% per year), the western U.S. (-4.7% per year), BBS Pinyon-Juniper Ecoregion (-7.3% per year), BBS Basin & Range Ecoregion (-7.2% per year), California (-8.4% per year), Colorado (-5.5% per year), and Nevada (-9.3% per year) during the period 1966-2005 (Sauer et al. 2006).



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Conservation:

The Pinyon Jay is a species of concern because of loss of pinyon-juniper habitat through conversion to other land cover types including clearing for residential development. Other factors include a widespread, prolonged drought in the Southwest, which has resulted in diminished cone crop production, engraver beetle infestations, and tree mortality. Also, this species is critically important in pinyon pine seed dispersal. Partners in Flight has called for a 100% increase in the continental populations of this species (Rich et al. 2004).

What you can do:

- Retain patches of mature pinyon or pinyon-juniper of at least 7 square miles (18 km^2), which is approximately the area of each flock's home range (Balda and Bateman 1971).
- Retain large trees (which are the most prolific cone-producers) since successful breeding is very closely tied to pinyon pines and seed production.
- Inventory lands under your control on an annual or biannual basis to locate nesting sites.
- Develop roads, picnic areas, or other sources of disturbance no closer than 0.6 miles (1 km) from known nesting sites since colonies are sensitive to human disturbance.

CN

Juniper Titmouse

Breeding Habitat:

This species is a resident of juniper woodlands or pinyon-juniper woodlands where juniper is dominant; it requires large trees that provide natural cavities for nesting (Cicero 2000). In Wyoming, this species preferred mature juniper stands with high juniper cover, senescent trees, dead limbs, and presence of pinyon pines; the old trees provide cavity-nest sites; the pinyon pines may be preferred foraging substrate (Laudenslayer and Balda 1976).

Nest:

This species nests in tree cavities excavated by woodpeckers or formed by rot or broken branches. It will also use artificial nest boxes. It may partially excavate its own nest cavity if the wood is soft or rotten (Cicero 2000).

Wintering:

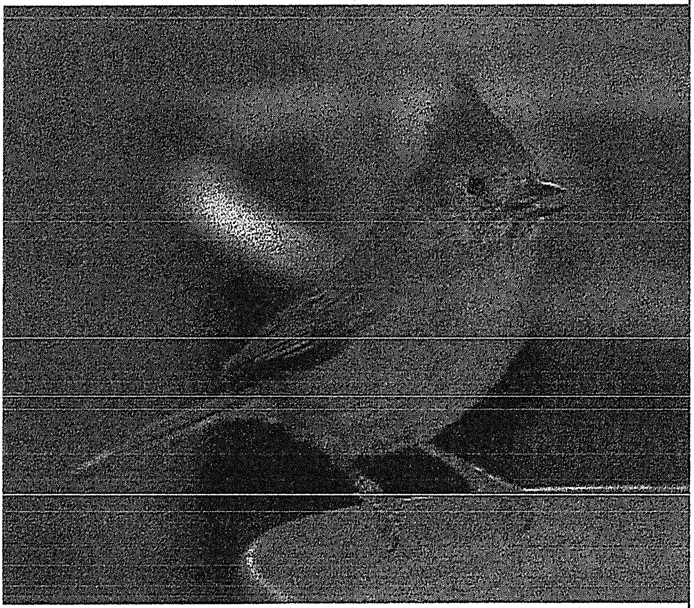
The Juniper Titmouse is nonmigratory, and pairs defend their territories year-round, although some birds may move upslope into ponderosa pine forests during winter.

Feeding:

This species eats seeds (and is known to be a major consumer of pinyon seeds), terrestrial invertebrates, and fruits. In fall and winter it eats mainly seeds and juniper berries. The insects are gleaned from the foliage and bark of trees and shrubs and from the ground (Cicero 2000).

Status:

Breeding Bird Survey (BBS) data for 1966-2005 do not show statistically significant population trends at the national scale or the scale of the western U.S., but they do show significant declines for the BBS Pinyon-Juniper Woodland Ecoregion (-2.7% per year), Colorado (-4.3% per year), and New Mexico (-2.8% per year) (Sauer et al. 2006).



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Conservation:

This species is a pinyon-juniper obligate. Because it is a cavity-nester, it needs mature woodlands, which provide trees large enough to support nest cavities.

What you can do:

- Retain older juniper trees with large, twisted trunks (Cicero 2000), which provide nest cavities.
- Retain trees with cavities.
- Retain large snags and partially dead trees.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit the use of pesticides to the periods outside of the breeding season.
- Install nest boxes with floor dimensions of 4" x 4" and an entrance hole 1.25" diameter, 6-10" above the floor; mount the boxes at least 5' high on a tree or post.

Townsend's Solitaire

Breeding Habitat:

This species is found in montane habitats up to and even above treeline. It inhabits a variety of open coniferous forest types, occasionally including pinyon-juniper (Bowen 1997).

Nest:

This species nests on the ground in a well-protected site, such as among tree roots, or in a rocky niche on a slope or rock wall.

Wintering:

During winter, the Townsend's Solitaire moves into juniper or pinyon-juniper woodlands where it establishes and defends a feeding territory. The size of the territory is linked to the availability of juniper berries --- smaller territories where juniper berries are plentiful, larger territories where the berries are scarce (Salomonson and Balda 1977).

Feeding:

Individuals commonly catch flying insects in mid-air by sallying out from an exposed perch. Solitaires collect spiders, other invertebrates, and small berries such as mistletoe from vegetation or the ground. In winter, this species lives almost exclusively on juniper berries, which it picks off the ground or from tree branches. Each solitaire eats an estimated 42,000-84,000 one-seed juniper berries each winter (Salomonson and Balda 1977).

Status:

Breeding Bird Survey data for 1966-2005 reveal a statistically significant population trend for the United States (2.5% per year) (Sauer et al. 2006).

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Conservation:

This species serves an important role in the dispersal of juniper seeds, and is almost completely dependent on junipers for food during winter. Mature trees produce the greatest berry crop.

What you can do:

- Maintain open woodlands through thinning or prescribed burning.
- Retain mature juniper trees, which provide winter food --- mature junipers produce more berries than do younger trees.
- Retain mistletoe-infected trees.
- Retain fruiting shrubs such as currants and elderberries.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit the use of pesticides to the periods outside of the breeding season.

CN

Western Bluebird

CN

Breeding Habitat:

This species breeds in open woodlands with snags and grassy areas, and burned forest with standing snags. It is most commonly associated with ponderosa pine and aspen forests, but also breeds in open pinyon-juniper woodlands (Guinan et al. 2000).

Nest:

This species nests in tree cavities, either those formed naturally by the tree or excavated by woodpeckers. It will also use artificial nest boxes. Nesting may begin as early as March, followed by a second brood as early as May. In some areas, pairs will produce a third brood (Guinan et al. 2000).

Wintering:

During winter, the Western Bluebird moves down in elevation from its montane breeding sites to open woodlands, shrublands, and riparian areas. Some populations also move south as far as central Mexico.

Feeding:

The summer diet is primarily invertebrates while the winter diet includes small berries such as mistletoe and juniper. It frequently captures flying insects by sallying out from an exposed perch, and collects invertebrates and small berries from foliage or on the ground.

Status:

Breeding Bird Survey (BBS) data for 1966-2005 fail to reveal a statistically significant population trend for any geographic region other than U.S. Fish and Wildlife Service Region 6 (Montana, Wyoming, Utah, Colorado, Kansas, Nebraska, and the Dakotas), where the population is increasing at a rate of 7.3% per year (Sauer et al. 2006).

Conservation:

The Western Bluebird is dependent on large trees and snags, which provide substrate for nest cavities. It also serves an important role in the dispersal of juniper seeds.

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What you can do:

- Leave snags and dying trees for foraging perches and nesting sites, especially those that already contain cavities.
- Use thinning and prescribed burning to convert dense forest into more favored open woodland, but retain all large trees and snags.
- Avoid insecticide use; adopt IPM (Integrated Pest Management) practices; limit the use of pesticides to the periods outside of the breeding season.
- Retain fruiting shrubs such as currants and elderberries.
- Install nest boxes with floor dimensions of 5" x 5" and an entrance hole 1.5" diameter, 6-10" above the floor; mount the boxes 4-6' above the ground on a post. Mount them at least 1/4 mile (400 m) from buildings to reduce competition with the non-native House Sparrow.

Virginia's Warbler

Breeding Habitat:

This species is typically associated with pinyon-juniper and oak woodlands with a shrubby understory (Olson and Martin 1999). A dense, tall shrub layer is critical for foraging and nesting, and the species may breed in various montane shrub habitats.

Nest:

This species places its small, shallow cup nest on the ground, where the nest is concealed by vegetation. Nesting occurs from mid-May to mid-July. Virginia's Warbler pairs raise a single brood each season.

Wintering:

Virginia's Warbler winters in south-central Mexico in open woodlands and shrubby areas.

Feeding:

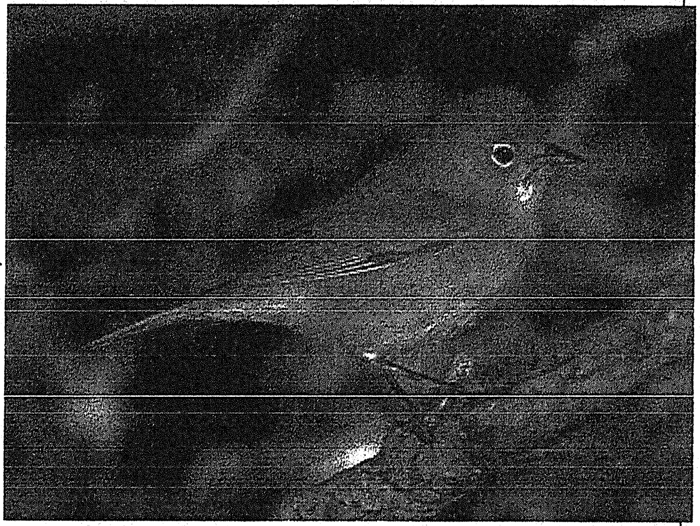
This species is insectivorous, gleaning insects from vegetation and the ground and occasionally capturing insects in flight.

Status:

This is a Partners in Flight continental *Watch List Species* (Rich et al. 2004), a U.S. Fish and Wildlife Service *Species of Conservation Concern* in Bird Conservation Region 16 (Southern Rockies/Colorado Plateau) (U.S. Fish and Wildlife Service 2002), and an Idaho Department of Fish and Game *Species of Special Concern*. The Breeding Bird Survey does not show a statistically significant population trend for any geographic region during the 1966-2005 period (Sauer et al. 2006). Reasons for this lack of a clear trend could be highly variable counts or too few birds counted.

Conservation:

The rate of Brown-headed Cowbird nest parasitism is comparatively low for this species, possibly due to its well-hidden nest. However, introducing livestock into the range of this species could result in increases in cowbird nest parasitism. Other potential hazards associated with grazing include nest trampling and livestock range projects that reduce shrub density.



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What you can do:

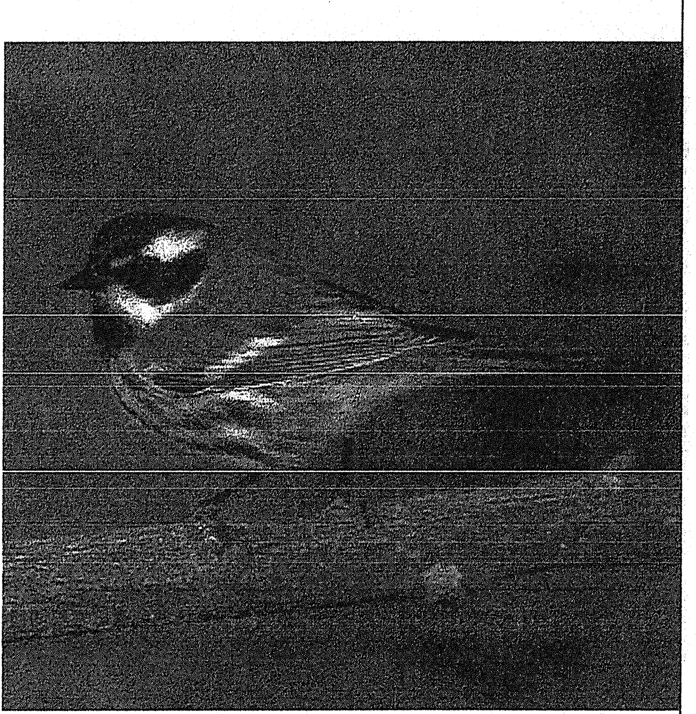
- Maintain livestock at levels that preserve shrub cover. Accept some shrub stands that are too thick for ungulates to penetrate.
- Protect shrub cover from prescribed burns, wildfires, or management actions that could remove them. Shrub removal negatively impacts this species in the short term by removing its nesting cover and principal foraging substrate (Olson and Martin 1999).
- Avoid the use of pesticides in nesting or foraging areas, especially during the breeding season.

√

Black-throated Gray Warbler

Breeding Habitat:

This species breeds in open or dense pinyon-juniper or other coniferous woodlands with a shrubby understory (Guzy and Lowther 1997; Neel 1999) and in mature pinyon-juniper (Versaw 1998). In Wyoming juniper stands, it was more commonly found with pinyon pines with high seedling and sapling density and a strong shrub component; the pinyon pines and shrubs might provide more invertebrate prey for this insectivore (Pavlacky and Anderson 2001).



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Nest:

The nest is a deep open cup nest of grasses and other plant fibers lined with feathers and fur, commonly placed on a horizontal tree branch, 3 to 10 feet (1 to 3 m) away from the trunk (Guzy and Lowther 1997). Nesting occurs from early April into August. Each pair raises a single brood each year, but double-brooding is suspected (Guzy and Lowther 1997).

Wintering:

This warbler winters in western and central Mexico in habitat similar in species composition (or at least in structure) to breeding habitat, and in shrubby areas and tall montane and riparian forest (Guzy and Lowther 1997).

Feeding:

This species is insectivorous, and gleans insects from the foliage of trees and shrubs (Guzy and Lowther 1997).

Status:

This is a Partners in Flight continental *Stewardship Species* (Rich et al. 2004) and a U.S. Fish and Wildlife Service *Species of Conservation Concern* in Bird Conservation Region 16 (Southern Rockies/Colorado Plateau) (U.S. Fish and Wildlife Service 2002). Breeding Bird Survey data for 1966-2005 show statistically significant population decline for the Arizona-New Mexico-Texas-Oklahoma state group of -3.7% per year (Sauer et al. 2006).

Conservation:

This species is a pinyon-juniper obligate throughout much of its range. It is a host species to Brown-headed Cowbirds (Goguen and Mathews 1998).

What you can do:

- Retain mature overstory pinyon-juniper trees (Guzy and Lowther 1997), tall shrubs, and seedling/sapling cover.
- Maintain at least a 15% pinyon-juniper canopy cover (Parrish et al. 2002).
- Avoid using insecticides in areas known to contain Black-throated Gray Warbler populations; if they must be used, adopt IPM (Integrated Pest Management) practices, and limit the use of pesticides to periods outside the nesting period.
- Take steps to discourage Brown-headed Cowbird populations (see page 10).

15% CC

✓

Scott's Oriole

Breeding Habitat:

The Scott's Oriole breeds in open pinyon-juniper or pure juniper woodlands, especially with yuccas, and in canyon forests, Joshua tree woodlands, and desert scrub (Flood 2002).

Nest:

This species uses fiber from yucca leaves, grass, and cactus fibers to build its hanging cup nest in juniper trees. Nesting occurs from late April through July.

Wintering:

The Scott's Oriole winters in open woodlands and desert scrub of western and central Mexico.

Feeding:

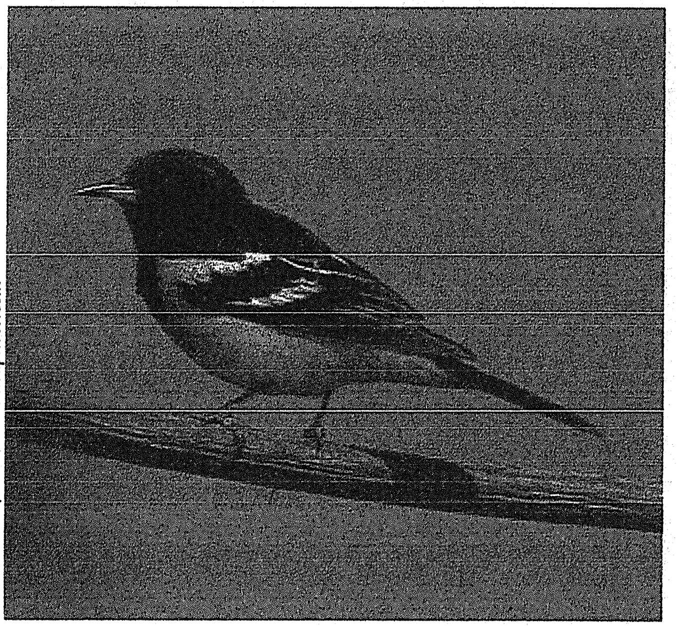
This species is insectivorous, collecting prey on the ground or low in vegetation. It also consumes berries, cactus fruits, and nectar.

Status:

This is an Idaho Department of Fish and Game *Species of Special Concern*. Breeding Bird Survey data for 1966-2005 do not demonstrate a statistically significant population trend for this species for any geographic region (Sauer et al. 2006) due to very low sample sizes.

Conservation:

Wildfires, prescribed fires, or management actions that remove shrubs negatively impact this species. However, where those same actions thin dense low-elevation stands of pinyon-juniper, they may benefit Scott's Oriole. Some west Texas populations of this species are significantly impacted by brood parasitism of the Bronzed Cowbird (Flood 2002).



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What you can do:

- Where practical, avoid insecticide use, adopt IPM (Integrated Pest Management) practices, or limit the use of pesticides to the periods outside of the breeding season.
- At the lowest elevational range of pinyon-juniper, manage for open juniper stands with no more than 60 trees per acre (150 trees per ha) (Colorado Partners in Flight 2000).
- Take steps to reduce cowbird brood parasitism (see page 10).

Other Pinyon-Juniper Birds

Western Scrub-Jay ✓

This species inhabits shrubby habitats, especially oaks, but is also found regularly in pinyon-juniper woodlands. It is nonmigratory, but individuals may wander widely in winter. It adapts well to suburbia and is not very sensitive to human disturbance. Like other jays, this species is omnivorous, eating invertebrates, small vertebrates, small fruits, acorns, and pine nuts (Curry et al. 2003).

What you can do: Preserve oak stands, especially mature mast-producing stands.

Bushtit

The Bushtit is a resident of open habitats with conifers, including pinyon-juniper woodlands. It is comfortable in suburban settings. It builds a pendulous, gourd-shaped nest of spider webs and plant material. It gleans insects and spiders from plant foliage, especially from pinyons (Sloan 2001).

What you can do: Reduce or eliminate the use of insecticides.

Bewick's Wren ✓

This species prefers dense shrubs mixed with open woodland such as pinyon-juniper. It picks invertebrates from vegetation or the ground. It is a cavity-nester, and will accept artificial nest boxes (Kennedy and White 1997).

What you can do: Preserve mature stands, especially junipers, which tend to have more cavities. Reduce or eliminate the use of insecticides. Install nest boxes with floor dimensions of 4" x 4" and an entrance hole of 1.25" diameter, 4-6" above the floor; mount 5-10' high on a tree. Note: nest boxes may not be successful where House Wrens are present, as they may usurp Bewick's Wrens.

Blue-gray Gnatcatcher ✓

This species inhabits pinyon-juniper woodlands and shrublands. It eats small invertebrates gleaned from vegetation. Its open cup nest leaves it susceptible to brood parasitism by the Brown-headed Cowbird; in some areas the rate is quite high (Ellison 1992).

What you can do: Take steps to reduce Brown-headed Cowbird parasitism (see page 10). Reduce or eliminate the use of insecticides.

Mountain Bluebird

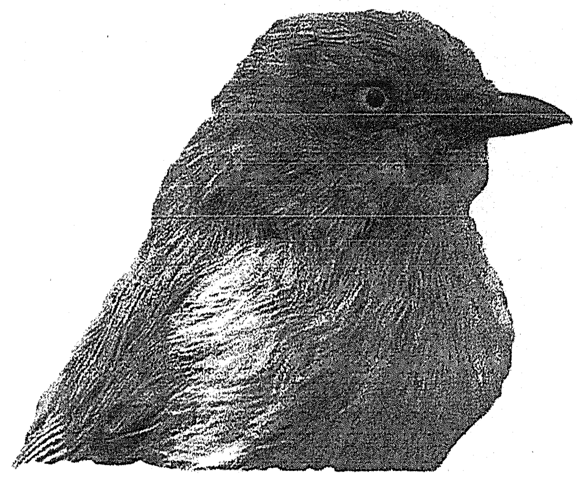
The Mountain Bluebird prefers open woodlands and forest edges. It consumes insects, spiders, and small fruits. A cavity-nester, it will use artificial nest boxes (Power and Lombardo 1996).

What you can do: Preserve trees with cavities, especially at forest or woodland edges or in open woodlands. Preserve standing dead trees as foraging perches. Reduce or eliminate the use of insecticides. Retain fruiting shrubs such as currants or elderberries. Install nest boxes to encourage nesting (see nest box dimensions for Western Bluebird on page 29); place nest boxes at least 1/4 mile (400 m) from buildings to discourage House Sparrows.

Chipping Sparrow

This species inhabits coniferous forests, including pinyon-juniper woodlands. In particular, it prefers open woodlands with little or no shrub cover, and often places its flimsy cup nest in trees or shrubs at the edge of a woodland. It feeds on insects and small seeds, which it collects from the ground. This is a frequent host to the Brown-headed Cowbird (Middleton 1998).

What you can do: Take steps to reduce Brown-headed Cowbird brood parasitism (see page 10).



Glenn Giroir, RMBO

Female Bushtit

Protections sought for Western bird linked to piñon forests amid deforestation

From: Michael Garrity (wildrockies@gmail.com)
To: sjjohnsonkoa@yahoo.com; katie@wildlandsdefense.org
Date: Wednesday, April 27, 2022, 11:38 AM MDT

Protections sought for Western bird linked to piñon forests amid deforestation

https://www.greatfallstribune.com/story/news/2022/04/27/endangered-species-protections-sought-western-bird-pinon-jays-linked-pinon-forests-new-mexico-nevada/9553646002/

Associated Press

April 27, 2022

ALBUQUERQUE, N.M. — Collecting piñon nuts has been tradition for Native American and Hispanic families in the Southwestern U.S. for generations.

But environmentalists are concerned that without the pinyon jay — a very social bird that essentially plants the next generation of trees by stashing away the seeds — it's possible the piñon forests of New Mexico, Nevada and other Western states could face another reproductive hurdle in the face of climate change, persistent drought and more severe wildfires.

The Washington, D.C.-based group Defenders of Wildlife filed a petition Monday with the U.S. Fish and Wildlife Service to protect the bird under the Endangered Species Act, saying the once common species plays an integral role in the high desert ecosystem.

The group points to research that shows pinyon jay numbers have declined by an estimated 80% over the last five decades, a rate even faster than the greater sage grouse.

Patricia Estrella, who represents the group in New Mexico, said that while population declines are well documented, the exact cause remains unclear as multiple threats are at play.

"Not only is it difficult to tease apart the effects of interacting factors, together they create even greater threats through

"Not only is it difficult to tease apart the effects of interacting factors, together they create even greater threats through positive feedbacks," Estrella wrote in the petition. "Successful conservation of the pinyon jay requires addressing and ameliorating multiple threats simultaneously."

Piñon-juniper forests cover more than 75,000 square miles (194,249 square kilometers) in the United States, and wildlife managers in Arizona, Colorado, Idaho, Montana, Nebraska, Nevada, and New Mexico already have classified the bird as a species of greatest conservation need.

Nearly 60% of the jay's remaining population can be found in New Mexico and Nevada, but its range also includes central Oregon and parts of California, Utah, Wyoming, Oklahoma and Mexico's northern Baja California.

Pale blue with a white bib, the pinyon jay typically mates for life. When food is abundant, they can nest more than once a year.

Their home range can be large, with the birds fanning out over hundreds of miles when food is scarce.

Research highlighted in the petition notes that more piñon and juniper woodlands are being cleared around the West for housing developments, agriculture, and solar and wind energy projects, and as land managers look to reduce the threat of wildfire.

The Biden administration's infrastructure push includes $500 million over five years for prescribed burns, $500 million for mechanical tree harvesting and another $500 million for clearing fuel breaks. Defenders of Wildlife and others are concerned that managers will be able to move ahead with many projects without public input or more detailed environmental reviews.

The petition states that studies documenting the effects of woodland reduction on pinyon jay populations are few, but some scientists are recommending that land managers avoid nesting and foraging sites.

More: These remarkable species were saved from the brink of extinction

When the piñon crop is good, jays start the morning by eating seeds. Then they collect seeds, congregate in the tree tops and depart together to caching areas that are usually open spaces where less snow accumulates in the winter.

Research has shown they have excellent memory and recover more cached seeds than other types of jays or nutcrackers, but the seeds that go uncollected are left to germinate.

According to the petition, the loss of piñon trees would disproportionately affect Native American and Hispanic communities in the Southwest. Each fall, families make the trek to the forest to harvest the seeds.

The nuts are usually roasted and salted, but their popularity now extends beyond Southwest cuisine, including

The nuts are usually roasted and salted, but their popularity now extends beyond Southwest cuisine, including pancakes, brownies, pizza and salad toppings.

It will be up to the Fish and Wildlife Service to determine whether there's enough information in the petition to warrant further review. That could take months.

Cattle country

The West's landscapes have been indelibly altered by livestock.

BY JONATHAN THOMPSON
DATA VISUALIZATION BY JENNIFER DI-MAJO

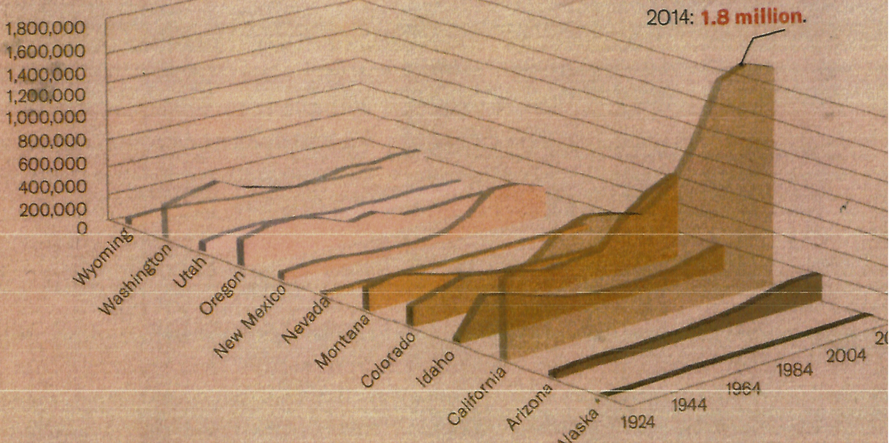
IN THE MID-TO-LATE 1800s, well-financed livestock operations drove tens of thousands of cattle onto the "public domain" — i.e., onto the lands stolen from Indigenous people in the Interior West, where the grass grew as high as a pony's belly and appeared to be free for the taking. The livestock industry, along with mining, soon dominated the region's colonial-settler culture, economy and politics.

By the end of the century, however, the big cattle drives were becoming a thing of the past. In the ensuing decades, ranches gave way to energy fields and suburban sprawl, and the industry's economic power faded. And yet, the West is still Cattle Country: The cowboy myth endures, fueling tourism. Ranching wields an outsized influence over state and federal politics. And the cattle themselves are still here, millions of them, squeezed into massive feedlots, scattered across public lands and pumping out milk in industrial-scale dairies.

More of the region's irrigation water and farmland goes to alfalfa and other livestock feed than to any other crop. Cows are walking, cud-chewing methane dispensers, creating massive "hotspots" of greenhouse gas above overcrowded feedlots. And they continue to roam the West's public lands, decimating grasslands, facilitating the spread of noxious weeds, destroying cryptobiotic crusts, trampling riparian areas and fouling desert streams.

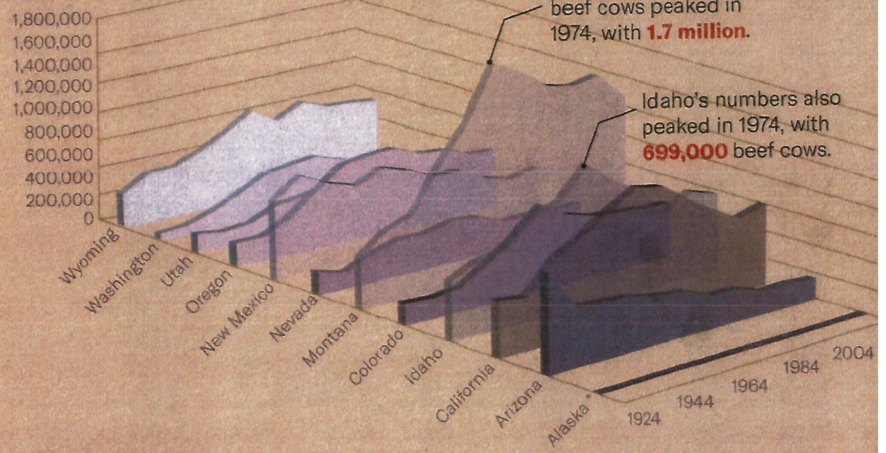
In 1965, Arizona researchers found that cattle grazing in the Sonoran Desert had caused a "shift in the regional vegetation of an order so striking that it might be better associated with the oscillations of Pleistocene time than with the 'stable' present." The landscape has been so altered by livestock that we can barely imagine what it looked like before the herds arrived. Forget the Anthropocene; the West is still stuck in the Beefocene.

Number of milk cows by decade



With a steady incline, California had the largest number of milk cows in 2014: **1.8 million**.

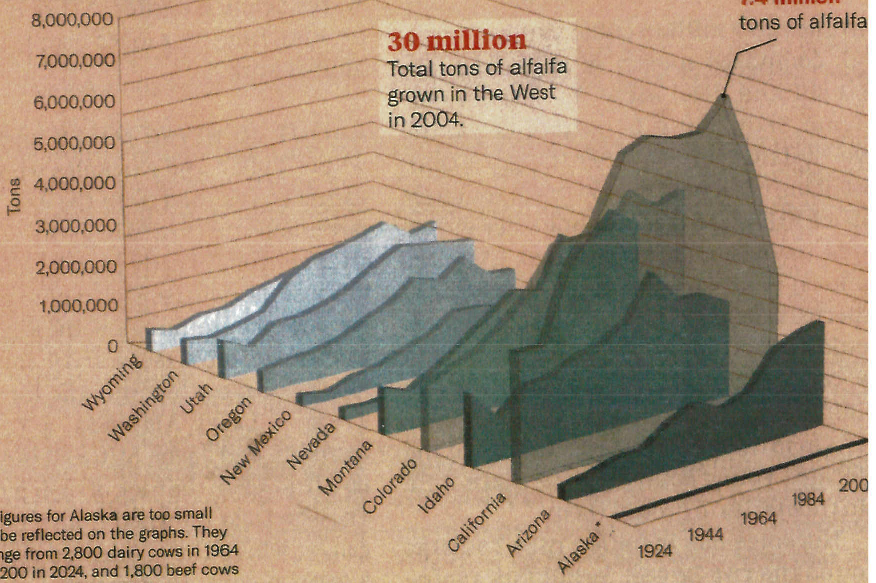
Number of beef cows by decade



Montana's number of beef cows peaked in 1974, with **1.7 million**.

Idaho's numbers also peaked in 1974, with **699,000** beef cows.

Alfalfa grown by decade



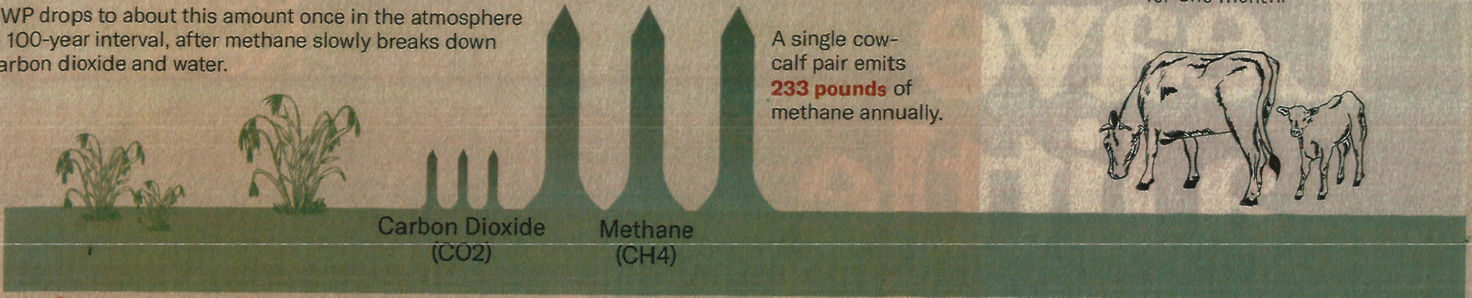
30 million
Total tons of alfalfa grown in the West in 2004.

In 2004, California gre
7.4 million
tons of alfalfa

* Figures for Alaska are too small to be reflected on the graphs. They range from 2,800 dairy cows in 1964 to 200 in 2024, and 1,800 beef cows in 1964 to 8,400 in 2004.

6
obal warming potential (GWP) of methane over a 20-year interval, meaning s 86 times more potent than carbon dioxide in the near term.

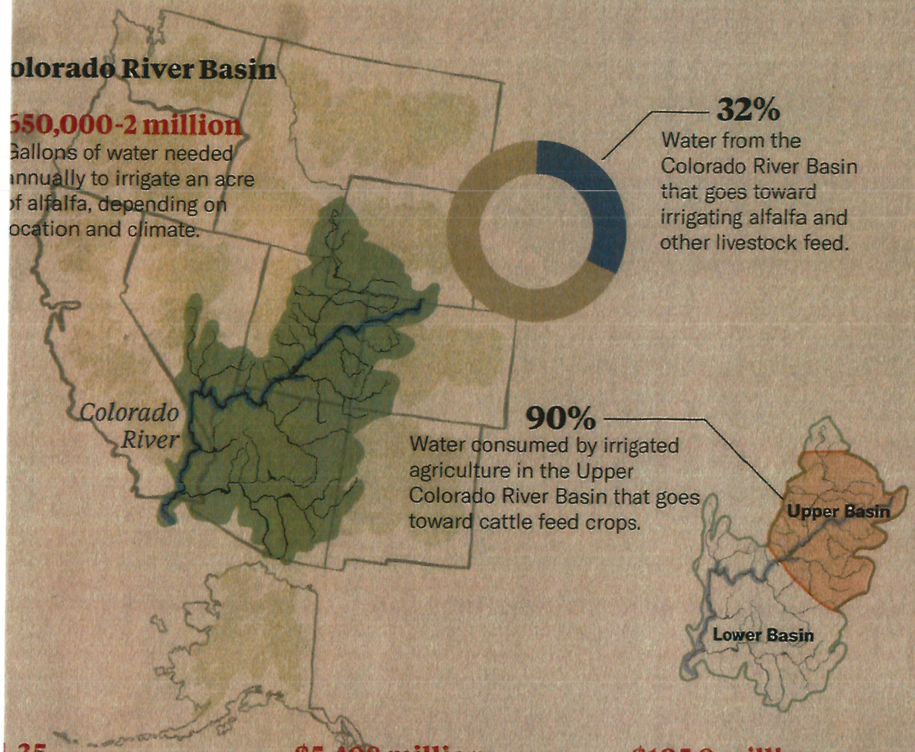
4
e GWP drops to about this amount once in the atmosphere er a 100-year interval, after methane slowly breaks down o carbon dioxide and water.



A single cow-calf pair emits **233 pounds** of methane annually.

31.3 million acres
Minimum amount of land in the Western U.S. dominated by cheatgrass, a noxious, **fire-prone** weed spread by grazing, as of 2000.

123.5 million
Tons of carbon lost to the atmosphere as of 2000 due to the conversion of native rangelands to cheatgrass in the Wyoming big sagebrush biome.



olorado River Basin

550,000-2 million
Gallons of water needed annually to irrigate an acre of alfalfa, depending on location and climate.

32%
Water from the Colorado River Basin that goes toward irrigating alfalfa and other livestock feed.

90%
Water consumed by irrigated agriculture in the Upper Colorado River Basin that goes toward cattle feed crops.

.35
azing fee per AUM on BLM nd in 2024 and the previous veral years, meaning that's w much it costs a rancher keep one cow and calf public land for a month, ring which they'll consume 0-to-1,000 pounds of age. This is the minimum ount Congress allows the M to charge.

-$12 Administrative st per AUM to manage estock on public lands.

$5.498 million
Amount that industry, including livestock lobbying groups, donated to Frank Mitloehner, a UC Davis animal science professor who downplays cattle's contribution to climate change.

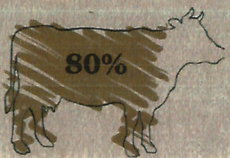
$36
Social cost of greenhouse gas —the estimated cost of damage done to the climate — for one AUM on Western public lands.

$105.9 million
Amount budgeted to the Interior Department for rangeland management in 2020, meaning taxpayers are subsidizing grazing operations to the tune of $90 million per year.

$2.5 billion
Total amount of federal conservation, disaster, commodity and crop insurance subsidies paid to ranchers and farmers in the 11 Western states between 1995 and 2020.

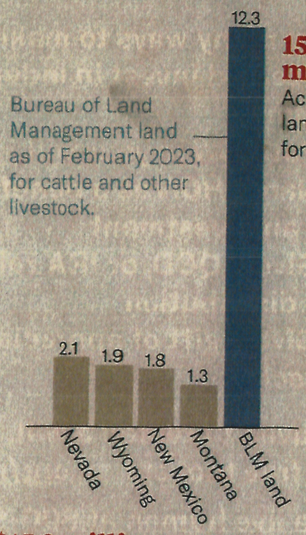
Animal Unit Month (AUM)

Forage required to feed one 1,000-pound cow-calf pair grazing on public land for one month.



Percentage of a cow's body weight the animal will eat in one month. This can increase in desert environments, where cows must walk farther to reach forage.

Number of active AUMs (in millions)



Bureau of Land Management land as of February 2023, for cattle and other livestock.

155 million
Acres of BLM land allotted for grazing.

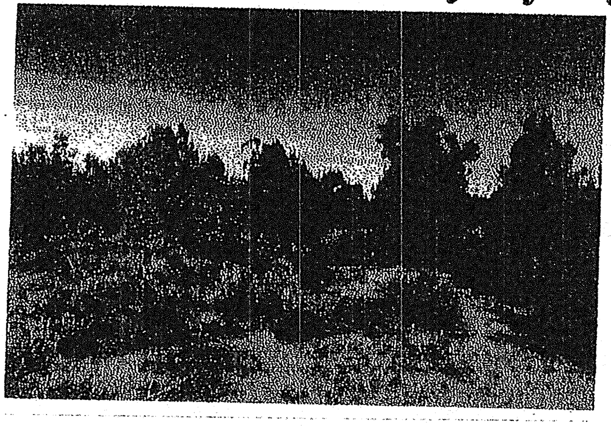
$15.9 million
Revenue the BLM received from grazing fees for all livestock categories in 2020.

SOURCES: U.S. Department of Agriculture, Bureau of Land Management, Bureau of Reclamation, Environmental Working Group, Environmental Protection Agency; "Water Scarcity and Fish Imperilment Driven by Beef Production," by Brian Richter, et al.; "The animal agriculture industry, US universities, and the obstruction of climate understanding and policy," by Viveca Morris and Jennifer Jacquet; "Livestock Use on Public Lands in the Western USA Exacerbates Climate Change: Implications for Climate Change Mitigation and Adaptation," by J. Boone Kauffman, et al.

High Country News Vol. 56, May 2024

The Salt Lake Tribune

The juniper mystery: Why is a tree that's supposed to withstand drought suddenly dying in southern Utah?



(Courtesy photo | Kate Magargal, University of Utah) Scientists and federal land managers are scrambling to investigate widespread juniper mortality in southeast Utah. Researchers shot this photo last month on Cedar Mesa, where junipers started dying in droves in September after the driest and warmest year on record. Foresters are stumped because juniper is seen as a drought-tolerant species.

By Brian Maffly • Published: November 18
Updated: November 20, 2018

The Utah juniper is considered the West's most drought-tolerant and resilient conifer, withstanding even the worst dry spells while nearby pinyon and ponderosa drop their needles and die.

So it was with some alarm for Kay Shumway, a retired science educator from Blanding, when he noticed yellowing among the juniper on southeastern Utah's Moki Dugway last spring, a time of year when these trees' needles should carry a vivid shade of green as they ramp up their photosynthetic capacity.

"During summer, the dying junipers were such a bright yellow color. It was easy to see. You could look out over the landscape and see thousands of those dying trees. Now a lot have dropped their needles and are becoming a skeleton," said Shumway, a former botanist who has kept himself busy as a nature photographer since retiring from the College of Eastern Utah 20 years ago. "They are dying. They are not going to come back [even if they get] some moisture."

He frequents the Moki Dugway on the southern tip of Cedar Mesa, which provides photogenic views of the buttes and canyons falling toward the San Juan River in what was initially part of Bears Ears National Monument. Shumway has since documented dying juniper in other parts of San Juan County, which is in the midst of a severe drought.

"My first reaction was the drought was causing it. The interesting thing is the pine trees, which are the most susceptible to drought, aren't affected. They are still green and healthy," Shumway said. "There were more and more of [the juniper] turning yellow. By July, I went all over the county, like Mustang Mesa and Alkali Ridge [east of Blanding]. Lo and behold, it was happening there, too."

Shumway has shared his photos and drone footage with academic scientists and forestry officials, who are now scrambling to determine the extent and causes of Utah's latest round of tree mortality.

The Colorado Plateau just experienced its driest year on record. Bluff saw barely 2 inches of precipitation in the water year that ended Sept. 30, about a fourth of normal, according to a National Weather Service database.

But juniper should be able handle a dry year.

"You can't look at the weather data for one year. Often the stress from drought is lagged," said entomologist Liz Hebertson. "We would have to look at conditions back three years, six years, 10 years. When a tipping point occurs might be different between species and even individual trees."

Other factors, working in combination with climate change, are probably at play, according to Hebertson, who works with the U.S. Forest Service's Forest Health Protection program in Ogden.

"It could be the perfect storm of other factors," Hebertson said. "Right now, we have a lot of questions."

Like what is the extent of the mortality? What are the elevation profiles, soil conditions and other environmental characteristics of the affected areas? Are there signs of beetle infestation? Is the wood discolored? Are other tree species showing signs of distress?

Utah is no stranger to conifer die-offs. Dixie National Forest outside Cedar City is ground zero in a beetle epidemic that wiped out spruce, while lodgepole pines are falling victim to beetles in the Uinta Mountains. Ghost forests stretch across basins and plateaus for miles around the West.

Hebertson and her colleagues have been preoccupied this year with a new infestation afflicting forests in northern Utah, where a nonnative beetle, the balsam woolly adelgid, is decimating subalpine fir. This tree mortality is just as troubling as the one unfolding in San Juan County.

In search of clues in the juniper deaths, Hebertson is leading numerous scientists on a tour near Blanding later this month, along with Shumway, who wishes the group came earlier while the yellowed foliage was still hanging from the juniper branches.

“I’m thrilled to know there is some work being done on it,” Shumway said. “I’m going to have this group get up close and personal with these trees, though it won’t be as easy to see. You can still see there are dying trees all over the place.”

Since retiring from CEU (now part of Utah State University), where he served as dean of instruction after helping to establish the Blanding campus in 1977, Shumway has explored Bears Ears country with cameras in tow. While stressed junipers have been documented across the state line in Colorado since last year, Shumway was the first to systematically record the phenomenon in Utah.

Almost more a shrub than a tree, Utah junipers crowd valley bottoms and mesa tops all over the Beehive State. They often mistakenly were called cedars in pioneer times; hence the appearance of the word “cedar” in so many Utah place names.

Junipers, along with the pinyons they usually grow with, are the subject of vegetation-removal operations in the name of habitat improvement. But these trees play a vital ecological role, stitching landscapes together and providing habitat for countless animals.

Widespread loss of junipers would have far-reaching consequences for southern Utah’s fragile desert environments.

Among those taking a keen interest is University of Utah biology professor Bill Anderegg, who studies the impact of climate change on forest ecosystems.

“What we see is worrying and it’s mind-boggling because juniper has weathered drought. In the drought of 2002, they were not affected,” Anderegg said. “To see juniper die on a landscape scale is pretty disturbing. The pinyon don’t seem to be dying at the same rate.”

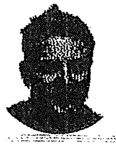
A possible factor is the Western cedar borer or cedar bark beetle, insects known to attack juniper. Shumway has pulled bark off dead trees to find larvae munching on the phloem, the critical tree tissues that conduct the fruits of photosynthesis to the root system.

But the presence of beetle larvae is just a clue, not a complete causal picture.

Last year was not only the driest on record for Utah, but it also was the second warmest. Could warm weather be compromising the trees' health, undermining their ability to repel beetle attacks? Or could it be enabling populations of hungry beetles to explode?

"We want to know how bad is the mortality event, how prevalent it is across the Four Corners, pin down the drivers," Anderegg said. "What are the thresholds that push junipers over the edge, and why are those thresholds being crossed now?"

Anderegg has applied for a Forest Service grant to explore such questions. At this stage, however, the extent of juniper mortality has yet to be established, much less why it might be happening, although few doubt climate change is playing some role.



bmaffly@sltrib.com
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Sage Thrasher

Priority Level: II MT Score: 19 AI: 3

Reason for Concern. This species is a sagebrush obligate. Although population trends of Sage Thrashers are currently stable, the conversion and fragmentation of big sagebrush habitat continues. Continued loss of habitat will likely result in declining populations.

Distribution. Sage Thrashers winter from southern California, Arizona, and New Mexico to central Mexico (Bent 1948). They breed throughout parts of southcentral and eastern Montana (Bergeron et al. 1992).

Habitat Requirements. Sage Thrashers are sagebrush obligates, occupying shrub-steppe communities dominated by big sagebrush. In microhabitat studies, the size and spatial distribution of sagebrush appear to be the most important variables in nest-site selection. Nests are most often placed in tall, large sagebrush plants that may provide support and concealment for the thrasher's conspicuous nests. They also tend to nest in areas where there are dense clumps of large sagebrush shrubs (Petersen and Best 1991). The presence of thrashers is positively correlated with sagebrush and shrub cover, shrub patch size, bare ground, and negatively correlated with spiny hopsage, budsage, grasses, and exotic plant species (Wiens and Rotenberry 1981, Knick and Rotenberry 1995). Bare ground near nests may provided nearby foraging opportunities (Petersen and Best 1991).

Dominant Plant Species Composition. The Sage Thrasher is almost always associated with shrubsteppe communities dominated by big sagebrush. In the northern Great Basin, breeding and feeding habitats are primarily tall sagebrush/bunchgrass, juniper / sagebrush / bunchgrass, mountain mahogany/shrub, and aspen/sagebrush/bunchgrass communities (Maser et al. 1984). In central Montana, Sage Thrashers were found in areas dominated by big sagebrush, and other dominant shrubs such as broom snakeweed and rubber rabbitbrush, which occurred at lower densities (Feist 1968).

Vegetation Physiognomy/structure. Sage Thrasher nest-site selection is very specific within sagebrush stands. Nests are almost always located either within or beneath sagebrush shrubs. Most nests are located in shrubs with a high foliage density (75-100% live) and many branches within 30 cm of the ground. Nests located on either the ground or a branch are usually placed toward the main axis of the shrub (Petersen and Best 1991). The density of foliage shading or covering a nest appears to be important in nest placement (Reynolds and Rich 1978). Nests are usually oriented southeast, perhaps to maximize morning sun and afternoon shade, and for protection from prevailing winds (Petersen and Best 1991). In Oregon, Sage Thrashers were not present in habitats dominated by crested wheatgrass or annual grasses and forbs, but may be found once sagebrush covers 2% to 5% of the area (Pers. comm. from A. Baumman to C. Paige).

Amount of Habitat. Few data are available. Territory size averaged 0.96 ha in Idaho (Reynolds and Rich 1978). In Montana, Thrasher nesting density averaged a low of 2.5 breeding pairs/100 ha (Feist 1968). Sage Thrasher densities during the breeding season in the Great Basin have been as high as 30 individuals/100 ha (Wiens and Rotenberry 1981) and 40 individuals/100 ha (Medin 1992). Manage for maintaining large patches (100 ha) of big sagebrush.

Adjacent Habitat Matrix. Concern for the introduction and spread of noxious weeds that may compete with native vegetation and thus, decrease the suitability of thrasher nesting habitat. The effects of fragmentation and, therefore, include a potential increase in predation of adults and nest contents.

Abiotic factors. Sage Thrashers are usually found nesting at elevations between 1300-2000 m (Bent 1948).

Management Issues. Large, contiguous stands of tall, dense clumps of big sagebrush interspersed with native bunchgrasses and forbs are vital. Some studies have shown that fragmentation and alteration of sagebrush stands have negative impacts on Sage Thrashers. Areas where crested wheatgrass has been planted as livestock forage have experienced a significant reduction in the number of Sage Thrasher nests (Reynolds and Trost 1980). Alteration and destruction of sagebrush habitat promotes the establishment of exotic grasses, primarily cheatgrass. The dominance of cheatgrass encourages annual wildfires, which converts shrubsteppe to annual grasslands (Knick and Rotenberry 1995). Edge effects due to fragmentation may increase the potential for predation and parasitism. Rich and Rothstein (1985) documented Thrashers ejecting eggs of Brown-headed cowbirds from their nests.

- Loss and fragmentation of Sage Thrasher habitat through conversion and alteration of big sagebrush stands is a major threat.
- Livestock grazing can have a positive or negative effect depending on the plant community composition, timing, and duration.
- Prescribed burning can have a negative effect (particularly in the short term) if there is significant shrub canopy cover reduction. There can be some positive long-term effects if the fire is cool and spotting, and creates openings amid really dense sagebrush stands.
- Off road vehicle use can negatively impact nest sites as well as plant species composition (spread of noxious weeds).

Research/Monitoring Needs: Due to the lack of data on Sage Thrashers in Montana, steps should be taken to gather baseline information on habitat use, quality and amount of big sagebrush habitat, and a fine scale systematic survey of breeding birds.

Population Objective. Analysis of Breeding Bird Survey Data indicates Sage Thrasher population trends are stable in Montana and the western region. Our objective is to maintain this stability over time.

Back to PIF Plan Table of Contents

Loggerhead Shrike

Priority Level: II MT Score: 17 AI: 3

Reason for Concern. This species has shown declines throughout much of its range, although populations in Montana appear to be stable (PT/PTU = 2/2). Because of their predatory nature and use of open habitats, they are highly susceptible to pesticides on both their breeding and wintering areas. Continent-wide declines have been attributed to both the use of pesticides and breeding habitat losses (Yosef 1996). The relative reproductive success of shrikes nesting in grassland and agricultural habitats (e.g. windrows), and of those nesting in sagebrush habitats (notably in south west Montana) is unknown.

Distribution. Loggerhead Shrikes breed throughout the southern United States, from Florida to California, and north throughout the great plains into the southern prairie provinces of Canada (Price et al. 1995). It is a confirmed breeder in drier, open habitats throughout much of Montana east of the continental divide (Montana Bird Distribution Committee 1996). Redmond et al. (1998) predicted there are over 10 million ha of potential habitat for this species in the state.

Habitat Requirements. Loggerhead Shrikes nest in a wide variety of open habitats, as long as woody nesting strata (often thorny shrubs) are available. These may include grassland prairie with scattered trees, riparian areas or woody draws, cultivated lands with shelterbelts, or even badlands with few shrubs, in addition to the sagebrush shrubland and shrubsteppe habitats considered here. Research in Idaho has shown shrikes to have equivalent nesting success in sagebrush, bitterbrush, or greasewood, although 65% of the nests were in big sagebrush (Woods 1993, Woods and Cade 1996). Nests are usually located well within the structure of shrubs 1-2m tall, but almost always within 1m of the ground (mean 79 cm, Woods and Cade 1996). Various studies in Alberta (Prescott and Collister 1993, Telfer 1993) have shown local populations of this species to be reliant on shortgrass, mid-grass, or tall grass, so clearly there is a great deal of variation in preferred habitat, and indeed grass height may not play a role nest selection.

Ecology. The Loggerhead Shrike is a predatory bird which relies primarily on grasshoppers and beetles during the nesting season, but also takes a wide variety of mammal, bird and reptile prey (Yosef 1996). They prefer to hunt in areas with sparse vegetation, from an exposed perch, and often utilize roadside wires. Larger prey is often secured on thorns, broken branches or barbed wire to facilitate feeding.

Management Recommendations. The habitat needs of Loggerhead Shrikes can likely be met by providing for a suitable distribution of large sagebrush plants (1-2m tall), interspersed with open habitats for feeding, and by controlling the application of pesticides in known nesting areas. Additional data should be collected to delineate the distribution of the species in the state, particularly in shrubsteppe and grassland habitats, and to determine relative nesting success in the various habitats used by the species.

Back to PIF Plan Table of Contents

Montana Partners in Flight | **Bird Conservation Plan** | Table of

Brewer's Sparrow

Priority Level: II MT Score: 20 AI: 3

Reason for Concern. The nominate form of this species is a sagebrush obligate which has shown significant population declines throughout much of its range, including the two physiographic areas which overlap Montana. Very little is known about the distribution and habitat needs of the "Timberline" form of this species in the state.

Distribution. There are scattered breeding records throughout the State with only three latilongs not represented with evidence of breeding (Montana Bird Distribution Committee, 1996). Most suitable habitats are concentrated in the southern half of the State with few sightings in the northwest and north of the Missouri River. Gap (Redmond et al. 1998) modeling predicted four million ha of suitable habitat (or 11% of the state) for the species within Montana. The "Timberline" (Brewer's) Sparrow is a subspecies that is found breeding in high elevation shrubfields and krumholz. Within Montana, they have been located on the east side of the divide in Glacier National Park.

Habitat Requirements. Brewer's Sparrow is a species characteristically found within sagebrush habitat (Rising, 1996), and indeed is considered by most to be a sagebrush obligate (but see Timberline Sparrow). Both Johnsgard (1979) and Saab and Rich (1997) felt that Brewer's Sparrow is closely associated with shortgrass prairie with scattered to abundant amounts of sagebrush or other shrub-steppe vegetation. Others have shown a negative correlation with grass cover, with the species preferring dense sagebrush stands (Rotenberry and Wiens 1980). Best (1970) found Brewer's Sparrow utilizing sagebrush as important nesting cover; Bock and Bock (1987) found they preferred unburned to burned sagebrush. One Idaho study indicated that large patch size and robust shrub cover both increase the likelihood of use by this species (Knick and Rotenberry 1995). It is often the most common breeding bird where it occurs.

Ecology. This species feeds on insects found in the foliage of sagebrush and on the ground, and on the seeds of grasses and forbs. It nests in large, living sagebrush, rarely using shrubs <50 cm tall (Peterson and Best 1985). The nests are near the ground, and are usually placed in the finest branches of new growth near the tips of the branches (Rich 1980), so shrubs in good vigor are important to nesting. They show strong site fidelity, returning from year to year to nest in the same area (Wiens and Rotenberry 1985).

Management Issues. The species is vulnerable to parasitism by Brown-headed Cowbirds, especially where the sagebrush landscape has been broken up by agriculture and pastures. Reductions in sagebrush cover and vigor from control actions such as burning or herbicides will reduce or eliminate habitat suitability for the species. The long-term viability of the species in Montana will depend on the maintenance of large stands of sagebrush in robust condition throughout the species' range in the state. Wide distribution of suitable habitat is essential, due to their tendency toward site fidelity.

Management Recommendations. Implementing recommendations for SageGrouse should encompass all the needs of Brewer's Sparrows.

J. Range Manage.
45:257-262, May 1992

Interactions of grazing and plant protection on basin big sagebrush *(Artemisia tridentata* ssp. *tridentata)* seedling survival

M.K. OWENS AND B.E. NORTON

Authors are with Texas A&M University Agricultural Research and Extension Center, 1619 Garner Field Road, Uvalde 78801; and the Department of Range Science, Utah State University, Logan 84322. At the time of the research Owens was with the Department of Range Science, Utah State University, Logan 84322.

Abstract

The impact of grazing animals and plant protection on shrub seedling establishment was studied in 2 separate experiments. A total of 3,665 seedlings were monitored for survival during a sheep grazing trial in 1984, and 5,755 seedlings were monitored during a cattle grazing trial in 1986. Approximately 1/2 of the seedlings were located under the canopy of mature plants and 1/2 were located in the interspaces between plants. The presence of domestic livestock and the seedling location affected both the overall survival at the end of the growing season and the pattern of survival during the growing season. The interaction between these independent variables resulted in the highest survival (0.11) for sheltered seedlings in the grazed pastures and the lowest survival (0.009) for unprotected seedlings in the grazed pastures. Seedlings in the ungrazed pastures had survival rates intermediate between these 2 rates.

The pattern of seedling survival was similar in both experiments. Seedlings in the grazed pastures experienced high mortality during the actual grazing event and immediately after grazing. Seedlings which were unsheltered experienced the lowest survival due to trampling. Survival rates late in the summer were not affected by grazing but were dependent on receiving precipitation during this normally dry period of the year.

The interaction between grazing and seedling location may partially explain the aggregated distribution of *Artemisia* found in many communities. This aggregation should affect interspecific competition and may play a role in later stages of plant succession within these shrub-dominated communities.

Key words: seedling survival, sagebrush

The increase of shrub populations in grassland communities is a widespread occurrence in a variety of ecosystems (van Vegten 1983, MacDonald and Frame 1988, Archer et al. 1988). Shrub establishment does not occur as an advancing front from undisturbed shrub stands but seems to occur from isolated shrubs remaining in the grass stand. Two examples are basin big sagebrush *(Artemisia tridentata* spp *tridentata)* which often invades pastures seeded to grasses such as crested wheatgrass *(Agropyron desertorum)* from shrubs which were not killed prior to seeding the grass (Marlette and Anderson 1986), and the shrub *Baccharis pilularis* which invades the annual grasslands of California (Williams et al. 1987).

Native sagebrush is a long-lived shrub which has little value for grazing herbivores. Vast areas associated with long-term heavy grazing are now dominated by basin big sagebrush communities, with little forage available for domestic livestock (Young et al. 1990). Periodic treatment of these communities with herbicides or prescribed fires can successfully reduce the dominance of basin big sagebrush and promote herbaceous forage production, but regeneration of the shrub population usually occurs. Understanding the population biology of the expanding shrub population is a critical step in successfully managing these rangelands for cattle production.

50 milli
seeds

Basin big sagebrush establishment does not seem to be limited by seed production, although dispersal distance is usually short (Young 1988). An average stand of basin big sagebrush can produce 50 million seeds per ha annually and a plant can begin producing seed in its second year (Young et al. 1989). Harniss and McDonough (1976) found that basin big sagebrush seed germination rates were high enough to exclude seed germination as a limiting factor in the establishment process for the 3 consecutive years of their study. Establishment remains as an episodic event (Young et al. 1989, West et al. 1979).

Environmental factors, especially precipitation, competition, and seedbed factors, play an important role in the growth of shrub populations. Therefore, factors that alter the environment of individual seedlings may also alter the probability of survival and successful establishment. Factors that could affect seedling survival include grazing by large herbivores, seedling location in relation to other plants, and the interaction between these factors. Grazing may actually promote the establishment of an unpalatable species such as basin big sagebrush by reducing competition from the preferred forage species. Seedling location relative to other plants may affect seedling establishment by either protecting the seedling from grazing or by altering the microenvironment.

This study was designed to answer the following questions concerning the expansion of basin big sagebrush populations into crested wheatgrass communities: (1) Does grazing by domestic livestock increase the probability of seedling survival? (2) Does the location of the seedling relative to mature conspecific plants affect seedling survival? and (3) What is the combined effect of grazing and seedling location on overall survival and the pattern of survival during the growing season?

Materials and Methods

The study was conducted on the Tintic Research Area (elevation 1,830 m) located in the Intermountain region of the western United States (39° 53'N and 122° 03'W). The study pastures within the Research Area were mechanically treated for brush removal in the early 1950's and seeded to crested wheatgrass. Mean annual precipitation is 374 mm with 75% falling from October to May as snow (Wraith et al. 1987). Summer precipitation is highly variable in both intensity and location. Precipitation during the first year of

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This research was supported by the Utah Agricultural Experiment Station, Utah State University, Logan 84322-4845. Approved as journal paper 3456. The authors wish to thank Drs. D. Pyke and C.A. Call for critical review and helpful suggestions to improve this manuscript.

Manuscript accepted 9 August 1991.

Table 3. Probabilities of seedling survival in sheltered and unsheltered microsites during a 110 day sheep grazing trial.

		Days after emergence										
		10*	20	30	40	50	60	70	80	90	100	110
Grazed												
Sheltered	$\bar{x}$	0.18	0.42	0.83	0.96	0.94	0.86	0.74	0.83	0.85	0.91	0.99
	se	0.02	0.03	0.04	0.02	0.02	0.03	0.04	0.04	0.03	0.03	0.01
Unsheltered	$\bar{x}$	0.11	0.32	0.87	0.89	0.90	0.69	0.56	0.55	0.71	0.85	1.00
	se	0.01	0.03	0.03	0.03	0.03	0.04	0.05	0.07	0.07	0.07	0.01
Ungrazed												
Sheltered	$\bar{x}$	0.71	0.61	0.80	0.95	0.95	0.81	0.45	0.75	0.63	0.74	0.99
	se	0.02	0.02	0.02	0.01	0.01	0.02	0.03	0.04	0.04	0.05	0.01
Unsheltered	$\bar{x}$	0.68	0.56	0.90	0.92	0.89	0.81	0.55	0.52	0.59	0.81	0.99
	se	0.02	0.02	0.01	0.01	0.01	0.02	0.02	0.03	0.03	0.04	0.01

*Grazing occurred during this time period.

tered treatment (120 survivors from a cohort of 2,715) than in the unsheltered treatment (50 survivors from 3,040 seedlings).

CDA indicated the best model included all of the 2-way interactions between replications, grazing treatment, and shelter (Table 4).

Table 4. Categorical data analysis of basin big sagebrush seedlings during the cattle grazing experiment. The X^2 statistic indicates the significance of the overall model and the G^2 analysis indicates the significance of the specific model.

Effect	Significance tests of heirarchical models X^2	df	Partitioned G^2 within the best fit model G^2	df
Grazing	29.72**	4	10.77**	1
Shelter	53.22**	4	35.19**	1
Grazing × Shelter	9.71**	2	9.31**	1
Replication × Shelter	8.34**	2	8.03**	1
Replication × Grazing	13.12**	2	12.30**	1
Replication × Grazing × Shelter	0.38	1	0.40	1

* $P<0.05$
** $P<0.01$

The G^2 partitioning of the model demonstrates that all of the effects are statistically significant within this model. The interaction of replications with the other factors was due to the near total mortality found in the first replication (Fig. 1B). There were no significant differences between the logit values in the first replication and the values were generally very small. Seedlings in an unsheltered location in the grazed pastures had the lowest probability of survival while protected seedlings had the highest chances for survival in both grazed and ungrazed pastures in the second replication. Seedlings without the protection of mature basin big sagebrush were 5 times more likely to die in the grazed pasture than in the ungrazed pasture. Within grazed pastures, the difference was more pronounced, with protected seedlings being up to 6 times more likely to survive than unprotected seedlings (Fig. 1B). In the other replication, over 99% mortality in all treatments precluded the detection of treatment effects.

As with the sheep study, grazing by cattle caused differences in the pattern of mortality during the 80-day trial (Fig. 3). During the first 2-day grazing period, survival rates were significantly higher for the ungrazed seedlings in the sheltered (0.91) and unsheltered (0.85) microsites than in the same microsites in the grazed pastures (0.84 and 0.49, respectively, Table 5). During the second grazing period 18 days later, survival was much lower in all treatments but survival was significantly lower in the grazed, unsheltered treatment than in any other treatment. A dry period began 20 days after seedling emergence and continued throughout the summer (Fig. 3). Seedling survival in all treatments decreased as the dry summer progressed, but survival in the ungrazed pastures was sharply depressed during the period of 10 to 20 days after the beginning of the summer drought (40 days after emergence, Table 4).

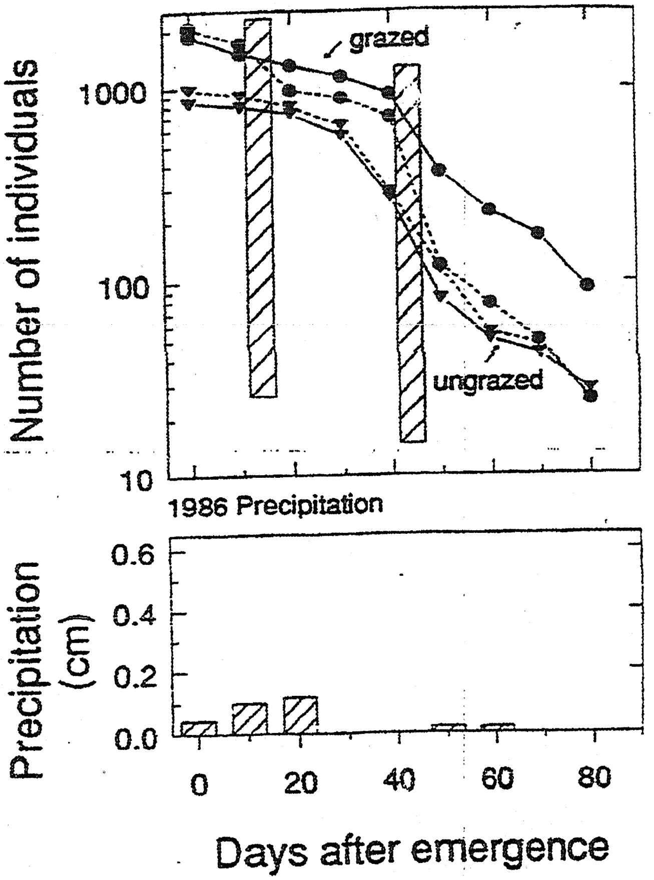

Fig. 3. Basin big sagebrush seedling survival in pastures grazed by cattle (circles) and in ungrazed controls (triangles). Seedlings were in sheltered (solid lines) and unsheltered (dashed lines) locations. Hatched areas represent the actual grazing periods of 28–30 April and 18–20 May 1986. Precipitation is presented as 10-day totals.

Discussion

Grazing by domestic livestock or native herbivores has been reported to decrease seedling survival of some shrubs and grasses (Eckert et al. 1986, Gardiner 1986, Salihi and Norton 1987) but

Probabilities of seedling survival in sheltered and unsheltered microsites during a cattle grazing trial.

		Days after emergence							
		10	20*	30	40	50*	60	70	80
Grazed									
Sheltered	$\bar{x}$	0.81	0.84	0.85	0.69	0.27	0.54	0.56	0.67
	se	0.01	0.01	0.01	0.02	0.02	0.03	0.04	0.03
Unsheltered	$\bar{x}$	0.84	0.49	0.89	0.68	0.08	0.57	0.42	0.63
	se	0.01	0.01	0.01	0.02	0.01	0.05	0.0[illegible]	0.06
Ungrazed									
Sheltered	$\bar{x}$	0.95	0.91	0.71	0.29	0.18	0.55	0.68	0.76
	se	0.01	0.01	0.02	0.02	0.02	0.06	0.09	0.06
Unsheltered	$\bar{x}$	0.93	0.85	0.76	0.26	0.27	0.40	0.76	0.65
	se	0.01	0.01	0.02	0.02	0.03	0.05	0.08	0.06

*Grazing occurred during this time period.

may have a slight positive effect on survival of other species (West et al. 1979). Grazing by either sheep or cattle sharply depressed basin big sagebrush seedling survival for a brief time after the short but intense grazing period (Fig. 2 and 3). Later in the season the remaining seedlings were less prone to mortality than seedlings in the ungrazed pastures. The effects of grazing, evident early in the season, were masked at the end of the season by the late summer moisture-dependent mortality. The net result was no difference in overall seedling mortality between the grazed and ungrazed pastures at the end of the growing season. The seedlings that were not killed from trampling during the grazing period experienced high mortality rates from another source, late summer drought. This is in direct contrast to crested wheatgrass seedlings in the same pastures as this study (Salihi and Norton 1987). The effects of grazing on crested wheatgrass seedling survival, following a similar 2-day grazing period, an 18-day rest and an additional 2-day grazing period, was evident up to 1 year after grazing. For both crested wheatgrass and basin big sagebrush, the individual seedlings were too small to be grazed but were subjected to trampling. Almost 90% of the crested wheatgrass seedlings occurred in the interspaces rather than under mature plants and may have been more vulnerable to trampling than the basin big sagebrush seedlings in this study. Only 60% of the basin big sagebrush seedlings in this study were in the interspaces so potential trampling would have been less. Balph et al. (1989) state that vegetation cues are used by cattle to avoid trampling on mature plants, therefore seedlings in the interspaces would be at a higher risk of mortality.

The differential seedling survival rates caused by the sheltered and unsheltered microsites and the presence or absence of grazing may partially explain the aggregated plant distribution found within some semiarid communities (Brown and Archer 1989, Archer et al. 1988). The short dispersal distance (generally<1 m) and favorable microenvironment found beneath mature basin big sagebrush shrubs are also major factors causing aggregation (Young 1988). However, survival rates of seedlings were not affected by the microsites in this study unless the pastures were grazed. The beneficial attributes of increased water holding capacity and nutrient status (Young 1988) did not change survival rates of first-year seedlings. The combined effect of microsites and grazing emphasized the aggregated dispersion pattern. Another arid land shrub, *Gutierrezia microcephala*, also demonstrates an aggregated dispersion pattern which may be related to differential survival rates for seedlings in various microsites (Parker 1982). Survival rates of *Gutierrezia* seedlings were generally higher in the vicinity of adult plants than in the interspaces. Herbivores were also a very important influence on survival and distribution of shrub seedlings in that system.

Parker (1982) reported that large seedlings of *Gutierrezia microcephala* were more likely to die from herbivory and small plants were likely to die from summer desiccation. Since basin big sagebrush seeds were small and emergence was synchronous in this study, with all seedlings emerging within 1 week, plant size was very similar and could not affect the observed survival rates. Grazing sharply reduced the number of seedlings in the grazed pastures, but desiccation resulting from summer drought caused late summer mortality (Fig. 2). In 1986, the lack of late summer rains in August had a marked effect on mortality rates compared to the steady decline in seedling numbers through the relatively moist late summer of 1984 (Fig. 3). This would support the hypothesis that the timing of precipitation is as important as the total amount (Hennessy et al. 1984). Ungrazed annual grasses in California transpired water in the top portion of the soil and prevented *Baccharis pilularis* shrubs from becoming established unless spring precipitation was relatively high (Williams and Hobbs 1989). The high basin big sagebrush survival noted for sheltered seedlings in the grazed pastures may be affected by this phenomenon. We hypothesize that the heavily grazed grasses in this study were transpiring less water than the ungrazed grasses and that consequently more soil moisture was available for the basin big sagebrush seedlings. Grazed crested wheatgrass plants did not use water as quickly as ungrazed plants on this study site, which would have maintained a more favorable soil water status in the grazed pastures (Wraith et al. 1987).

If grazing is limiting seedling distribution to areas under the protection of mature plants, at some point in the future intraspecific competition may limit either growth or survival. Although the water-holding capacity of the soil is higher under the canopy of basin big sagebrush shrubs, the potential competition between mature shrubs and seedlings is great (Young 1988). Competition may be expressed early, as with *Cirsium vulgare* which can inhibit seedling growth with litter from mature plants (de Jong and Klinkhamer 1985), or may not be expressed for several years. The aggregated distribution of juvenile basin big sagebrush found on this site suggests that intraspecific competition may play a more important role between juveniles rather than seedlings (Owens 1987). The resultant aggregation of seedlings and juveniles in the grazed pastures may affect intraspecific competition as the plants become larger. Density-dependent mortality of juvenile sagebrush may become an important factor regulating population size and hence establishment rates of shrubs into grasslands.

Literature Cited

Archer, S., C. Scifres, C.R. Bassham, and R. Maggio. 1988. Autogenic succession in a subtropical savanna: Conversion of grassland to thorn woodland. Ecol. Mono. 58:111-127.

Balph, D.F., M.H. Balph, and J.C. Malechek. 1989. Cues cattle use to avoid stepping on crested wheatgrass tussocks. J. Range Manage. 42:376-377.

1 meter dispersal

Cite as: K. V. Rosenberg *et al.*, *Science*
10.1126/science.aaw1313 (2019).

Decline of the North American avifauna

Kenneth V. Rosenberg[1,2]*, Adriaan M. Dokter[1], Peter J. Blancher[3], John R. Sauer[4], Adam C. Smith[5], Paul A. Smith[3], Jessica C. Stanton[6], Arvind Panjabi[7], Laura Helft[1], Michael Parr[2], Peter P. Marra[8]†

[1]Cornell Laboratory of Ornithology, Cornell University, Ithaca, NY 14850, USA. [2]American Bird Conservancy, Washington, DC 20008, USA. [3]National Wildlife Research Centre, Environment and Climate Change Canada, Ottawa, ON K1A 0H3, Canada. [4]Patuxent Wildlife Research Center, United States Geological Survey, Laurel, MD 20708-4017, USA. [5]Canadian Wildlife Service, Environment and Climate Change Canada, Ottawa, ON K1A 0H3, Canada. [6]Upper Midwest Environmental Sciences Center, United States Geological Survey, La Crosse, WI, USA. [7]Bird Conservancy of the Rockies, Fort Collins, CO 80521, USA. [8]Migratory Bird Center, Smithsonian Conservation Biology Institute, National Zoological Park, PO Box 37012 MRC 5503, Washington, DC 20013-7012, USA.

***Corresponding author. Email: kvr2@cornell.edu**

†Present address: Department of Biology and McCourt School of Public Policy, Georgetown University, 37th and O Streets NW, Washington, DC 20057, USA.

Species extinctions have defined the global biodiversity crisis, but extinction begins with loss in abundance of individuals that can result in compositional and functional changes of ecosystems. Using multiple and independent monitoring networks, we report population losses across much of the North American avifauna over 48 years, including once common species and from most biomes. Integration of range-wide population trajectories and size estimates indicates a net loss approaching 3 billion birds, or 29% of 1970 abundance. A continent-wide weather radar network also reveals a similarly steep decline in biomass passage of migrating birds over a recent 10-year period. This loss of bird abundance signals an urgent need to address threats to avert future avifaunal collapse and associated loss of ecosystem integrity, function and services.

Slowing the loss of biodiversity is one of the defining environmental challenges of the 21st century (*1*–*5*). Habitat loss, climate change, unregulated harvest, and other forms of human-caused mortality (*6*, *7*) have contributed to a thousand-fold increase in global extinctions in the Anthropocene compared to the presumed prehuman background rate, with profound effects on ecosystem functioning and services (*8*). The overwhelming focus on species extinctions, however, has underestimated the extent and consequences of biotic change, by ignoring the loss of abundance within still-common species and in aggregate across large species assemblages (*2*, *9*). Declines in abundance can degrade ecosystem integrity, reducing vital ecological, evolutionary, economic, and social services that organisms provide to their environment (*8*, *10*–*15*). Given the current pace of global environmental change, quantifying change in species abundances is essential to assess ecosystem impacts. Evaluating the magnitude of declines requires effective long-term monitoring of population sizes and trends, data which are rarely available for most taxa.

Birds are excellent indicators of environmental health and ecosystem integrity (*16*, *17*), and our ability to monitor many species over vast spatial scales far exceeds that of any other animal group. We evaluated population change for 529 species of birds in the continental United States and Canada (76% of breeding species), drawing from multiple standardized bird-monitoring datasets, some of which provide close to fifty years of population data. We integrated range-wide estimates of population size and 48-year population trajectories, along with their associated uncertainty, to quantify net change in numbers of birds across the avifauna over recent decades (*18*). We also used a network 143 weather radars (NEXRAD) across the contiguous U.S. to estimate long-term changes in nocturnal migratory passage of avian biomass through the airspace in spring from 2007 to 2017. The continuous operation and broad coverage of NEXRAD provide an automated and standardised monitoring tool with unrivaled temporal and spatial extent (*19*). Radar measures cumulative passage across all nocturnally migrating species, many of which breed in areas north of the contiguous U.S. that are poorly monitored by avian surveys. Radar thus expands the area and the proportion of the migratory avifauna that is sampled relative to ground surveys.

Results from long-term surveys, accounting for both increasing and declining species, reveal a net loss in total abundance of 2.9 billion (95% CI = 2.7-3.1 billion) birds across almost all biomes, a reduction of 29% (95% CI = 27-30%) since 1970 (Fig. 1 and Table 1). Analysis of NEXRAD data indicate a similarly steep decline in nocturnal passage of migratory biomass, a reduction of 13.6 ± 9.1% since 2007 (Fig. 2A). Reduction in biomass passage occurred across the eastern U.S. (Fig. 2, C and D), where migration is dominated by large numbers of temperate- and boreal-breeding songbirds; we observed no consistent trend in the Central or Pacific flyway regions (Fig. 2, B to D, and table S5). Two completely different and independent monitoring techniques thus signal major population loss across the continental avifauna.

Species exhibiting declines (57%, 303/529) based on long-term survey data span diverse ecological and taxonomic groups. Across breeding biomes, grassland birds showed the largest magnitude of total population loss since 1970—more than 700 million breeding individuals across 31 species— and the largest proportional loss (53%); 74% of grassland species are declining. (Fig. 1 and Table 1). All forest biomes experienced large avian loss, with a cumulative reduction of more than 1 billion birds. Wetland birds represent the only biome to show an overall net gain in numbers (13%), led by a 56% increase in waterfowl populations (Fig. 3 and Table 1). Surprisingly, we also found a large net loss (63%) across 10 introduced species (Fig. 3, D and E, and Table 1).

A total of 419 native migratory species experienced a net loss of 2.5 billion individuals, whereas 100 native resident species showed a small net increase (26 million). Species overwintering in temperate regions experienced the largest net reduction in abundance (1.4 billion), but proportional loss was greatest among species overwintering in coastal regions (42%), southwestern aridlands (42%), and South America (40%) (Table 1 and fig. S1). Shorebirds, most of which migrate long distances to winter along coasts throughout the hemisphere, are experiencing consistent, steep population loss (37%).

More than 90% of the total cumulative loss can be attributed to 12 bird families (Fig. 3A), including sparrows, warblers, blackbirds, and finches. Of 67 bird families surveyed, 38 showed a net loss in total abundance, whereas 29 showed gains (Fig. 3B), indicating recent changes in avifaunal composition (table S2). While not optimized for species-level analysis, our model indicates 19 widespread and abundant landbirds (including 2 introduced species) each experienced population reductions of >50 million birds (data S1). Abundant species also contribute strongly to the migratory passage detected by radar (*19*), and radar-derived trends provide a fully independent estimate of widespread declines of migratory birds.

Our study documents a long-developing but overlooked biodiversity crisis in North America—the cumulative loss of nearly 3 billion birds across the avifauna. Population loss is not restricted to rare and threatened species, but includes many widespread and common species that may be disproportionately influential components of food webs and ecosystem function. Furthermore, losses among habitat generalists and even introduced species indicate that declining species are not replaced by species that fare well in human-altered landscapes. Increases among waterfowl and a few other groups (e.g., raptors recovering after the banning of DDT) are insufficient to offset large losses among abundant species (Fig. 3). Importantly, our population loss estimates are conservative since we estimated loss only in breeding populations. The total loss and impact on communities and ecosystems could be even higher outside the breeding season if we consider the amplifying effect of "missing" reproductive output from these lost breeders.

Extinction of the Passenger Pigeon (*Ectopistes migratorius*), once likely the most numerous bird on the planet, provides a poignant reminder that even abundant species can go extinct rapidly. Systematic monitoring and attention paid to population declines could have alerted society to its pending extinction (*20*). Today, monitoring data suggest that avian declines will likely continue without targeted conservation action, triggering additional endangered species listings at tremendous financial and social cost. Moreover, because birds provide numerous benefits to ecosystems (e.g., seed dispersal, pollination, pest control) and economies (47 million people spend 9.3 billion U.S. dollars per year through bird-related activities in the U.S. (*21*)), their population reductions and possible extinctions will have severe direct and indirect consequences (*10, 22*). Population declines can be reversed, as evidenced by the remarkable recovery of waterfowl populations under adaptive harvest management (*23*) and the associated allocation of billions of dollars devoted to wetland protection and restoration, providing a model for proactive conservation in other widespread native habitats such as grasslands.

Steep declines in North American birds parallel patterns of avian declines emerging globally (*14, 15, 22, 24*). In particular, depletion of native grassland bird populations in North America, driven by habitat loss and more toxic pesticide use in both breeding and wintering areas (*25*), mirrors loss of farmland birds throughout Europe and elsewhere (*15*). Even declines among introduced species match similar declines within these same species' native ranges (*26*). Agricultural intensification and urbanization have been similarly linked to declines in insect diversity and biomass (*27*), with cascading impacts on birds and other consumers (*24, 28, 29*). Given that birds are one of the best monitored animal groups, birds may also represent the tip of the iceberg, indicating similar or greater losses in other taxonomic groups (*28, 30*).

Pervasiveness of avian loss across biomes and bird families suggests multiple and interacting threats. Isolating spatio-temporal limiting factors for individual species and populations will require additional study, however, since migratory species with complex life histories are in contact with many threats throughout their annual cycles. A focus on breeding season biology hampers our ability to understand how seasonal interactions drive population change (*31*), although recent continent-wide analyses affirm the importance of events during the non-breeding season (*19, 32*). Targeted research to identify limiting factors must be coupled with effective policies and societal change that emphasize reducing threats to breeding and non-breeding habitats and minimizing avoidable anthropogenic mortality year-round. Endangered species legislation and international treaties, such as

Table 1. Net change in abundance across the North American avifauna, 1970-2017. Species are grouped into native and introduced species, management groups (landbirds, shorebirds, waterbirds, waterfowl), major breeding biomes, and nonbreeding biomes (see data S1 in (*18*) for assignments and definitions of groups and biomes). Net change in abundance is expressed in millions of breeding individuals, with upper and lower 95% credible intervals (CI) shown. Percentage of species in each group with negative trend trajectories are also noted. Values in bold indicate declines and loss; those in italics indicate gains.

Species Group	Number of Species	Net Abundance Change (Millions) & 95% CI			Percent Change & 95% CIs			Proportion Species in Decline
		Change	LC95	UC95	Change	LC95	UC95	
Species Summary								
All N. Am. Species	529	**-2,911.9**	-3,097.5	-2,732.9	**-28.8%**	-30.2%	-27.3%	57.3%
All Native Species	519	**-2,521.0**	-2,698.5	-2,347.6	**-26.5%**	-28.0%	-24.9%	57.4%
Introduced Species	10	**-391.6**	-442.3	-336.6	**-62.9%**	-66.5%	-56.4%	50.0%
Native Migratory Species	419	**-2,547.7**	-2,723.7	-2,374.5	**-28.3%**	-29.8%	-26.7%	58.2%
Native Resident Species	100	*26.3*	7.3	46.9	*5.3%*	1.4%	9.6%	54.0%
Landbirds	357	**-2,516.5**	-2,692.2	-2,346.0	**-27.1%**	-28.6%	-25.5%	58.8%
Shorebirds	44	**-17.1**	-21.8	-12.6	**-37.4%**	-45.0%	-28.8%	68.2%
Waterbirds	77	**-22.5**	-37.8	-6.3	**-21.5%**	-33.1%	-6.2%	51.9%
Waterfowl	41	*34.8*	24.5	48.3	*56.0%*	37.9%	79.4%	43.9%
Aerial Insectivores	26	**-156.8**	-183.8	-127.0	**-31.8%**	-36.4%	-26.1%	73.1%
Breeding Biome								
Grassland	31	**-717.5**	-763.9	-673.3	**-53.3%**	-55.1%	-51.5%	74.2%
Boreal forest	34	**-500.7**	-627.1	-381.0	**-33.1%**	-38.9%	-26.9%	50.0%
Forest Generalist	40	**-482.2**	-552.5	-413.4	**-18.1%**	-20.4%	-15.8%	40.0%
Habitat Generalist	38	**-417.3**	-462.1	-371.3	**-23.1%**	-25.4%	-20.7%	60.5%
Eastern Forest	63	**-166.7**	-185.8	-147.7	**-17.4%**	-19.2%	-15.6%	63.5%
Western forest	67	**-139.7**	-163.8	-116.1	**-29.5%**	-32.8%	-26.0%	64.2%
Arctic Tundra	51	**-79.9**	-131.2	-0.7	**-23.4%**	-37.5%	-0.2%	56.5%
Aridlands	62	**-35.6**	-49.7	-17.0	**-17.0%**	-23.0%	-8.1%	56.5%
Coasts	38	-6.1	-18.9	8.5	-15.0%	-39.4%	21.9%	50.0%
Wetlands	95	*20.6*	8.3	35.3	*13.0%*	5.1%	23.0%	47.4%
Nonbreeding Biome								
Temperate North America	192	**-1,413.0**	-1,521.5	-1,292.3	**-27.4%**	-29.3%	-25.3%	55.2%
South America	41	**-537.4**	-651.1	-432.6	**-40.1%**	-45.2%	-34.6%	75.6%
Southwestern Aridlands	50	**-238.1**	-261.2	-215.6	**-41.9%**	-44.5%	-39.2%	74.0%
Mexico-Central America	76	**-155.3**	-187.8	-122.0	**-15.5%**	-18.3%	-12.6%	52.6%
Widespread Neotropical	22	**-126.0**	-171.2	-86.1	**-26.8%**	-33.4%	-19.3%	45.5%
Widespread	60	-31.6	-63.1	1.6	-3.7%	-7.4%	0.2%	43.3%
Marine	26	**-16.3**	-29.7	-1.2	**-30.8%**	-49.1%	-2.5%	61.5%
Coastal	44	**-11.0**	-14.9	-6.7	**-42.0%**	-51.8%	-26.7%	68.2%
Caribbean	8	-6.0	1.4	-15.7	12.1%	-2.8%	31.7%	25.0%

The Influence of Mammals and Birds in Retarding Artificial and Natural Reseeding of Coniferous Forests in the United States

1947 J. For. 45: 361– 369

Clarence F. Smith and Shaler E. Aldous[1]

FORESTERS have long been searching for an economical method of re-establishing trees on deforested lands. Three methods have been used, namely: the planting of nursery-raised stock and natural seedlings, direct seeding, and the encouragement of natural reproduction. To date the planting program has been given the most attention but the other two methods are older and considerable hope for their satisfactory use is always being expressed. Chief among the factors that have pushed reproduction by seeding into the background is the depredation of seed-eating rodents and birds.

This does not mean, however, that animals are the chief factors in preventing reforestation. Among the many other important factors are: availability of seed source, soil type and condition, prevalence of destructive insects and diseases, moisture supply, temperature, slope, and many others. So it should not be inferred that if rodent and bird pressures could be removed reforestation would always be successful. The role of mammals and birds in aiding reforestation is not discussed in this paper. Although actual consumption of coniferous seeds implies destruction, it must be recalled that in the process of feeding, mammals through caching, and birds by transportation and dropping of seeds may function as seed distributors.

The first known effort at direct seeding on a national forest in the United States was in 1901 (*11*) on the San Bernardino National Forest, California. This and several other trials (*12*, *38*) failed due to a combination of factors, one of which was the activity of seed-eating animals. Periodic attempts at direct seeding were continued with little or no success through about 1912 (*6*, *20*, *27*, *39*, *59*, *62*). In the early twenties an interest was again shown in this work (*30*, *41*, *42*, *50*), but still one of the main obstacles to success was the animal factor. During the late thirties and up until the present time, a renewed interest has again been extended to this method of reforestation (*1*, *2*, *8*, *10*, *16*, *25*, *26*, *32*, *33*, *34*, *35*, *36*, *37*, *40*, *45*, *46*, *47*, *48*, *49*, *58*, *61*, *63*).

It has been during this latter period that the authors and other workers in the Fish and Wildlife Service have been making extensive investigations of the relation of animals to the forest. Reproduction by seed and the animal pressure thereon has received considerable attention. It is the purpose of this paper to point out the degree and type of pressure exerted by birds and mammals on forest seeds, the tree species affected, and to describe methods of reducing or eliminating these losses.

MANY ANIMAL AND BIRD SEED EATERS

A large number of different mammals and birds feed on the seeds of coniferous trees, both when supplied naturally and when provided by planting procedures. From the literature and the authors' studies, a total of 44 mammals and 37 birds have been found to eat coniferous seeds (Table 1). This list could be materially lengthened if the data from unpublished stomach analyses in the files of the Fish and Wildlife Service were assembled. Van Dersal (*64*) lists the animals that feed on all species of conifers in the United States, but from his descriptions it is impossible to separate seed eaters from vegetative feeders.

Although the list of seed eaters is long, only a few species are of significance in forest regeneration. The tree squirrels, chipmunks, white-footed mice, and a few species of seed-eating fringillid birds are responsible for the greater part of all trouble caused by mammals and birds in regenerating forests by seeding. Habitat is the governing factor which determines which species of animal might be present to consume the seeds.

Tree squirrels (*Sciurus*) are not commonly found very far from trees with the result that their depredations are usually on the cones or on the seed dropping from them. On denuded areas that are being artificially reseeded, these squirrels are of minor importance except around the tree-fringed borders. On the other hand, they

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[1] Clarence F. Smith, formerly biologist of U. S. Fish and Wildlife Service Control Methods Research. Stationed at Mt. Shasta, Calif. Shaler E. Aldous, biologist with U. S. Fish and Wildlife Service, Public Lands Wildlife Research Section, St. Paul, Minn.

TABLE 1.—LIST OF BIRDS AND MAMMALS KNOWN TO EAT THE SEEDS OF CONIFEROUS TREES (AND THE AUTHORITY REFERENCE)[1]

Common name	Scientific name	Reference
Birds		
✓Blue grouse	*Dendragapus obscurus*	4
Sierra grouse	*Dendragapus fuliginosus sierrae*	52
Ruffed grouse	*Bonasa umbellus*	52
Bobwhite	*Colinus virginianus*	8
California quail	*Lophortyx californica*	52
Band-tailed pigeon	*Columbia fasciata*	52
Eastern mourning dove	*Zenaidura macroura carolinensis*	8, 58, 60
✓Red-shafted flicker	*Colaptes cafer collaris*	57
✓Lewis woodpecker	*Asyndesmus lewis*	52
Canada jay	*Perisoreus canadensis*	59
✓Blue jay	*Cyanocitta cristata*	59
✓Steller's jay	*Cyanocitta stelleri*	52, 57
✓Crow	*Corvus brachyrhynchos*	52
✓Clark's nutcracker	*Nucifraga columbiana*	52
✓Winter wren	*Nannus hiemalis*	36
✓Robin	*Turdus migratorius*	15, 59
English sparrow	*Passer domesticus*	59
Southern meadowlark	*Sturnella magna argutula*	8
Red-wing blackbird	*Agelaius phoeniceus*	8
Rusty blackbird	*Euphagus carolinus*	8
Brewer's blackbird	*Euphagus cyanocephalus*	8
Blackbirds	No species named	15, 59
Cowbird	*Molothrus ater*	8
Bronzed grackle	*Quiscalus quiscalus aeneus*	15
Blue grosbeak	*Guiraca caerulea*	59
✓Evening grosbeak	*Hesperiphona vespertina*	52
California purple finch	*Carpodacus purpureus californicus*	36
✓Pine grosbeak	*Pinicola enucleator*	52
✓Redpoll	*Acanthis linaria*	59
✓Goldfinch	*Spinus tristis*	Text
✓Red crossbill	*Loxia curvirostra*	52
Towhee	*Pipilo maculatus*	36
✓Slate-colored junco	*Junco hyemalis hyemalis*	Text
Carolina junco	*Junco hyemalis connectens*	13
Oregon junco	*Junco oreganus*	36
Red-backed junco	*Junco phoeonotus dorsalis*	39, 52
✓Chipping sparrow	*Spizella passerina*	2
Song sparrow	*Melospiza melodia*	36
Mammals		
Cinereous shrew	*Sorex cinereus*	Text
Trowbridge shrew	*Sorex Trowbridgii*	37
Wandering shrew	*Sorex vagrans*	37
Baird's dusky shrew	*Sorex obscurus bairdi*	37
Yaquina shrew	*Sorex yaquinae*	37
Gibbs shrew-mole	*Neurotrichus gibbsii*	37
Columbia ground squirrel	*Citellus beecheyi*	23
Rock squirrel	*Citellus variegatus*	24
California ground squirrel	*Citellus beecheyi*	23
Fisher ground squirrel	*Citellus beecheyi fisheri*	52
White-tailed antelope squirrel	*Citellus leucurus*	24
Mantled ground squirrel	*Citellus lateralis*	24, 59
	Citellus lateralis arizonensis	57
	Citellus lateralis chrysodeirus	52
✓Western chipmunk	*Eutamias minimus*	52
	Eutamias minimus jacksoni	Text
	Eutamias amoenus	52
	Eutamias quadrivittatus	52
	Eutamias quadrivittatus speciosus	52
	Eutamias cinereicollis	52, 57
	Eutamias townsendii	26, 52
	Eutamias quadrimaculatus	52
	Eutamias sp.	12, 30, 49, 58, 61
✓Eastern chipmunk	*Tamias striatus striatus*	53
	Tamias striatus griseus	Text

are probably the heaviest consumers of coniferous seeds under natural conditions. Heavy cutting of sugar pine cones (even before they are mature) by the California gray squirrel was reported by Berry (*6*) and Jotter (*27*). As much as 70 to 75 percent of the seed crop was reported taken by these squirrels and as many as 702 ponderosa pine cones were found under one tree.

Voracious Rodents Are Heavy Seed Consumers

The appetite for pine seeds by squirrels was shown by Hatt (*18*) when he reported one pair of captive red squirrels ate the seeds from 422 cones of eastern white pine in one week's time. Mollenhauer (*35*) found red squirrels clipped from 50 to 100 percent of the cone-bearing limbs of the table mountain pine. It was the opinion of Pulling (*41*) that red squirrels took all the white pine seed produced during a poor year in one area in New York State. An examination of three red squirrel stomachs from a jack pine slash area in Minnesota showed that they had consumed 1,178 jack pine seeds or an average of 392 seeds per squirrel. In this latter case, the squirrels were collected at the edge of the cutting area. None was observed in the main body of the clearing. This localization was further illustrated by Smith (*53*) when he wrote, "The greatest rodent damage occurred on the fully-timbered area where red squirrels invaded the quadrats several times searching for ungerminated seeds."

Ground squirrels (*Citellus*) are not commonly pests on sites requiring reforestation. These animals occur only in the western states in open types of habitat where reforestation is not likely to be practiced. However, these animals are known to retard forest regeneration at times, as Tinsley (*58*) reported the Columbian ground squirrel in Idaho to have consumed practically all the seeds (pondersosa and western white pine) sown.

One associated quite closely with the forest is the mantled squirrel. Throughout the ponderosa pine belt, this group is common and believed to be an important ponderosa pine seed consumer by Wahlenberg (*60*) and Taylor and Gorsuch (*57*).

Chipmunks play an important role in preventing the reseeding of forests. The western chipmunk (*Eutamias*) has been a greater factor in this respect than the eastern form (*Tamias*). Most seeding studies have been made in the West and in the Lake States where the western form is much more abundant. In the Lake States, both chipmunks occupy the same range, but *Eutamias* is found more in the logged or burned areas and is much more numerous over the whole area.

In northern Minnesota, chipmunks were found to be pronounced pine seed destroyers. The examination of stomachs from 88 chipmunks col-

TABLE 1—(*Continued*)

Common name	Scientific name	Reference
Red squirrel	*Tamiasciurus hudsonicus*	*5, 9, 15, 26, 29, 41, 43, 49, 53, 56*
	Tamiasciurus hudsonicus baileyi	*41, 43, 49, 53, 56, 18*
	Tamiasciurus hudsonicus loquax	*65*
	Tamiasciurus hudsonicus ventorum	*21*
	Tamiasciurus hudsonicus richardsoni	*36, 54*
	Tamiasciurus douglasii	*36, 52*
	Tamiasciurus douglasii albolimbatus	*17*
	Tamiasciurus fremonti fremonti	*19*
Western gray squirrel	*Sciurus griseus*	*36, 52*
Flying squirrel	*Glaucomys sabrinus latipes*	*3*
Pocket mouse	*Perognathus fasciatus*	Text
	Perognathus parvus mollipilosus	*51*
Kangaroo rat	*Dipodomys heermanni*	*52*
White-footed mouse	*Peromyscus maniculatus*	*52*
	Peromyscus maniculatus gracilis	Text
	Peromyscus leucopus noveboracensis	*53*
	Peromyscus sp.	*55, 62*
Woodrat	*Neotoma fuscipes*	*14*
Red-backed mouse	*Clethrionomys gapperi*	Text
Meadow mouse	*Microtus montanus*	*58*
House mouse	*Mus musculus*	*53*

[1] This list does not include species eating Juniper and Taxis, and includes references only from published articles and findings from the author's field work.

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lected in a jack pine slash area revealed that 72 percent contained the remains of jack pine seeds. They averaged 31 seeds per stomach, but one chipmunk had eaten 796 and another had 204 in its stomach and 168 in its cheek pouches. The 88 animals had disposed of a total of 2,749 jack pine seeds. Although chipmunks formed only 39.5 percent of the rodent population, they consumed 61 percent of the jack pine seeds removed by rodents on this area.

Coniferous seed destruction by chipmunks has been reported by Dearborn (*12*) from the Black Hills, Tinsley (*58*) in the Northern Rocky Mountains, Korstian and Baker (*30*) in the intermountain region of Utah, Taylor and Gorsuch (*57*) in the Southwest, Willis (*62*) and Moore (*36*) in the Pacific Northwest, L. Smith (*53*) for the New England area, and Horn, in an unpublished (1932) report from California.

Mice as a group, because of their wide habitat and geographic distribution, are perhaps the greatest animal factor in retarding forest regeneration by seeds. A great many of the authors reviewed, mentioned mice as one of the chief offenders.

White-footed mice (*Peromyscus*) have been found by the authors to be the most widely distributed and the most influential species in their studies. Where other authors have specified species, the white-footed group are usually mentioned. According to Silver (*50*), white-footed mice damaged from 10 to 90 percent of the seed spots on a planting program in the Ocala National Forest, Florida, Willis (*62*) attributed 98 percent failure of seed spots of Douglas-fir on the Columbia National Forest to the work of mice, and cage tests showed that a mouse could consume 300 seeds of this species daily. In the Lake States white-footed mice were found to be second only to chipmunks in numbers and in the consumption of jack pine seeds. The examination of 30 stomachs revealed that 70 percent had eaten jack pine seeds and the whole lot averaged 19 seeds per stomach. The greatest number eaten by a single animal was 156.

Other mice known to feed on coniferous seeds are red-backed mice (*Clethrionmys*), meadow mice (*Microtus*), jumping mice (*Zapus*), pocket mice (*Perognathus*), and house mice (*Mus musculus*).

The red-backed mice are perhaps the most characteristic forest species, but their food preferences run more to succulent vegetation than to seeds. After a logging operation where most of the timber is cut, these mice are rapidly replaced by the white-footed species. That they can be a minor factor in seed consumption, however, is shown by the analysis of 59 stomachs of mice trapped on a jack pine slash area. Only 14 percent of them had eaten jack pine seed, the greatest number per individual being 25, and the average .6.

BIRDS ALSO TAKE MUCH SEED

Birds are more of an influence in seed consumption than is generally believed. Many workers have referred to seed destruction by birds and rodents but few have specified which birds are involved. As early as 1916, Toumey (*59*) listed mourning doves, redpoll, Canada jay, blue jay, English sparrow, robin, blackbird, finch, junco, and blue grosbeak as forest seed eaters. In the longleaf pine area of Mississippi, Burleigh (*8*) found heavy consumption of seeds by mourning doves, bobwhites, meadow larks, Brewer's rusty, and red-winged blackbirds, and cowbirds. He found that damage by these birds was most severe during years of heavy seed production and relatively light during scanty seed years, the explanation being that the seed crop when abundant attracted and held larger bird populations. Pearson (*39*) and Krauch in a 1937 report for the southwest named the junco as a cause of failure of seeding experiments.

In the northwest, Moore (*36*) found Douglas-fir seeds attractive to winter wrens, purple finches, song sparrows, towhees, and juncos.

Four juncos, eight chipping sparrows, eight goldfinches, and one Brewer's blackbird were collected on a jack pine slash area in northern Minnesota. Of these 21 birds only three (chipping sparrows) had failed to feed on jack pine seeds. Most of these birds were present in fair-sized flocks, which could have been responsible for the consumption of a large part of the seed crop.

It is rather surprising to find such insectivorous mammals as shrews consuming coniferous seeds, but Moore (*37*) states, "They play a definite and deleterious part in natural reforestation." He lists five species of shrews that were found to eat Douglas-fir seeds. Jack pine seeds were found in the stomach of one long-tailed shrew in Minnesota.

Many workers have deplored serious losses of seed by rodents and birds without specifying the identity of the animals concerned. In fact, the writers have found it impossible to attribute

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Pine Seeds

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Competition Among Insects, Birds and Mammals for Conifer Seeds

CHRISTOPHER C. SMITH AND RUSSELL P. BALDA

Department of Biology,
Museum of Northern Arizona,
Route 4, Box 720,
Flagstaff, Arizona 86001

Department of Biological Sciences,
Northern Arizona University,
Flagstaff, Arizona 86011

1979

SYNOPSIS. Species of at least 5 orders of insects, 6 families of birds, and 2 orders of mammals, in various combinations, can exploit the cones and seeds of most species of conifers. Lodgepole pine is the exception to this pattern of broad taxonomic diversity of seed predators in that only pine squirrels and coreid bugs attack its serotinous cones. The contrast between lodgepole pine and other conifers demonstrates that large intrinsic variation in the abundance of a resource fosters the evolution of a broad range of taxonomic groups to exploit the resource. The diverse groups are limited by different predators and alternate resources when conifer seeds independently decrease in abundance.

INTRODUCTION

Because conifers are such an important source of lumber, a great amount of effort has been put into the study of their biology (Fowells, 1965), their seeds (USDA, 1974), and the predators of their seeds (Keen, 1958). People concerned with the lumber industry are interested in how many seeds escape predation to allow reforestation of logged or burned areas. Questions of general biological interest that do not relate directly to reforestation may go unasked even though pertinent information is available. In this paper, we use the extensive literature on conifer seeds and their predators to outline the nature of these seeds as a food resource and the manner in which the various seed predators use this resource. The information can then be used as a basis for answering three questions: 1) Why are there many different distantly related taxa of animals exploiting a distinctive and easily delimited resource? 2) How do the various taxa interact in using the resource? 3) How do the different taxa avoid competitive exclusion? Our personal experience is mainly limited to the conifers of western North America and our discussion will be concentrated on those, although some of the patterns probably apply generally to conifers in the Northern Hemisphere (Bock and Lepthien, 1976).

THE RESOURCE

As a food resource, single viable conifer seeds are highly uniform as to size (weight of female gametophyte and embryo) and quality (caloric value of female gametophyte and embryo) within an individual cone, tree or species population (Table 1). The total number of viable seeds in single cones within a single annual crop for one tree or a species population is more variable than seed size or quality, especially in lodgepole pine (*Pinus contorta* Dougl.) (Table 1). The sizes of annual cone crops for single trees or species populations are by far the most variable feature of conifer seeds in all western North American species, except lodgepole pine (Table 1).

The variation in caloric value of seeds given in Table 1 is probably an underestimation because 5 to 10 seed kernels were burned to derive each value thus masking the variation between individual seeds. The similar variations between samples from the same cone, cones from the same tree,

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We wish to thank Jim Reichman for stimulating us to think about competition between distantly related taxa. Gary Bateman and Gilbert Schubert made helpful comments in reviewing the manuscript. Anne Slobodchikoff was very helpful in editing and typing the manuscript.

TABLE 1. *The source of variation in conifer seeds as a food resource.*

Species	Source of units	Number	Range	Mean	Coefficient of variation
	Caloric value of female gametophyte and embryo (Cal/g dry weight):				
Lodgepole pine	groups of 6 to 10 seeds from one cone	5	6.55–6.91	6.74	2.0%
Lodgepole pine	groups of 6 to 10 seeds from one tree	6	6.63–7.04	6.79	2.3%
Lodgepole pine	groups of 5 to 10 seeds from one population	7	6.54–6.96	6.83	2.2%
	Weight of female gametophyte and embryo (mg/seed):				
Engelmann spruce	seeds from one cone	40	1.6–2.0	1.91	7.5%
Lodgepole pine	seeds from one cone	25	1.7–2.3	1.88	9.5%
Lodgepole pine	10 means of seeds from 10 cones from one tree	10	2.01–3.27	2.49	14.8%
Lodgepole pine	6 means of seeds from 6 cones from different trees	6	1.88–3.63	2.53	24.0%
	Filled seed content of cones (filled seeds/cone):				
Lodgepole pine	cones from one tree	5	9–23	15.8	40.4%
Lodgepole pine	10 means of cones from 10 trees	10	4.4–18.8	11.6	39.5%
Ponderosa pine	cones from one tree	5	53–89	70.4	22.9%
Ponderosa pine	7 means of cones from 7 trees	7	39.0–70.4	53.5	20.6%
	Annual cone crops (cones/tree or cones/acre):				
Douglas-fir	11 annual means for several trees (cones/tree)	11	0–14,244	4219	124.6%
Ponderosa pine	10 annual crops (cones/acre)	10	316–7521	2437	93.1%
	Ratios of successive cone crops (%):				
Douglas-fir	smaller crop/larger crop × 100 for 10 pairs of successive years	10	0.0–67.2	6.9	
Ponderosa pine	smaller crop/larger crop × 100 for 9 pairs of successive years	9	4.2–94.4	40.2	
Lodgepole pine	smaller crop/larger crop × 100 for 4 pairs of successive years	4	50.8–73.5	62.4	

TABLE 3. *continued*

Taxonomic group of seed predators	Genera of trees exploited	Method of conifer exploitation	Method of escaping starvation during cone crop failure
BIRDS:			
Family: Psittacidae			
parrots genus: *Rhynchopsitta*	*Pinus*	extracted seeds from closed and open cones	emigration, nomadic
Family: Picidae			
Hairy Woodpecker White-headed Woodpecker genus: *Picoides*	*Pinus*	extracted seeds from closed and open cones eaten by adults and juveniles	alternate foods
Family: Corvidae			
Piñon Jay genus: *Gymnorhinus*	*Pinus*	cache seeds extracted from closed and open cones to feed young and adults	alternate foods, emigration
Clark's Nutcracker genus: *Nucifraga*	*Pinus*	cache seeds extracted from open and closed cones to feed young and adults	alternate foods, emigration
Gray Jay genus: *Perisoreus*	*Abies*, *Picea*	may extract seeds from open cones or retrieve from ground to eat and store as boli	alternate foods, territorial
Steller's Jay genus: *Cyanocitta*	*Pinus*	cache seeds from open cones to feed adults	alternate foods, territorial, emigration
Scrub Jay Mexican Jay genus: *Aphelocoma*	*Pinus*	cache seeds from open cones and ground to feed adults	alternate foods, territorial
Black-billed Magpie genus: *Pica*	*Pinus*	cache seeds from open cones and ground to feed adults	alternate foods, territorial
Family: Paridae			
Mountain Chickadee	*Abies*, *Picea*, *Pseudotsuga*, *Pinus*	cache seeds from open cones and ground to feed adults	alternate foods
Boreal Chickadee	*Abies*, *Picea*, *Pseudotsuga*	cache seeds from open cones and ground to feed adults	alternate foods
Plain Titmouse genus: *Parus*	*Pinus*	cache seeds from open cones and ground to feed adults	alternate foods, territorial

genus: *Parus*		ground to feed adults	
Family: Sittidae White-breasted Nuthatch	*Pinus*	uses and makes cache of seeds and finds seeds on ground	alternate foods
Red-breasted Nuthatch	*Abies*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	seeds from open cones cached and eaten by adults	alternate foods, emigration
Pygmy Nuthatch genus: *Sitta*	*Abies*, *Picea*, *Pseudotsuga*, *Pinus*	seeds from open cones cached and eaten by adults	alternate foods
Family: Fringillidae crossbills genus: *Loxia*	*Abies*, *Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	seeds from closed and open cones fed to young and adults	nomadic nesters, emigrate to alternate foods
Pine Siskins genus: *Carduelis*	*Abies*, *Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	seeds from open cones and the ground fed to young and adults	emigration, alternate foods
MAMMALS:			
Order: Rodentia pine squirrels genus: *Tamiasciurus*	*Abies*, *Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	cache and feed from whole closed cones	territoriality, emigration, alternate foods
tree squirrels genus: *Sciurus*	*Pinus*	seeds from closed cones	alternate foods
chipmunks genus: *Eutamias*	*Abies*, *Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	cache and eat seeds from closed or open cones	alternate foods
golden-mantled ground squirrel *Spermophilus lateralis*	*Pinus*	forage for single seeds	alternate foods
deer mice genus: *Peromyscus*	*Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	forage for single seeds and cache seeds	alternate foods
red-backed voles genus: *Clethrionomys*	*Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	forage for single seeds and cache seeds	alternate foods
Order: Insectivora shrews genus: *Sorex*	*Larix*, *Picea*, *Pinus*, *Pseudotsuga*, *Tsuga*	forage for single seeds	alternate foods

2020

Conservation Strategy for the Pinyon Jay (*Gymnorhinus cyanocephalus*)

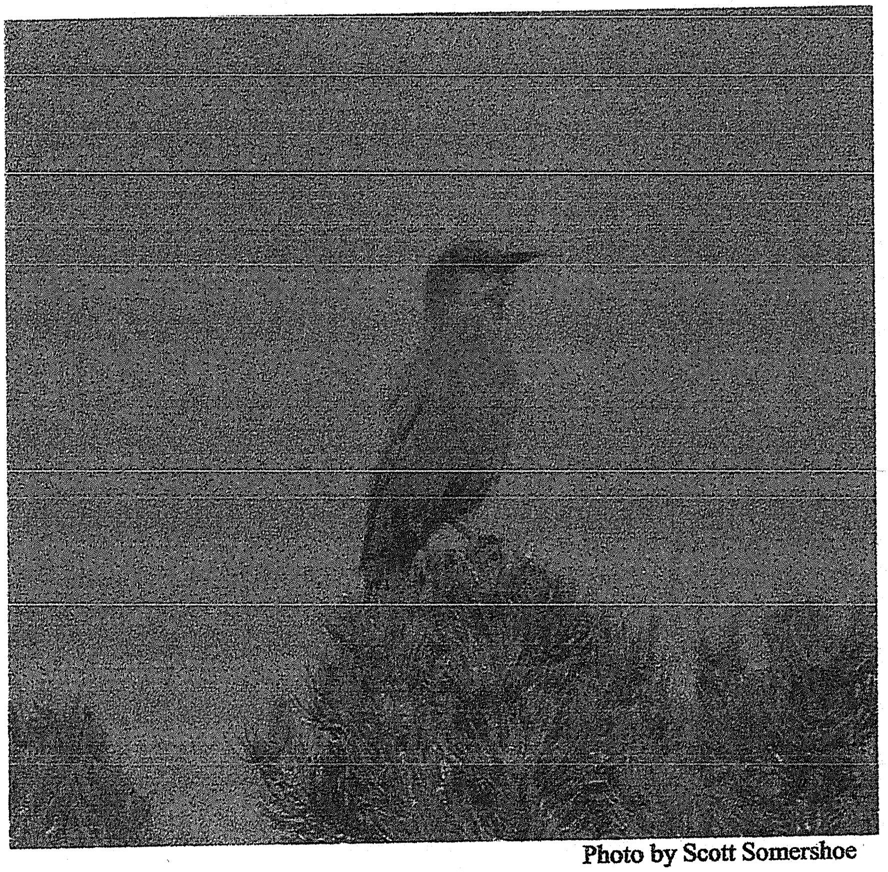

Photo by Scott Somershoe

Version 1, February 2020

Western Working Group • Grupo de Trabajo de Occidente

U.S. FISH & WILDLIFE SERVICE

Executive Summary

The Pinyon Jay (*Gymnorhinus cyanocephalus*) is an obligate bird of piñon-juniper and other pine-juniper woodlands that has experienced significant population declines and is of increasing conservation concern. The purpose of this strategy is to summarize current knowledge about Pinyon Jays and identify research, monitoring, and conservation actions required to improve their population status.

Pinyon Jays are highly social and maintain year-round flocks that occupy large home ranges and use a variety of woodland habitats for foraging, caching piñon nuts, nesting, and roosting. The Partners in Flight North American Landbird Conservation Plan estimates a current global population of 770,000 Pinyon Jays. Over the period from 1967–2015, populations declined by 3.69% annually for an estimated total loss of 83.5%. The causes of these declines are unclear, largely because Pinyon Jays remain understudied. Most earlier research has occurred in New Mexico and Arizona and has focused on behavior or on habitat associations at nest colonies. More recent and ongoing research efforts are focused on understanding habitat requirements, resource needs, and movement patterns throughout the entire annual cycle in different regions of the species' range. Future research needs include: identifying home ranges and nesting colony sites, habitat use and requirements, nesting biology and survival, assessing causes of local and regional declines, and assessing effects of management.

As Pinyon Jay populations have declined, the piñon-juniper woodlands that provide most of their habitat also face potential threats, including removal of trees to accomplish other management priorities, long-term fire suppression, changes in woodland age and tree density, and changing climatic conditions that cause reduced piñon nut production and increased piñon pine mortality.

Effective management and conservation of the Pinyon Jay depends on a better understanding of the species' habitat requirements, identification of the factors that limit population size, and a clearer understanding of woodland dynamics and health. The information provided in this strategy to inform management is based on the best available science and is intended to help minimize unintended negative impacts to Pinyon Jays associated with current vegetation management activities. Collaboration between land managers and Pinyon Jay researchers provides a compelling approach to increase our knowledge of the species, better understand management trade-offs, and identify positive actions that could improve habitat for the species and reverse negative long-term population trends.

Chart = 4/

Chapter 2. Pinyon Jay Distribution, Natural History and Habitat

Distribution Patterns and Natural History

The Pinyon Jay is most abundant in the southern Great Basin and the southwestern U.S. (Figure 4), where it is usually associated with the piñon pines (e.g.; *Pinus edulis*, *P. monophylla*) for which it is named (Figure 6). Pinyon Jays are found in lower densities outside the range of piñon pines, in South Dakota, Nebraska, California, Oregon, Wyoming, and Montana, where they occur in habitat types dominated by ponderosa pine (*P. ponderosa*), Jeffrey pine (*P. jeffreyi*), and limber pine (*P. flexilis*) (Burleigh 1972, Grenfell and Laudenslayer 1983, Balda 2002, Faulkner 2010, Marks et al. 2016, Drilling et al. 2018, Silcock and Jorgensen 2018). Pinyon Jays are associated with juniper (in the absence of piñon pine) in southern Idaho (Brody 1992) and may also use juniper-dominated areas in Montana (J. Marks, pers. comm.), New Mexico (Johnson et al. 2014) and other areas.

Pinyon Jays are highly social throughout the year. They occur in flocks of up to a few hundred individuals and nest colonially, sometimes cooperatively, often using the same general nesting colony site annually (Marzluff and Balda 1992). Home ranges are large, typically 8,645 acres (3,500 hectares) to 15,800 acres (6,400 hectares) (Marzluff and Balda 1992; Balda 2002; Johnson et al. 2016, 2017a). Unlike traditionally territorial species, their nests are spatially clumped in nesting colonies that can cover 141 acres (57.4 hectares) or more (Figure 7, Johnson et al. 2014, 2015, 2018a).

141

Across their geographic range, Pinyon Jays inhabit varying elevations and latitudes with diverse woodland structures, dominant tree species, and landscape characteristics, and thus home range attributes may vary substantially from region to region. Additionally, local home ranges may shift seasonally, and jays may occasionally make large-scale movements (up to hundreds of miles) out of their normal home range when food resources are limited in fall and winter (Balda 2002). Pinyon Jays may also adjust daily and annual habitat use within home ranges as habitat conditions and resources vary over space and time. As a result, well-established occupancy patterns can change with seasons and with habitat conditions in unanticipated ways.

Pinyon Jays have a mutualistic relationship with piñon pines where they co-occur (Ligon 1978, Lanner 1996). Piñon pines are a masting species, producing highly nutritional nuts in large crops that historically occurred within local stands or regions at irregular intervals, from one to three crops every 10 years for *P. edulis* (Forcella 1981) and one crop every one to three years for *P. monophylla* (Sutton 1984). These intermittently abundant nuts sustain Pinyon Jays throughout the winter, support successful nesting, and significantly influence population viability (Marzluff and Balda 1992). Pinyon Jay nesting success and productivity are higher following mast years than non-mast years (Ligon 1978, Marzluff and Balda 1992), and adult survivorship is highest after moderate cone crops (Marzluff and Balda 1992). In turn, Pinyon Jays serve as the primary long-distance seed disperser for piñon trees within the species' range. They transport piñon nuts to caching areas up to several miles from the source woodland and may cache in sites favorable for seed germination (Ligon 1978). Therefore, Pinyon Jays likely serve an important role in overall woodland health and regeneration.

Outside the range of piñon pines, Pinyon Jays feed on and cache nuts of other pines, such as Jeffrey, limber, and ponderosa pines, but no information is available on the importance this seed dispersal by Pinyon Jays on the ecology and distribution of these other tree species. The effects of what may be a lower-quality food source on Pinyon Jay ecology, diet, and movements are also unknown. In addition to consuming piñon and other pine nuts, Pinyon Jays also forage extensively for insects in the shrub and grass understory.

Although Pinyon Jay mutualism with piñon pines has been emphasized, the species also eats insects (Ligon 1978, Balda 2002). Jays may rely on insects when piñon nuts are unavailable. Pinyon Jays have been reported to nest successfully in response to a large emergence of periodical cicadas (Ligon 1978), and most foraging observed during the nesting season involved capture of insects in the shrub understory of woodlands (Balda 2002, J. Boone and E. Ammon, unpublished data).

Piñon-Juniper Woodland Types

Piñon and piñon-juniper woodlands vary considerably across the Pinyon Jay's range. Perhaps the best framework for classifying these woodlands is the U.S. National Vegetation Classification (NVC, http://usnvc.org/), a collaborative effort of NatureServe and the Ecological Society of America (ESA). This classification system has been formally adopted by the U.S. Forest Service, the ESA, NatureServe, and the U.S. Geological Survey Core Science System. The NVC is a hierarchical classification of all U.S. vegetation types which provides common reference points for the various land management and conservation entities in the U.S.

Within the NVC, in the Southern Rockies and Colorado Plateau region, Pinyon Jays occur in the Intermountain Pinyon-Juniper Woodland Macrogroup, including the Colorado Plateau Pinyon-Juniper Woodland Group, and associated alliances. Jays also occur in the Southern Rocky Mountain Two-needle Pinyon-Juniper Woodland Macrogroup and associated groups and alliances. In New Mexico, Pinyon Jays have been documented nesting in the Southern Rocky Mountain Pinyon-Juniper Open Woodland Macrogroup, including the Southern Rocky Mountain Juniper Open Woodland Group (juniper-dominated woodland and savanna) and Southern Rocky Mountain Pinyon-Juniper Woodland Group (woodlands dominated by *P. edulis* and *Juniperus monosperma*) (Johnson et al. 2014, 2015). Here they tend to winter at lower-elevation, juniper-dominated woodland and savanna types of the Southern Rocky Mountain Juniper Open Woodland Group (Johnson et al. 2014). In the Great Basin, Pinyon Jays primarily occupy the Intermountain Pinyon-Juniper Woodland Macrogroup, including the Great Basin Pinyon-Juniper Woodland Group and associated alliances.

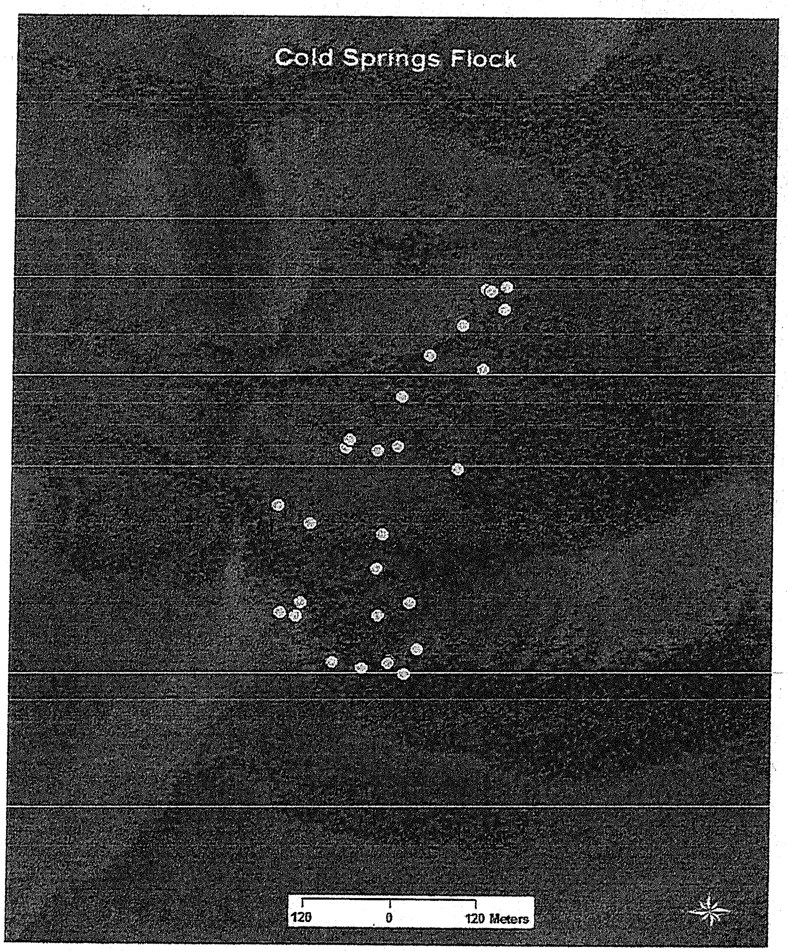

Figure 7. Spatial extent of a typical nesting colony in Nevada, with known nest sites as yellow points (J. Boone and E. Ammon, unpublished data).

Two other classification systems for piñon-juniper vegetation provide useful additional perspectives (Table 2, Figures 8–20). First, Romme et al. (2009) define three main piñon-juniper vegetation types based on historical disturbance regimes (Table 2) that vary in geography, site condition, and tree species. A second scheme (Miller et al. 2008 and references therein) classifies piñon-juniper vegetation in the Great Basin in terms of "phases". This system has become widely referenced in areas where Greater Sage-Grouse (*Centrocercus urophasianus*) are a management focus. The descriptions shown in Table 2 refer to the early (Phase I), middle (Phase II), and late (Phase III) successional phases of woodland development (Miller et al. 2008).

Somershoe et al. 2020

Table 2. Descriptions of piñon-juniper woodland types (Romme et al. 2009) and successional stages (phases) in the Great Basin (Miller et al. 2008).

Piñon-juniper vegetation types
Persistent piñon-juniper woodlands: sparse to dense tree cover, typically in rugged areas with coarse soils, and with minimal herbaceous ground cover (Figures 8–13)
Piñon-juniper savannas: low to moderate tree density with dense, nearly continuous grass and forb understory on coarse- to fine-textured soils (Figures 14 and 15)
Wooded shrublands: variable tree density, from very sparse to relatively dense, and shrubs are the dominant understory plants, including sagebrush (*Artemisia* spp.) (Figures 16–20)
Piñon-juniper woodland phases (Great Basin)
Phase I: trees present, but shrubs and herbaceous cover are the dominant vegetation type (Figures 16–18)
Phase II: trees co-dominant with shrubs and herbaceous cover (Figures 8 and 18)
Phase III: trees are dominant vegetation (Figures 9, 11, and 18)

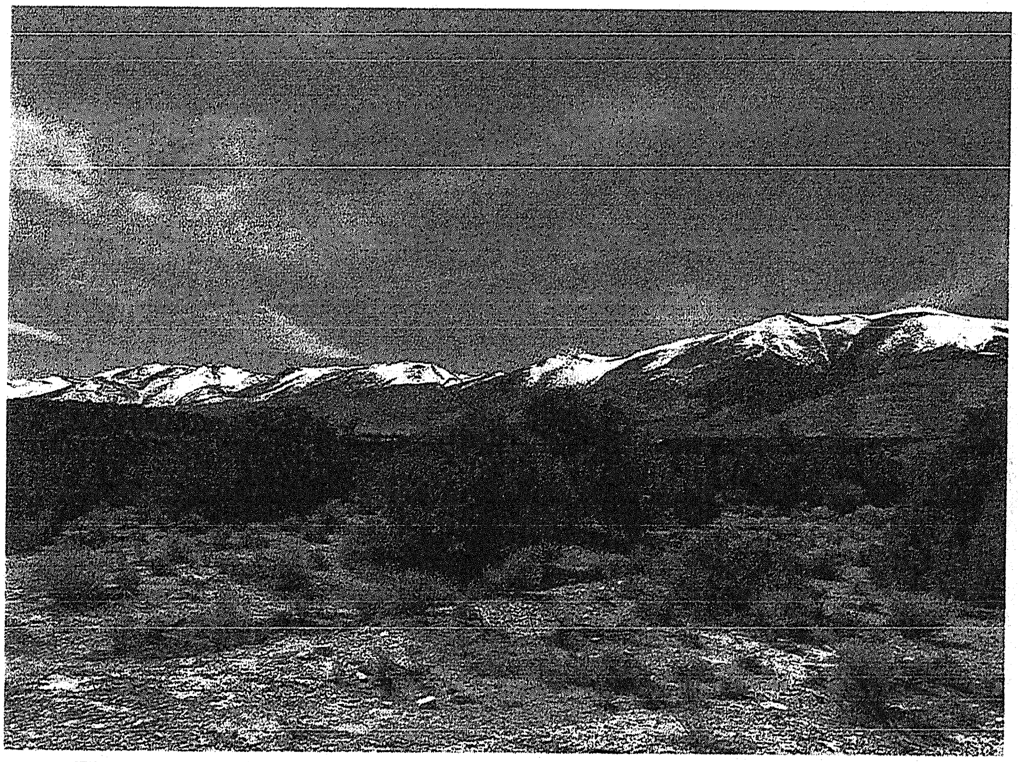

Figure 8. Medium density piñon-juniper wooded shrubland (Phase II) near a nesting colony, western Nevada, April 2018. Photo by Scott Somershoe.

Figure 9. High density persistent piñon-juniper woodland (Phase III, background), western Nevada, April 2018. Photo by Scott Somershoe.

Figure 10. Persistent piñon-juniper woodland with Utah juniper and Colorado piñon, northwestern New Mexico. Photo by Natural Heritage New Mexico.

Figure 11. Persistent piñon-juniper woodland in Nevada (Phase III). Photo by Great Basin Bird Observatory.

Figure 12. Persistent piñon-juniper woodland with Colorado piñon, White Sands Missile Range, southern New Mexico. Photo by Natural Heritage New Mexico.

Figure 13. Persistent piñon-juniper woodland, Mesa Verde National Park, Colorado. Photo by Scott Somershoe.

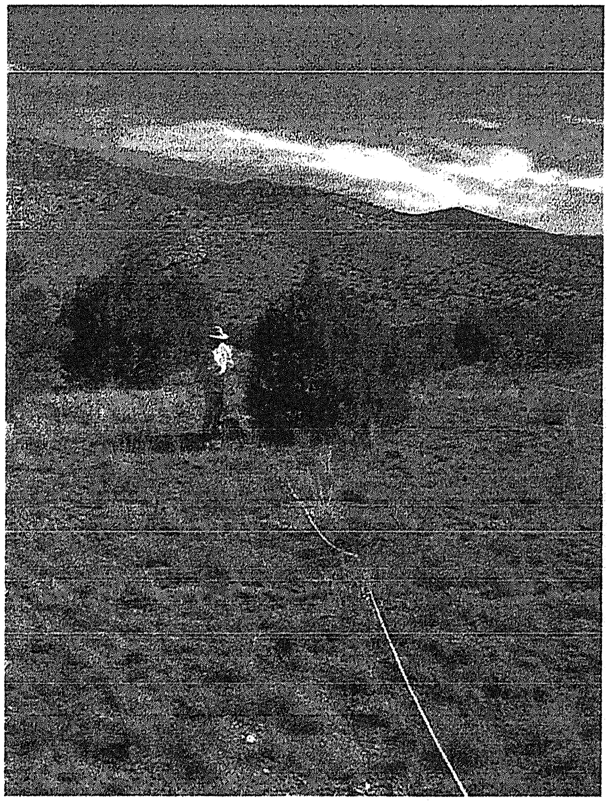

Figure 14. Piñon-juniper savanna, New Mexico. Photo by Natural Heritage New Mexico.

Figure 15. Piñon-juniper savanna, central New Mexico.
Photo by Natural Heritage New Mexico.

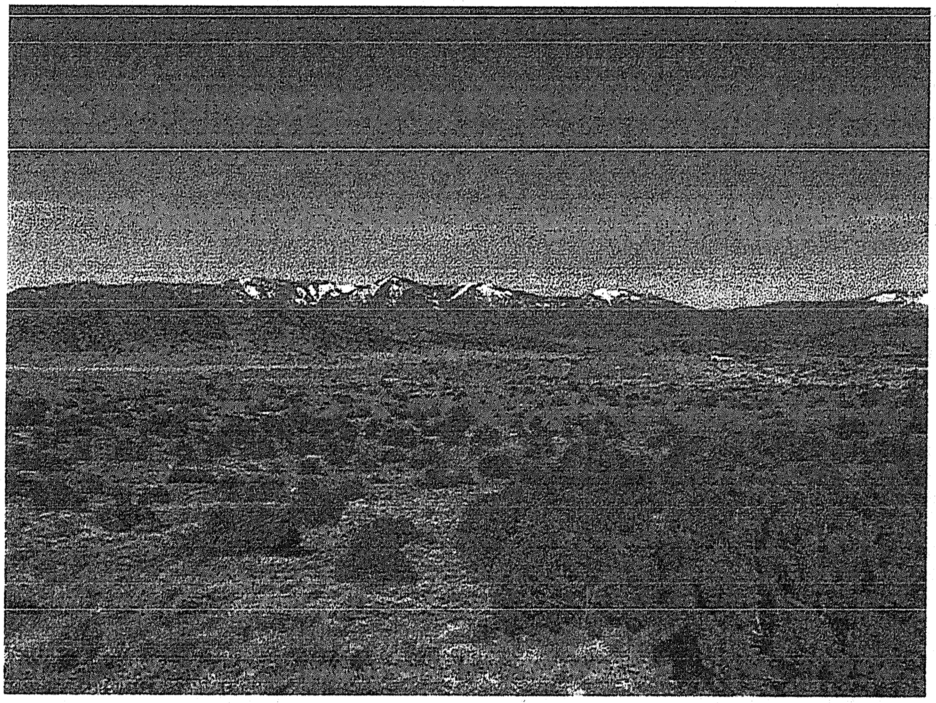

Figure 16. Low density piñon-juniper wooded shrubland (Phase I), western Nevada, April 2018. Photo by Scott Somershoe.

Figure 17. Wooded shrubland (Phase I), Nevada.
Photo by Great Basin Bird Observatory.

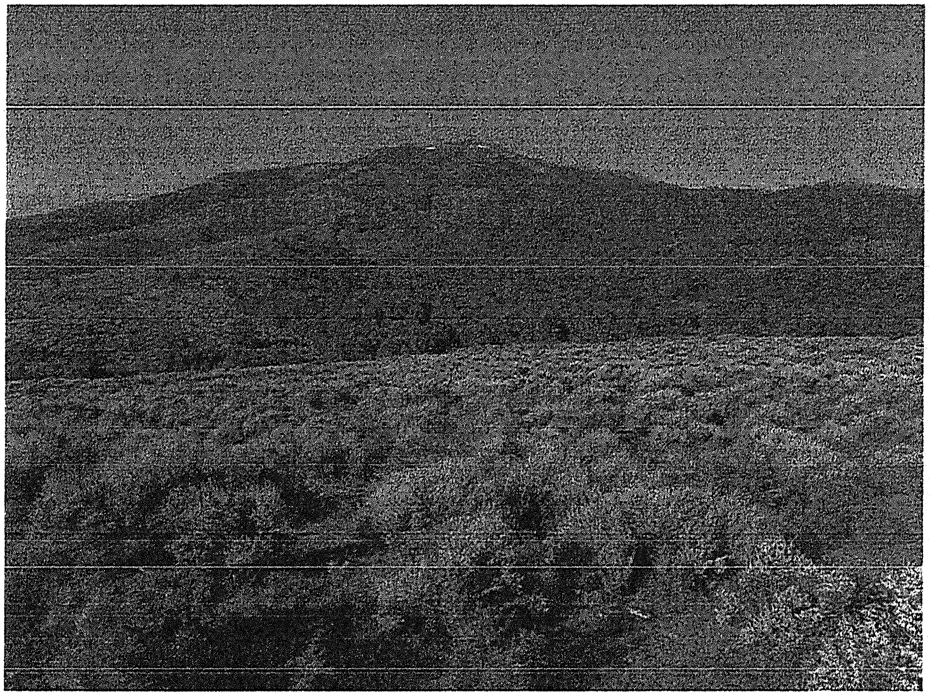

Figure 18. Wooded shrubland (multiple phases), Nevada.
Photo by Great Basin Bird Observatory.

Figure 19. Wooded shrubland with Colorado piñon and big sagebrush, northwestern New Mexico. Photo by Natural Heritage New Mexico.

Figure 20. Wooded shrubland with Colorado piñon and big sagebrush, northwestern New Mexico. Photo by Natural Heritage New Mexico.

Regional Patterns of Pinyon Jay Habitat Use and Associations

Overview

Studies of Pinyon Jay habitat use and associations have been concentrated in New Mexico and the central Great Basin in Nevada. Available descriptions of vegetation type and habitat characteristics associated with Pinyon Jay nesting, foraging, and caching are relatively general, but appear to vary between these two regions in several respects (Johnson et al. 2014, 2015, 2018a; Boone et al. 2018; J. Boone and E. Ammon, unpublished data). Additionally, characteristics of habitat used in either region may vary seasonally within and among flock home ranges. Pinyon Jays also appear to select distinctive subsets of the woodland landscape for different activities. For instance, in both New Mexico and the Great Basin, jays use lower density persistent piñon-juniper woodlands, wooded shrublands, and piñon-juniper savannas for caching piñon nuts and foraging for other food items (Phase I, Table 2, Miller et al. 2008), but tend to use denser persistent piñon-juniper woodlands and wooded shrublands for nesting, though there is significant variation in tree density and percent canopy cover across nesting sites (Johnson et al. 2014, 2015, 2016, 2018a; Johnson and Sadoti 2019; J. Boone and E. Ammon, unpublished data). During the non-breeding season, Pinyon Jays may use a wider variety of habitats, including suburban neighborhoods and bird feeders that provide additional and/or supplemental food resources when piñon nut availability is limited (Balda 2002).

New Mexico

In New Mexico, Pinyon Jays nest in various piñon-juniper habitats, including dense to sparse persistent piñon-juniper woodland (Johnson et al. 2014, 2017b; Petersen et al. 2014) and juniper-dominated savanna (Petersen et al. 2014). Compared to random sites on the landscape at the nesting colony scale, a study in New Mexico found colonies more likely to occur on gradual slopes, with a low heat load, in large patches of dense piñon-juniper habitat (Johnson et al. 2017b). Jays nest in larger than average trees within persistent piñon-juniper woodlands in areas of relatively high canopy cover (Johnson et al. 2014, 2015) and high tree density (e.g., up to 2,725 tree per acre, 1,102 trees her hectare) (Johnson et al. 2018a). Large trees with dense crowns, but not the largest emergent trees, appear to be favored as nest trees (Wiggins 2005, Johnson et al. 2014, 2015). Larger trees also produce more nuts (Parmenter et al. 2018). Johnson and Sadoti (2019) also found that tree height and diameter, stem density, and canopy cover within 5.5 yards (5 meters) were predictive of nest location in nest models at four study sites and nine nesting colonies. An assessment of model transferability between nesting sites indicated that the application of information from one nesting area to the management of another may be effective but can also result in significant loss of nesting habitat; hence, there is apparently no one-size-fits-all habitat management prescription for Pinyon Jays (Johnson and Sadoti 2019).

Pinyon Jays in New Mexico use lower-elevation piñon-juniper savanna, shrubland, and grassland habitats in the nonbreeding season (Johnson et al. 2014). Cache sites are variable, including open areas of juniper savanna or previously burned areas (Johnson et al. 2010) and dense piñon woodland (Johnson and Smith 2006).

Great Basin

In the Great Basin region of Nevada and southern Idaho, Pinyon Jays are most likely to occur in the lower-to-middle elevation portions of the piñon-juniper zone (Figures 21 and 22) (J. Boone and E. Ammon, unpublished data). They tend to cache in flatter areas where canopy cover is low and where there is pronounced interspersion of woodlands and adjoining shrublands (Phase I), and they sometimes cache in pure sagebrush up to four miles (6.4 kilometers) from a woodland edge. They typically forage within 437 yards (400 meters) of the woodland-sagebrush ecotone during the nesting and fall seasons. Nest colony sites tend to have somewhat denser tree cover (typically Phase II) than caching sites (J. Boone and E. Ammon, unpublished data). Roost sites are found in relatively high-density stands, usually within ~550 yards (500 meters) of the nesting colony. Denser woodland interiors (Phase III) at higher elevations tend to be avoided for most daily activities, with the possible exception of roosting (J. Boone and E. Ammon, unpublished data).

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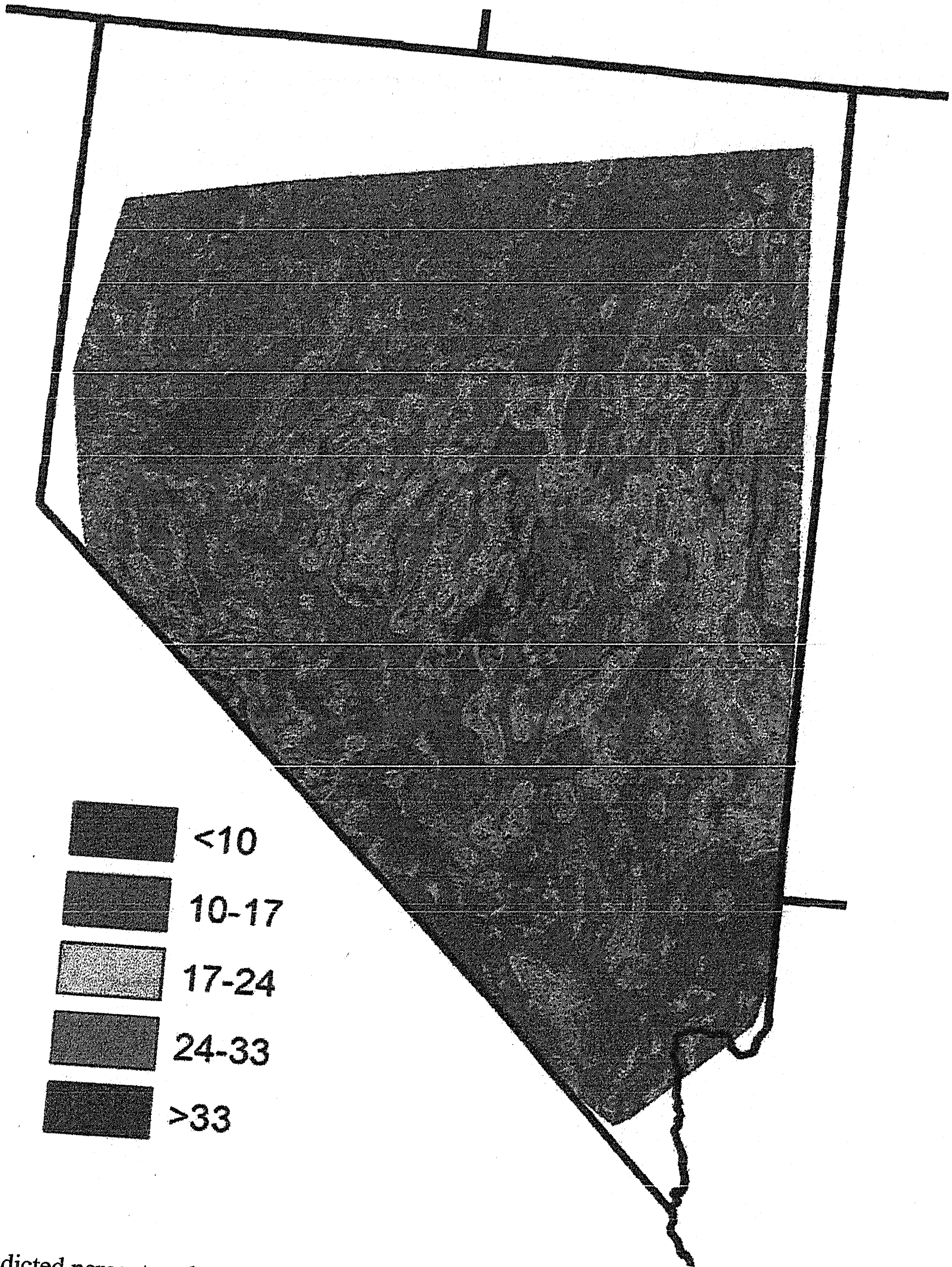

Figure 21. Predicted percent probability of occurrence for Pinyon Jays within the species' range, based on random forest tree regression analysis of Nevada Bird Count data (J. Boone and E. Ammon, unpublished data). The model predicts highest occurrence probability at lower elevations of the mountain ranges along the woodland-shrubland ecotone. Map courtesy of Great Basin Bird Observatory.

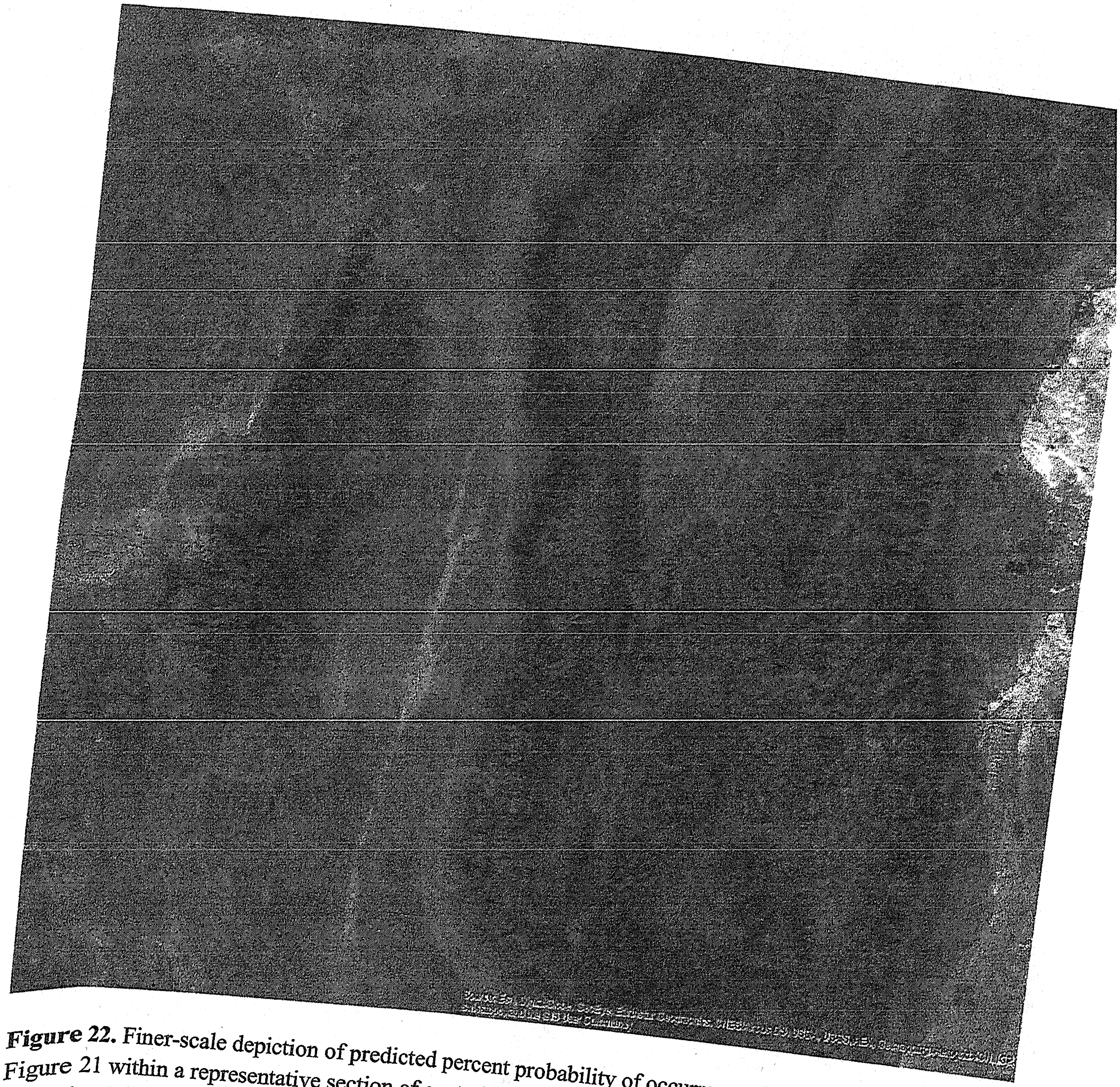

Figure 22. Finer-scale depiction of predicted percent probability of occurrence for Pinyon Jays from Figure 21 within a representative section of central Nevada based on random forest tree regression analysis of Nevada Bird Count data (J. Boone and E. Ammon, unpublished data). Deeper reddish shading indicates higher probability of occurrence. This figure highlights Pinyon Jay preference for lower elevation piñon-juniper woodlands that are adjacent to and transitioning into sagebrush shrubland. Map courtesy of Great Basin Bird Observatory.

Other Regions

Outside of the range of piñon pine, Pinyon Jays are usually found in low-elevation pine-juni woodlands or open pine woodlands with less juniper. In many of these areas, there is considerable interspersed grassland or shrub-steppe where Pinyon Jays forage and cache nuts nesting colony site in Carbon County, Montana was dominated by Utah juniper (*J. osteosperm* with a very low density of limber pine, and extensive bare ground (Figure 23; J. Marks, pers. comm.). Recent nesting in South Dakota has been concentrated at low elevations of the southwestern Black Hills in dry, sparse ponderosa pine woods and scrub habitat with interspersed grassland (Drilling et al. 2018). Silcock and Jorgenson (2018) suggest jays use ponderosa pine in western Nebraska. In California, Pinyon Jays use Jeffrey pine woodlands (Grenfell and Laudenslayer 1983), while in Oregon they occupy juniper and ponderosa pine transition habitats (Marshall et al. 2003).

In Colorado, nesting is primarily restricted to piñon-juniper woodlands, but adults feeding fledglings have also been observed in ponderosa pine, riparian and shrubland habitats (Wickersham 2016). In southwestern Colorado, jays were not observed in extensive ponderosa pine forests (Wickersham 2016). In 2019, Pinyon Jay nests were found primarily in junipers within moderately dense to sparse piñon-juniper woodland/shrub areas (L. Rossi, unpublished data).

In southeastern Idaho, Brody (1992) found nesting colonies in woodlands consisting solely of junipers, with no piñon pines present (n = 64 nests). Another flock nested in a mixed site with both piñon and junipers, though 12 of 13 nests were placed in junipers.

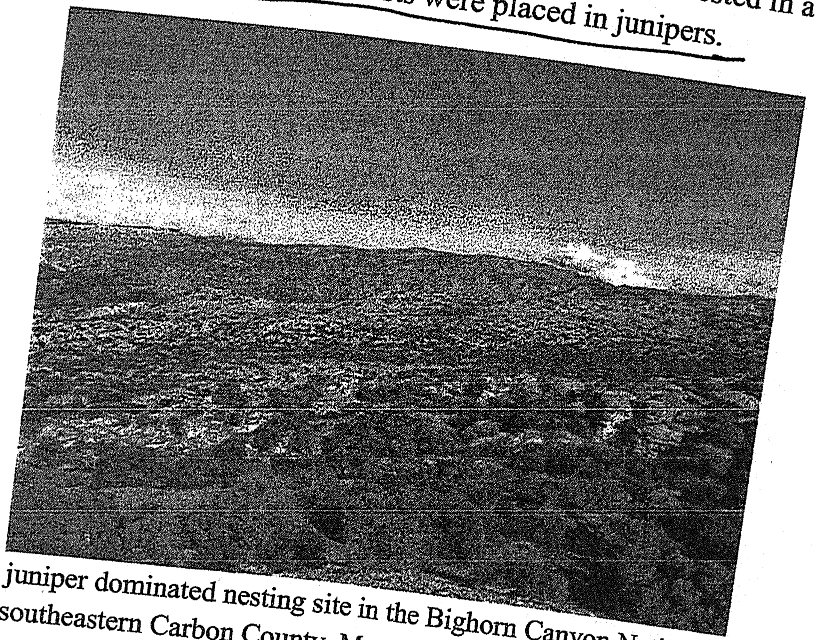

Figure 23. Utah juniper dominated nesting site in the Bighorn Canyon National Recreation Area, southeastern Carbon County, Montana. Photo by Jeffrey Marks.

Chapter 3. Causes of Population Declines: Historical and Current Threats

Overview

The causes of Pinyon Jay population declines are poorly understood. Several causal factors have been hypothesized, and these factors could interact, vary spatially within the species' range, and may have varied temporally over recent decades and centuries. In this section, we describe some of the processes that could contribute to Pinyon Jay declines, but we emphasize that additional investigation is needed to understand which of these potential threats are most important, and how they function and interact in different portions of the species' range.

Changes in Habitat Extent and Vegetation Structure

In some regions and during some periods, significant areas of piñon-juniper woodland have been removed by humans for a variety of reasons (see below). In other regions and at other times, the extent of piñon-juniper woodlands is thought to have increased compared to the pre-settlement period, along with the changes to the properties of those woodlands, including tree density and age structure. Fire suppression, land use and climate change may also have altered habitat extent and structure in potentially important ways. Some or all of these factors could also have affected piñon nut productivity, but this question is largely unexplored. Complicating matters, there is uncertainty about the "normal" historical reference condition of woodlands used by Pinyon Jays, making it difficult to define and identify undesirable "departure" conditions.

dupe

In the Great Basin, large scale clearing of piñon-juniper woodlands began as early as the Nevada silver mining boom of 1859–1880 to provide cooking and heating fuel and material for fencing and other construction. Similar clearing occurred more broadly across the western United States to support mining activities and to create rangeland for livestock (Morris and Rowe 2014). At roughly the same time, a period of woodland expansion peaked during 1880–1920, which coincided with a relatively mild and wet climate, the introduction of livestock, and reduction of fire (Miller et al. 2008, 2019). Whether this expansion was a response to clearing activities (Lanner 1981) versus an independent process is unclear. Regardless, widespread down- and up-slope expansion and infilling of western juniper and piñon-juniper woodlands occurred in the Great Basin, sometimes culminating in a closed or nearly-closed canopy (Bradley and Fleischman 2008, Friggens et al. 2012, Morris and Rowe 2014, Miller et al. 2019). The extent, timing, and rate of expansion have been highly variable at the local scale, and often related to land use history, soil type, and climate (Gedney et al. 1999, Morris and Rowe 2014). Baker and Shinneman (2004) and Shinneman and Baker (2009) suggest that increasing tree density is probably not a result of fire suppression, but rather driven by climate, grazing, and long-rotation high-severity fires. Miller et al. (2019) note that the aforementioned causes of expansion and infilling are not well understood and there is no scientific consensus on the relative importance of each of these variables. Some researchers and managers have considered this expansion and infill dynamic to be a problematic departure condition that occurred at a landscape scale (Miller et al. references, Chamber et al. 2019), but others have suggested that expansion and contraction patterns have varied significantly at the local scale, and that the overall dynamic at the landscape scale may fall within the normal range of historical variability (Lanner 1981, Sallach 1986, Belsky 1996, Manier et al. 2005, Miller et al. 2008, Romme et al. 2009). Miller et al. (2019) suggests that increasing atmospheric carbon dioxide levels could be an important factor driving current woodland expansion.

Contemporary Habitat Dynamics and Vegetation Management Activities

From 1950 to the late 1990s, piñon-juniper expansion appears to have decreased considerably and may have ceased in some areas, possibly due to a prolonged drought across most of the piñon-juniper range, with regional exceptions (Miller et al. 2008, 2019; Sankey and Germino 2008; Bradley 2010). This general climate pattern is expected to continue (Thompson et al. 1998, Cole et al. 2008, McDowell et al. 2016). Concurrently, piñon-juniper woodlands have also been removed or thinned over the last 70 years by multiple management agencies and private landowners using primarily mechanical and also herbicide treatments. The reasons for these treatments include the following:

1) **Creating rangeland for livestock**: From 1950–1964, the U.S. Forest Service used the method called "chaining" to clear millions of acres of piñon woodlands to create pasture for cattle grazing (Lanner 1981). Many of these woodlands have regenerated and now support dense stands of young piñon or junipers (Romme et al. 2009). In Arizona, approximately 1.2 million acres (485,000 hectares) were cleared between 1950 and 1964 (Arnold et al. 1964) with the rationale that piñon-juniper had invaded historical grassland and shrub-steppe habitat, a hypothesis that has been challenged (Lanner 1981).
2) **Habitat enhancement**: Removing and thinning piñon-juniper woodlands to increase habitat suitability for focal game species (e.g., mule deer, pronghorn, elk), increase forage production, and improve watershed conditions became common in the 1950s–1970s (Johnson 1967, Terrel and Spillet 1975, Ffolliott and Stropki 2008). Starting in the 1970s, a focus on game species management continued (Gottfried and Severson 1994, Arizona Game and Fish Department 2014, Kramer et al. 2015), but there was also attention directed to other species of conservation concern across much of the Pinyon Jay's range (Gottfried and Severson 1994, Morris and Rowe 2014). The most notable and widespread example of this newer management practice is removal of woodlands (mostly at their lower elevation boundary) to maintain or create additional sagebrush habitat for Greater Sage-Grouse in the Great Basin. Sage-grouse numbers have declined in the West, and significant conservation and recovery efforts, including complete removal of pines and junipers within sagebrush (*Artemisia* spp.) communities, is one of the primary conservation strategies to restore habitat and promote species recovery (Baruch-Mordo et

al. 2013, NRCS 2015, Chambers et al. 2017, Donnelly et al. 2017). From 2010 to 2017, private landowners and partners treated over 617,000 acres (250,000 hectares) of conifers through the Natural Resource Conservation Service's (NRCS) Sage Grouse Initiative (NRCS, unpublished data, Maestas et al. unpublished data). Similarly, over 494,211 acres (200,000 hectare) have been treated in Utah through the Watershed Restoration Initiative (Chambers et al. 2017). ✓

3) **Fuels reduction and fire prevention**: Piñon-juniper woodlands may be thinned or cleared to slow movement of potential wildfires and/or create fire breaks (Schwilk et al. 2009, Miller et al. 2019), especially in the Southwest. Thinning is also implemented to improve biodiversity and ecological conditions within the woodland matrix (Bombaci and Pejchar 2016, Bombaci et al. 2017, Holmes et al. 2017), and to increase piñon pine health and drought resistance via reduction in conspecific competition. These assumptions have not been thoroughly tested, and existing data do not consistently support the ideas that thinning enhances biodiversity (Bombaci and Pejchar 2016, Bombaci et al. 2017) or piñon health (Clifford et al. 2008, Holmes et al. 2017, Morillas et al. 2017).

The effects of thinning treatments on Pinyon Jays have been studied, but little information is available about the effects of woodland removal, especially in the Great Basin. A recent review by Bombaci and Pejchar (2016) reported that although woodland thinning had non-significant impacts to most wildlife species, the majority of detected negative responses involved woodland birds. At a more local scale, one study found thinning treatments that reduced canopy cover from 36% to 5% reduced local-level occupancy by Pinyon Jays in treated areas in the Southwest (44–190 acres [18–77 hectares], Magee et al. 2019). Another southwestern study found that Pinyon Jays stopped nesting within parts of a known colony site after the colony site was significantly thinned (87% reduction of trees per acre, specifically 1,893 to 248 trees per acre [766 to 100 trees per hectare]). However, a few birds nested (with unknown reproductive success) in untreated woodlands immediately adjacent to the treated area, suggesting fidelity to the traditional site (Johnson et al. 2018b). These findings suggest that shifting nesting sites to an adjacent untreated area depends on the availability of potentially suitable habitat, which cannot be assumed. Other studies appear to confirm that Pinyon Jays may be sensitive to habitat "quality". For example, Pinyon Jays appeared to abandon colony sites when tree vigor declined in association with low winter precipitation (Johnson et al. 2017c). CEA

The cumulative effects of multiple woodland treatments on Pinyon Jays, especially at the landscape scale, have not been studied. Treatment planning is becoming more nuanced, and now considers ecological site potential, phases of woodland succession, and ecosystem resilience to treatments (disturbance) and resistance to invasive plants (Tausch et al. 2009, Miller at el. 2014); so opportunities may exist to incorporate measures for Pinyon Jay conservation.

Climate Effects on Habitat

The effects of climate change on piñon-juniper woodlands may include large-scale increased mortality rates of piñon pines (Breshears et al. 2005, 2008; Mueller, et al. 2005; Shaw et al. 2005; Clifford et al. 2008, 2013; Greenwood and Weisberg 2008; Adams et al. 2009; Romme et al. 2009; Meddens et al. 2015; Flake and Weisberg 2018; Friggens et al. 2018), significant reductions in canopy cover (Clifford et al. 2011), declines in piñon nut production (Redmond et al. 2012, Wion et al. 2019), and reductions in piñon tree vigor (Johnson et al. 2017c). Prolonged drought appears to facilitate outbreaks of Ips beetles (*Ips confusus*) causing mortality for both *P. edulis* and *P. monophylla* (Shaw et al. 2005, Clifford et al. 2008). Larger, older piñons, which generally produce the majority of piñon nut mast, are typically more susceptible to mortality by Ips beetles (Shaw et al. 2005, Clifford et al. 2008), but all tree sizes may be affected (Mueller et al. 2005, Shaw et al. 2005, Wiggins 2005, Clifford et al. 2008, Greenwood and Weisberg 2008). Various climate models predict distributional changes of piñon-juniper woodlands (Thompson et al. 1998; Rehfeldt et al. 2006, 2012; Cole et al. 2008; Rondeau et al. 2017) and widespread piñon and juniper mortality across the southwestern U.S. (Rehfeldt et al. 2006, 2012; McDowell et al. 2016). Indirect impacts of climate change could include increased incidence and severity of wildfire (Floyd et al. 2004, Miller et al. 2019) and insect outbreaks (Mopper and Whitham 1992, Romme et al. 2009, Gaylord et al 2013). Meddens et al. (2015) summarize studies assessing the effects of drought, tree density, elevation, beetle outbreaks, and their possible interactions on piñon pine mortality. Miller et al. (2019) summarizes effects of a changing climate and potential changes to piñon-juniper woodlands. Although most of the predicted effects of climate change on piñon-juniper woodlands are negative, the severity will likely vary regionally, and the overall impacts on Pinyon Jays remain to be further explored.

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Other Threats

Energy infrastructure development in piñon-juniper woodlands could be detrimental to local Pinyon Jay populations either through direct habitat loss or indirect impacts such as disturbance via traffic or noise (Kleist et al. 2018). Commercial piñon nut collection may also affect Pinyon Jays, owing to continued high demand, large economic value, and reduced availability of piñon nuts. Collection methods may damage trees and disturb the soils and hydrology if heavy machinery is used. The decline of insect populations (Collen et al. 2012, Dirzo et al. 2014) may also be a threat; further research is needed to determine which insects consumed by Pinyon Jays may be declining.

Chapter 4. Research Needs

Overview

The majority of research on Pinyon Jays has been conducted in the southern and western portions of their range, which coincide with core population centers of the species (Figures 2 and 4, Table 1). Social behavior has been thoroughly studied in a suburban population near Flagstaff, Arizona (Marzluff and Balda 1992, Balda 2002) and this work still provides most of the knowledge regarding the natural history of Pinyon Jays. It may, however, not be representative of Pinyon Jays across their range because the data are from a single population in a possibly-atypical suburban locality (Marzluff and Balda 1992, Balda 2002). Recent studies in New Mexico (Johnson et al. 2014, 2015, 2016, 2017a, 2017b, 2018a, 2018b) have modeled Pinyon Jay habitat use at the nest, colony, and landscape scales. In Nevada, Boone et al. (2018, and unpublished data) have studied Pinyon Jay habitat use, and characterized landscape use in species distribution modeling and home range mapping. Much remains to be studied about home ranges, nesting colonies, specific habitat requirements during nesting and other life stages, and other features of Pinyon Jay natural history across the majority of the species' geographic distribution. Studies that incorporate measures of Pinyon Jay productivity, non-breeding season survival, recruitment, and other demographic factors may shed light on the specific life cycle phases that are the strongest drivers of population declines. In addition, studies are needed to determine habitat and landscape factors driving habitat suitability at the home range scale, including: piñon nut productivity and distribution; the extent, structure, density, and understory of woodlands at the home range scale; and the differences between occupied and unoccupied woodlands. Because this species' range includes several ecoregions, geographically specific information is necessary to develop appropriate conservation and management strategies for the Pinyon Jay.

Specific Research Needs

Significant additional information is needed to effectively manage Pinyon Jays. These research needs are listed below in five categories, each with various subcategories. This presentation order does not represent a prioritization or reflect the relative importance of each topic. We also note where relevant information has already been collected using the heading "Current Knowledge".

Pinyon Jay Distribution, Abundance, and Population Trends

The overall Pinyon Jay breeding range has been fairly well delineated (Figures 2 and 4); however, finer-grained location data are still very limited, and increased survey coverage and development of standardized protocols will increase our knowledge and understanding of this species' local distribution, abundance, population trends, and habitat use and associations.

Specific research needs are:

a. Document locations of flocks, home ranges, and nesting colonies across the Pinyon Jay's range. Baseline information on Pinyon Jay local and regional distribution and populations, including locations of breeding colonies, home ranges, and population estimates, are needed across most of the range. It would also be useful to add data from parts of the Pinyon Jay range that occur away from **piñon** pines. **Current knowledge:** Home ranges and/or nesting colonies have been primarily located or mapped using VHF radio-tracked birds in New Mexico (Southern Rockies/Colorado Plateau, BCR 16) (Johnson et al. 2014, 2016, 2017a), the mountains of western and central Nevada and southern Idaho (Great Basin, BCR 9, see Figure 4) (J. Boone and E. Ammon, unpublished data), and Flagstaff, Arizona (Sierra Madre Occidental, BCR 34) (Marzluff and Balda 1992). Recently, successful documentation of home ranges, cache sites, roost sites, and nesting colonies was completed without radio-telemetry through direct observation and delineation of Pinyon Jay activity centers and movements (J. Boone and E. Ammon, unpublished data).

b. Monitor Pinyon Jay nesting colonies to assess stability in population size over time. By monitoring known nesting colonies over multiple years, the stability of flock size can be determined, along with nest colony site fidelity. **Current knowledge:** Nesting colonies have been located, primarily in New Mexico, Nevada, and Arizona (BCRs 9, 16, 34); some locations have been monitored over multiple years (see Balda 2002, Johnson et al. references).

c. Develop and implement standardized survey protocol to monitor Pinyon Jays across their range. A robust, standardized and repeatable survey protocol is needed in order to collect statistically rigorous data to estimate abundance and population trends at different spatial scales. **Current knowledge:** Peterson et al. (2014) provide a non-statistical monitoring protocol to find and monitor nesting colonies. A statistically robust breeding season survey was developed in Colorado and Utah in 2019. This survey could serve as a basis for a protocol that could be used across the range of the Pinyon Jay, while allowing for modifications of scale and other considerations (S. Somershoe, pers. obs.).

d. Determine causes of local/regional increases or decreases in Pinyon Jay populations. According to the BBS trend map for Pinyon Jay, populations appear to be increasing or decreasing significantly in different portions of the range (Figure 3). Additional monitoring data are needed in these areas to assess and refine trend estimates identified by BBS. Therefore, it would be informative to identify some core populations across the range, including suburban populations, and conduct an integrated trend analysis using BBS, IMBCR, eBird, and Christmas Bird Count (CBC) data, along with other relevant research. With trends identified, potential causes of trend direction and magnitude could be investigated analytically. **Current knowledge:** General summaries of BBS data (Sauer et al. 2017).

Pinyon Jay Habitat Use and Requirements

Little is known about how Pinyon Jays use different woodland structures throughout the year, how this use varies regionally, and what specific habitat or landscape features jays select for. Data from a limited number of local and regional studies may not be applicable outside a given region or BCR, and geographically specific information is required to develop effective conservation actions for the species.

Specific research needs are:

a. Assess and quantify habitat structure, composition, and piñon nut availability within Pinyon Jay home ranges. Available information suggests that habitat associations may vary regionally. Further work is needed to identify range wide or regional patterns of habitat use. Piñon nut availability could be a critical covariate of Pinyon Jay occupancy and should be included in data collection whenever possible. **Current knowledge:** Habitat structure and piñon nut production have been assessed primarily in home ranges in New Mexico (BCR16) (Johnson et al. 2014, 2015, 2017b; Johnson and Sadoti 2019), western and central Nevada (BCR 9) (J. Boone and E. Ammon, unpublished data), and southern Idaho (BCR 9) (Brody 1992).

b. Assess habitat structure used by Pinyon Jays during specific parts of the daily and annual cycle. Habitat used by Pinyon Jays for different parts of their daily cycles (e.g., nesting, foraging, caching, and roosting) and annual cycles should be characterized and compared. Further distinctions in habitat use and movements could be made between birds in breeding condition (e.g., males with cloacal protuberance present) and birds in non-breeding condition (e.g., immature birds and non-breeding adults), if these designations can be made when birds are captured. **Current knowledge**: Habitat use has been primarily assessed in New Mexico (BCR 16) (Johnson et al. 2014, 2015, 2017b, 2018a; Johnson and Sadoti 2019) and southern Idaho (BCR 9) (Brody 1992), and through the use of VHF radio telemetry in central Nevada (BCR 9) (J. Boone, unpublished data).

c. Assess Pinyon Jay responses to vegetation management within home ranges. Jay responses (e.g., occupancy, abundance or density, nest success, productivity, survival) to vegetation management activities at various locations and temporal scales (short- vs. long-term, 0-3 years vs. 10+ years) need to be documented across the range. Through collaborations with natural resource managers, opportunities may exist to design thinning, complete tree removal, or herbicide treatments to accomplish not only their

b. Assess environmental and vegetation structure characteristics that may influence Pinyon Jay nest colony and nest site selection. The environmental and vegetation structure attributes that may influence colony site and nest site selection remain to be determined. Candidate factors include structural properties of colony sites or nest trees, thermal properties, local piñon nut productivity, and distance to nut producing trees. Documenting shifts in colony location may also help to define suitable Pinyon Jay nesting habitat. **Current knowledge:** Unknown except for studies in New Mexico (BCR 16) (Johnson et al. 2014, 2015, 2016, 2017c, 2018b; Johnson and Sadoti 2019) and Nevada (BCR 9) (J. Boone and E. Ammon, unpublished data).

c. Investigate relationships between Pinyon Jay reproductive success and habitat and landscape covariates. Nest success could be related to specific habitat covariates at the colony and/or nest level, including distance to piñon nut sources and caches, size of piñon mast crops, woodland type and density, elevation, etc. Additionally, consider landscape-level covariates such as distance to water sources, various types of development (e.g., energy development and roads), and other land cover types such as residential and agricultural. **Current knowledge:** Assessments have been conducted in New Mexico (BCR 16) (Johnson et al. 2017a, 2018a, 2019), southern Idaho (BCR 9) (Brody 1992), and Flagstaff, Arizona (BCR 34) (Balda 2002).

d. Assess movement of Pinyon Jay nesting colonies. Pinyon Jays may sometimes shift location of nesting colonies between years. There is a need to understand causes of colony shift and the ability of a flock to move a colony site or home range in response to habitat changes or loss (e.g. fire, vegetation management, etc.). **Current knowledge:** In northern Arizona, one flock initiated nesting at 24 different sites over 12 years of observation (BCR 34) (Marzluff and Balda 1992). A nesting colony in New Mexico shifted location 550 and 1640 yards (500 and 1500 meters) when piñon tree condition declined (BCR 16) (Johnson et al. 2017c). At another New Mexico site, jays avoided nesting in treated areas within the boundaries of a traditional colony site after a treatment was implemented (Johnson et al. 2018b). A Nevada study found that three colonies shifted their colony site by as much as 550 yards (500 meters) without the presence of treatments at or near the colony site (BCR 9) (J. Boone and E. Ammon, unpublished data).

Piñon Pine Biology and Woodland Dynamics

Extensive research has been conducted on piñon pines and piñon-juniper woodlands, but limited information is available on how temperature, precipitation, changing climate, tree density, tree age, and other variables can affect tree mortality rates and mast production. Obtaining this information in a robust and regionally-specific manner may require long-term studies, but these are essential to better understand the dynamics of woodland ecology, effects to Pinyon Jays, and how to effectively manage woodlands for wildlife and improved ecological health.

Specific research needs are:

a. Assess woodland attributes that influence tree vigor, survival, and piñon nut production. Basic research on the microhabitat requirements of piñon pine, factors affecting piñon nut productivity, and tree survival is needed to manage Pinyon Jay habitats given their dependence on piñon pine health and piñon nut production. **Current knowledge:** Johnson et al. (2017c) found piñon tree vigor in New Mexico was negatively associated with increased tree size and density. Flake and Weisburg (2018) assessed how various environmental factors are related to piñon pine mortality in central Nevada (BCR 9). Meddens et al. (2015) summarize literature on the causes of piñon pine mortality and how they relate to variables such as tree density and elevation.

b. Assess weather and climate attributes that affect annual piñon nut production and tree vigor. Characterizing the climate, microclimate, and weather attributes that lead to nut production in non-mast and mast years will allow for better management of potentially critical pine stands in the overall landscape. **Current knowledge:** Zlotin and Parmenter (2008) and Parmenter et al. (2018) reported on a multi-decade study on weather variables and piñon mast production in New Mexico (BCR 16). They found that *P. edulis* mast production is negatively associated with spring and summer temperatures of the current year. While the current year's precipitation did not appear to affect production, total annual precipitation was positively correlated to production with a lag time of two years (during primordia formation) and one year (during strobili/green cone development). In semi-arid woodlands, these authors predicted that frequent drought and high heat will result in more years with low or no mast production. Results may not be applicable for other *Pinus* spp. or other BCRs. Breshears et al. (2005) (BCRs 9 and 16), Shaw et al. (2005) (BCR 16), and Greenwood and Weisberg (2008) (BCR 9) summarize impacts of drought on piñon pine mortality. Meddens et al. (2015) summarize literature on the causes of piñon pine mortality, including climatic variables. Johnson et al. (2017c) found piñon tree vigor declined with reduced winter precipitation (BCR 16).

Critical Pine Flowers.

c. Develop a spatially-explicit model to predict changes/shifts in piñon-juniper woodlands over time. Predictive models of temporal woodland dynamics as a function of climate change and other factors would facilitate identification of the most critical Pinyon Jay habitat in each state, with emphasis on designating occupied and climate-resilient areas as the highest priority for conservation (Rondeau et al. 2017, Friggens et al. 2018). These models would ideally predict how many acres will move into and out of "prime piñon nut productivity" over a defined time period, and could incorporate information about management and tree removal projects to obtain a more complete perspective of landscape change. **Current knowledge:** Rondeau et al. (2017) have of their nightly communal roost site (an entire flock roosts communally except during nesting when the females roost on the nest; E. Ammon, pers. comm.). Generally, a combination of direct observation assisted by telemetry works best to identify specific locations used by Pinyon Jays for various activities, such as caching, foraging, loafing, nesting, and roosting. However, it is critical that observers maintain sufficient distance from a flock to avoid altering their behaviors, a significant risk given that flocks post sentinel birds to keep the flock informed about any threats. Alternately, a skilled observer can, over time, become sufficiently familiar with flock movements to gather accurate locational and activity information from observation alone, without the assistance of radio-telemetered birds (J. Boone and E. Ammon, unpublished data).

Survey

Statistically robust point count surveys can be conducted during the breeding season to detect Pinyon Jays, and follow up can lead to discovery of nesting colonies. Peterson et al. (2014) provide recommendations on conducting field surveys and interpreting jay behavior. A minimum of three surveys of a given site are recommended in order to increase detection probabilities (Peterson et al. 2014). A lack of detections does not necessarily mean that the site is outside of a flock's home range, as it may be used during a different part of the annual or daily cycle. Using field methods recommended by Peterson et al. (2014), a grid-based sampling design using point counts for conducting standardized, repeatable surveys was developed and implemented in Colorado and Utah in 2019 (S. Somershoe, R. Norvell, S. Gibson, L. Rossi, pers. comm.). Based on these pilot projects, the Pinyon Jay Working Group plans to develop a scalable, standardized survey method and create a range wide survey data network for the species.

Investigations of Pinyon Jay habitat use and responses to management should consider multiple scales (e.g., Johnson et al. 2016), given flocks typically move within their home ranges, and home ranges may shift. Landscape-scale assessment of habitat use, including the identification and location of key resources utilized, typically relies on landscape-scale data such as remote sensing imagery and GIS, while characterizations of the nesting colony or nest site are best done at smaller scales. Colony-scale habitat modeling can employ mid-scale measures, including geographic and on-the-ground variables (Johnson et al. 2016, J. Boone and E. Ammon, unpublished data), while nest-site analysis typically focuses on data at the nest tree or small plot scale, including tree size and density (Johnson et al. 2014, 2015; Johnson and Sadoti 2019). On-the-ground management occurs at variable scales, ranging from 200 to 30,000 acres (80–12,140 hectares) treatments, treatments with different management objectives may occur in the same general area or adjacent to one another. These scales should also be considered when studying Pinyon Jays in areas where vegetation treatments are occurring.

Because Pinyon Jays have a complex life history and are a challenging species to study, we recommend contacting seasoned Pinyon Jay researchers willing to share their experience and knowledge and provide guidance in developing an effective research project. We recommend contacting the lead author on this strategy and other contributors to this document to identify these researchers.

Considerations for Implementation of Woodland Treatments

If a woodland treatment is conducted in an area that is, or could be, occupied by Pinyon Jays, particularly during the breeding season, the following considerations and implementation parameters may help to reduce negative impacts to Pinyon Jays. The information presented below is organized according to regions and by woodland treatment types and objectives. The bulk of this material is derived from research conducted in the Southwest (mostly New Mexico); less information is available about the Great Basin and other regions occupied by Pinyon Jays. Some of the information shown below may be applicable across the broader Pinyon Jay range, but this remains to be determined.

Southwest Region - Woodland Thinning and Herbicide Treatments

The information in this section is informed primarily by research conducted in New Mexico and to a lesser extent, Arizona. In this region, most woodland treatments involve thinning, with a limited occurrence of herbicide treatments and large scale woodland removal. The goals of these projects are often fuels reduction, but may also include reduction of juniper to increase grass and forbs for ungulates (i.e., big game) and cattle, management for other wildlife species, and watershed restoration. Treatment plans that include plans for collaboration with Pinyon Jay researchers allow managers to identify whether particular treatment parameters (e.g., different percentages of retained canopy cover and/or tree density) can meet primary management objectives while remaining within the range of suitable nesting, caching, and foraging habitat.

a. Pinyon Jays often use the same general area each year for nesting, with colony site shifts of up to 550 yards (~500 meters) between years (Marzluff and Balda 1992, Johnson et al. 2017c, J. Boone and E. Ammon, pers. comm.). Thus, if a buffer area of 550 yard (~500 meters) around a known breeding colony remains undisturbed, it allows for colony shifts across years (Johnson et al. 2017c).

b. If thinning in persistent piñon-juniper woodlands or wooded shrublands, creating a patchy-clumpy mosaic of suitable nesting habitat within the treated area, as opposed to evenly spaced thinning, allows for shifting colony locations. This treatment pattern also better mimics how fire would have impacted the landscape in persistent piñon-juniper woodlands and wooded shrublands (Romme et al. 2009). Provided that sufficient suitable habitat is retained throughout the treatment area, retaining as many larger trees as possible within areas of higher tree density and/or higher canopy cover will likely conserve more Pinyon Jay nesting habitat than thinning all size/age classes to a uniform density. For example, a guideline is to retain trees within the 25–75% quartiles of these measures at the target site or similar sites (Johnson and Sadoti 2019).

c. If using herbicide treatments in juniper or piñon-juniper woodlands, a mosaic of treated and untreated areas better mimics the natural landscape setting than large monomorphic

Chapter 7. Future Work to Advance the Conservation Strategy

This conservation strategy presents an assessment of the state of current knowledge about Pinyon Jays, identifies research and information needs, and describes various management considerations. In addition to information provided in this strategy, the Working Group has identified several projects, resources and actions needed to further facilitate planning, coordination, and data management to support Pinyon Jay conservation. These are:

a. Determine extent and acreage of piñon-juniper treatments that have occurred and are planned to assess cumulative impacts throughout the range of the Pinyon Jay.

b. Develop regional- and habitat-specific recommended management considerations.

c. Finalize and make available online standardized protocols for conducting robust surveys ~~to locate and monitor Pinyon Jays, locate and identify nests and nesting colonies, and~~ assess vegetation and habitat on the Pinyon Jay Working Group website. https://www.partnersinflight.org/resources/pinyon-jay-working-group/ survey

d. Promote and develop use of the Avian Knowledge Network (AKN) to host data from all parties conducting jay surveys. http://avianknowledge.net/

e. Develop a strategy to present information on the current status of piñon-juniper ~~woodlands, including the end of the expansion era, and begin to change the paradigm and~~ negative image and value of piñon-juniper woodlands across the Intermountain West. Evaluate utility and consider development of a conservation road show about piñon-juniper woodland systems and conservation issues.

[badkitty] Autopsy results..

From: Paul Sieracki via badkitty list (badkitty@groups.electricembers.net)
To: badkitty@groups.electricembers.net
Date: Monday, December 14, 2020, 09:06 AM MST

"The laboratory results are very informative but did not identify a single definitive cause of mortality.

Starvation, unexpected weather to blame in mass migratory songbird mortality

SANTA FE –The U.S. Geological Survey National Wildlife Health Center has analyzed representative samples of the migratory songbirds collected, cataloged and sent for analyses by New Mexico Department of Game and Fish biologists, in early September. The lab report indicates that the single abnormality shared by nearly all birds was body condition ranging from poor to severely emaciated.

Kerry Mower, the Department's wildlife disease specialist, said, "The laboratory results are very informative but did not identify a single definitive cause of mortality. However, they did find that nearly all birds were severally emaciated."

The single abnormality shared by nearly all birds was body condition ranging from poor to severely emaciated. These observations are evidence of physical exertion without nourishment to support recovery, including:

- the large breast muscles controlling birds' wings were severely shrunken;
- kidney failure was apparent in many of the birds;
- stomachs and intestines were empty of foodstuffs;
- many intestines contained small amounts of blood, which is one of the effects of starvation;
- fat deposits, the stored energy for migration, were depleted; and
- lung tissues were irritated.

The USGS National Wildlife Health Center, located in Madison, Wis., is renowned for the thoroughness of the diagnostic tests for wildlife disease diagnosis and management. The center conducted numerous tests during analyses, ruling out contagious bacterial disease, contagious viral disease including avian influenza and Newcastle disease and parasites as cause of death, as well as finding no evidence of smoke poisoning or pesticide poisoning.

From the lab reports, Department biologists know that migrating birds entered New Mexico in poor body condition and some birds were already succumbing to starvation. The unusual winter storm exacerbated conditions, likely causing birds to become disoriented and fly into objects and buildings. Some were struck by vehicles and many landed on the ground where cold temperatures, ice, snow and predators killed them.

“The Department would like to thank many partners and the public who reported mortalities across New Mexico,” said Erin Duvuvuei, the Department’s avian biologist. “Hundreds of reports were received through email, phone calls and the iNaturalist app.” Reports declined by late September.

The Department would also like to thank all of the partner organizations who were essential to this process. Partners include the: United States Fish and Wildlife Service, White Sands Missile Range, United States Forest Service, Bureau of Land Management, New Mexico Avian Conservation Partners, New Mexico State University, University of New Mexico, Audubon Southwest, Los Alamos National Laboratories and the New Mexico Wildlife Center.

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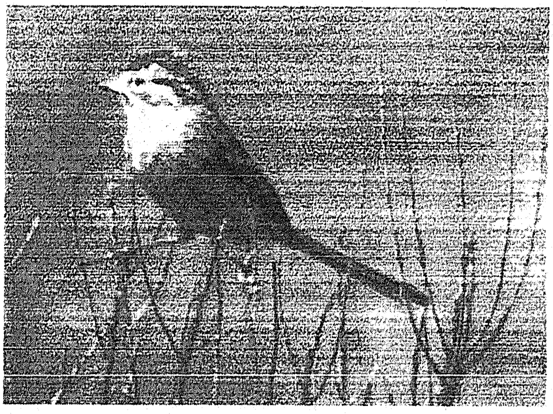
Paul Sieracki
208.217.0609
Please view my Facebook page @AltColvilleNF (Alt Colville National Forest)

BREWER'S SPARROW

(*Spizella breweri*)

Conservation Priority Score of 34



Frank Howe

ASSOCIATED SPECIES: Other Shrubsteppe obligates (Braun et al. 1976) such as Sage Thrasher, Sage Sparrow, and Sage-grouse as well as species which frequently inhabit Shrubsteppe habitats (but are not "obligates") such as Black-throated Sparrow, Northern Mockingbird, Loggerhead Shrike, Gray Flycatcher, Western Meadowlark, Brown-headed Cowbird, Mourning Dove, Lark Sparrow, Vesper Sparrow, Green-tailed Towhee, Horned Lark, Sharp-tailed Grouse, Burrowing Owl, Ferruginous Hawk, and Prairie Falcon.

DISTRIBUTION: Two subspecies of Brewer's Sparrows are recognized, but only *Spizella breweri breweri* breeds in Utah; the more northerly subspecies--Timberline Sparrow (*S. b. taverneri*)--is restricted to high elevation sites in Canada and Alaska. The Brewer's Sparrow (*S. b. breweri*) is primarily a Great Basin species, but it occurs in Shrubsteppe habitats in all western states; it breeds throughout Utah and Nevada as well as Montana, Wyoming, western Colorado, northern New Mexico, northern Arizona, eastern California, eastern Oregon, eastern Washington, and southern Idaho. Its range also extends to southwestern Saskatchewan and southeastern Alberta (Rotenberry et al. 1999). In Utah, Brewer's Sparrows are common (Behle et al. 1985) to very common (Hayward et al. 1976) summer residents, breeding throughout the state in appropriate habitats. Densities in Utah are high in the northern and western parts of the state and highest in Rich and Summit counties (Sauer et al. 1997). Brewer's Sparrows winter in southeastern California, southern Arizona, and southern New Mexico, south into Baja and the central states of Mexico (Rotenberry et al. 1999); they occur rarely in Utah during the winter (Behle et al. 1985) most often in the southwestern corner of the state (Sauer et al. 1997).

ECOLOGY: Brewer's Sparrows are considered Neotropical migrants, though some populations may travel only a short distance between breeding and wintering ranges. Northern populations migrate farthest south (Rotenberry et al. 1999) and Utah Brewer's Sparrows probably winter in the Sonoran and Chihuahuan deserts of southern US and northern Mexico in mixed-species flocks with other sparrows. Brewer's Sparrows typically arrive in Utah in mid April and depart in mid October (Behle and Perry 1975).

Upon arrival, male Brewer's Sparrows establish territories (usually about 0.5 ha [1.2 ac]) which are vigorously defended both vocally and physically (Reynolds 1981). Females arrive a few days after males; and while no noticeable courtship behaviors are exhibited, pairs form a few days later. Territory establishment and pair formation might be delayed by cold weather (Peterson and Best 1985).

Nests, tight cups of grass and forbs lined with finer materials such as hair, are constructed in mid May. Nest construction is primarily by the female and takes about 5 days. Nests are typically placed between 20 and 50 cm (8-20 in) high and average around 40 cm (15.5 in); they are usually in the top ½ of the shrub (Peterson and Best 1985). Nests are usually located in patches of sagebrush that are taller and denser, with more bare ground and less herbaceous cover, than the surrounding habitat. In Idaho, nest shrubs averaged 69 cm (27 in) and height of the surrounding habitat was 43 cm (17 in) (Peterson and Best 1985). The vast majority of Brewer's Sparrow nests are in sagebrush; however, other shrubs are occasionally used (Rotenberry et al. 1999).

Clutch size is usually 3-4 eggs, occasionally 2 and rarely 5, and the clutch is laid at a rate of 1egg/day. Incubation begins when the second-to-last egg is laid and lasts for 10-12 days (Rotenberry and Wens 1991). The female performs most of the incubation duties, but the male frequently remains near the nest and occasionally incubates and delivers food to the female. Brewer's Sparrows will renest in a few days if the initial clutch is lost. Hatching begins in late May and peaks in the first 2-3 weeks in June (Howe et al. 1996). Hatching of altricial young takes place over a 1 to 2 days (up to 4 days) period and usually occurs in the mornings (Rotenberry et al. 1999). Both parents brood nestlings for 8 - 9 days. Adults feed nestlings almost exclusively insects (Petersen and Best 1986, Howe et al. in press) which are caught within 50 m of the nests (Rotenberry et al. 1999). Food items are delivered to the nestlings on average every 8 - 13 minutes (Howe et al. 2000) and feeding frequency increases with age of the nestlings (Rotenberry et al. 1999). Nestlings leave the nest at 6 - 9 days; early fledging (< 8 days) is often the result of nest disturbance. Nestlings remain in the nest area (< 10 m [< 33 ft] from the nest) for several days before they are able to fly. Parents attend the nestlings after they leave the nest for several days, though it is not known exactly how long parents remain with the fledglings. Late nests (late June - mid July) may represent renesting after failed attempts or double brooding. Brewer's Sparrows are frequent Brown-headed Cowbird hosts and often abandon parasitized nests (Rotenberry et al. 1999).

Daily nest survivorship over the entire nesting period is typically above 95% (Rotenberry and Wens 1989, Peterson and Best 1985, Howe et al. 1996). The proportion of successful nests and the number of young produced varies greatly from year to year. Howe et al. (1996) reported 68% successful nests and 1.75 young/nest attempt in one year and 32% success with 0.7 young/nest attempt the following year. Rotenberry and Wens (1989) found similar fluctuations which they attributed to predation (by ground squirrels) and precipitation in the winter preceding nesting.

Brewer's Sparrows are primarily insectivorous during the breeding season though their diet consists mostly of grass and weed seeds in winter. They glean insects from shrub foliage and bark and take seeds from the ground. Insect foraging occurs primarily (< 75%) in shrubs as opposed to on the ground between shrubs or in bunch grasses; foraging shrubs are usually sagebrush and are larger and more vigorous than those generally available in the surrounding habitat. Brewer's Sparrows diets change throughout the breeding season and differ between years (probably in relation to food availability). They feed on a wide variety of arthropods including spiders, leaf bugs, cicadas, snout beetles, caterpillars, crane flies, ants, and grasshoppers (Rotenberry et al. 1999). Nestlings are fed a similar diet with most of the diet made up of caterpillars, butterflies, spiders, beetle larvae, crane flies, cicadas, and grasshoppers (Petersen and Best 1986, Howe et al. in press).

Snakes and ground squirrels are the primary egg and nestling predators and these predators may regulate productivity in some years. The primary predators of adults are probably Loggerhead Shrikes; American Kestrel and Prairie Falcon predation has also been recorded. This predation is not likely to regulate adult populations (Rotenberry et al. 1999).

HABITAT REQUIREMENTS

Brewer's Sparrows breed primarily in Shrubsteppe habitats in Utah and are considered to be Shrubsteppe obligates by Braun et al. (1976). However, Brewer's Sparrows may also be found in High

Desert Scrub (greasewood) habitats, particularly where these habitats are adjacent to Shrubsteppe. They may also breed in large sagebrush openings in Pinyon-Juniper habitat or coniferous forests.

Breeding habitats are usually dominated by Big Sagebrush (*Artemisia tridentata*) and canopy height of breeding habitats is almost always < 1.5 m (5 ft) and usually 30 - 70 cm (1 - 2.3 ft). Nesting and foraging areas are usually in patches, sometime individual shrubs, that are taller than the surrounding vegetation. These areas also tend to have a greater percentage of live shrub growth, less bare or rock-covered ground, and greater canopy coverage than surrounding patches. See Ecology for detailed description of nesting and foraging site characteristics.

HABITAT AND/OR POPULATION OBJECTIVES:

1. Population trend should be stable or increasing (p = 0.10) over a 5-year period Population should be maintained at the current population level with an average relative abundance of 11 birds/BBS route on at least 50 routes statewide over 5 year period. This is based on the 30 year average as measured by Breeding Bird Survey relative abundance.
2. Maintain or increase suitable nesting habitat in Shrubsteppe and High Desert Scrub areas of the state.

ASSUMPTIONS:

1. The current Brewer's Sparrow population in Utah is a viable population.
2. Sagebrush habitats can be maintained or restored to provide suitable Brewer's Sparrow breeding habitats.
3. BBS relative abundance provides an adequate index of gross population trend for Brewer's Sparrows.

While Brewer's Sparrow populations are declining range wide at 3.7%/year (Sauer et al. 1997), the Brewer's Sparrow population in Utah appears to be stable and possibly increasing. And, given that Brewer's Sparrows are one of the most common species in Shrubsteppe habitats around the state, it is probable that the Utah population is viable at the current level. Because Utah's population is doing well while the overall population is declining sharply, it is important to maintain or increase our population, since Utah may act as a refugium or source for other Brewer's Sparrow populations in the West.

Edwards et al. (1995) classified Shrubsteppe in 38,438 km^2 (14,841 mi^2) of Utah, i.e., 17.5% of Utah's land cover is Shrubsteppe. Also, High Desert Shrub was present in 55,354 km^2 (21,372 mi^2) or 25.2% of the state's land cover. This does not address the condition of these habitats, but indicates that there is a great potential to provide suitable Brewer's Sparrow habitats in Utah.

Density estimates range widely depending on habitat and year, Wiens and Rotenberry (1981) found densities from 29-533 individuals/km^2. Annual variation on a single plot can run from 50-350 individuals/km^2, and one site was unoccupied in one year then had a density of 150 individuals/km^2 the following year. A reasonable estimate of healthy Brewer's Sparrow populations appears to be 150-200 individuals/km^2 averaged over several years. Relative abundance, a rough index of density, as measured by Breeding Bird Surveys is 10.97 birds/route (n = 52 routes) over the last 30 years in Utah.

Mean territory size also ranges widely from 0.10 ha to 2.36 ha (Rotenberry 1999). Average territory size of about 0.50 ha was reported by Reynolds (1981) in southeastern Idaho and this seems like a reasonable estimate for territory sizes in Utah. As with density, year and habitat influence territory size; also territory size might be density-dependent. Regardless of territory size, territories tend to be contiguous with adjacent territories (Rotenberry et al.1999).

IMPLEMENTATION RECOMMENDATIONS AND OPPORTUNITIES

MANAGEMENT ISSUES WITH CONSERVATION RECOMMENDATIONS: Brewer's Sparrows are not listed as a state or federal Threatened, Endangered, or Sensitive Species. Brewer's Sparrows are listed as a UPIF Priority Species because of range wide population declines and the importance of Utah to the overall species. Sagebrush habitats, i.e., Shrubsteppe and High Desert Scrub UPIF habitats, have undergone a broad scale conversion from native shrub/Grasslands to monotypic annual and perennial grasses. This conversion is due to sagebrush control efforts through chemical and mechanical (primarily chaining) methods often combined with introduction of aggressive, nonnative grasses such as crested wheatgrass (*Agropyron cristatum*). Also, introduction of the exotic annual cheatgrass (*Bromus tectorum*) has drastically altered the natural fire regime of many Shrubsteppe areas; increases in the frequency and intensive of rangeland fires strongly favor annual grasses and forbs to the detriment of many native grasses and perennial shrubs such as sagebrush. These conversions have also increased the fragmentation of Shrubsteppe habitats throughout the intermountain West (Braun et al. 1976). Rotenberry (1998) stated that loss and fragmentation of habitats may be the reason for range wide Brewer's Sparrow declines. Furthermore, his modeling predicts that more than half of the remaining shrub lands will be lost.

Most shrubs and perennial grasses have adapted to historical fires regimes where range fires were infrequent and of low to moderate intensity. Increasing frequency and intensity of range fires across the Great Basin is a significant threat to native grasses and shrubs which makeup Brewer's Sparrow habitat. Frequent and intense fires kill native plants and deplete or destroy native seed sources; natives are then replaced by exotic annuals, such as cheatgrass. Post-fire vegetation restorations have often resulted in introduction of aggressive and/or nonnative plants which dominate burned areas and inhibit reestablishment of native Shrubsteppe habitats.

Additional causes of habitat loss and fragmentation include agricultural conversion (to crop land), urban encroachment, roads, and rights-of-way (e.g., power lines, pipelines). These and other sources of fragmentation are likely to increase Brown-headed Cowbird nest parasitism of Brewer's Sparrows. Cowbirds are attracted to fragmented habitats.

Brown-headed Cowbird parasitism varies greatly by site (0-52% of nests parasitized) (Rotenberry et al. 1999). Nest parasitism is likely related to fragmentation, presence of cattle, and distance to cowbird feeding areas, e.g., stockyards. Brewer's Sparrows frequently abandon parasitized nests (Rotenberry et al. 1999), but cowbird young have been observed in nests where nestlings are near fledging age (Howe pers. obs.). A few observations of fledgling cowbirds with adult Brewer's Sparrows have been reported (Friedman et al. 1977).

The effects of Malathion, a pesticide used to control grasshoppers, on nesting Brewer's Sparrows appear to be minimal. No direct mortality of adults or young was observed and food base reductions appeared to be compensated for in most cases. However, nestling growth and food delivery intervals were impacted during one year of study suggesting that pesticide application in years of low food abundance may be detrimental (Howe et al. 1996, Howe et al. 2000). Little information is available for other pesticides, but the potential to affect Brewer's Sparrows through both direct and indirect (e.g., food reduction) pathways is of concern, since many insect control efforts focus on Shrubsteppe habitats and habitats directly adjacent to Shrubsteppe.

Brewer's Sparrow management issues (in italics) and recommendations are listed below:

Habitat Loss and Fragmentation

1. Establish a "no net loss" policy for Shrubsteppe (sagebrush and sagebrush plus grass) habitats.
2. Maintain or modify existing grazing regimes to promote growth of native shrubs and grasses. Temporarily remove grazing from degraded habitats and habitats recovering from fire or other detrimental factors.

3. Promote use of grazing to reduce cheatgrass dominance and prepare areas for native grass and shrub reseeding.
4. Promote reestablishment of native Shrubsteppe habitats through the use of prescribed fire and revegetation. Burns should be timed to promote growth of native grasses, minimize loss of sagebrush, and minimize establishment/regrowth of exotic annuals; revegetation should promote native grass and shrub reestablishment.
5. Promote use of prescribed burning and revegetation to avoid catastrophic wildfires.
6. Post-wildfire revegetation should focus on reestablishment of native grasses and shrubs, avoid use of nonnative and aggressive species and strive to exclude cheatgrass.
7. Use green-stripping, if necessary, to prevent stand-replacing fires in high quality Shrubsteppe patches.
8. Avoid road and right-of-way construction in large, contiguous patches of Shrubsteppe habitat. Construction footprints should be minimized and all rights-of-way should be revegetated with native grasses and shrubs.
9. Manage large blocks of land for contiguous Shrubsteppe habitat and avoid activities that cause fragmentation. Revegetate old roads and other disturbance corridors to native grasses and shrubs.
10. Avoid conversion of existing Shrubsteppe habitats to crop land, urban areas, etc. Maintain or reestablish native Grassland/shrub land open spaces in urbanized areas.
11. Monitor all revegetation efforts for success and enhance areas with poor native plant reestablishment.
12. Establish economic and reliable sources of native seeds for revegetation efforts and stockpile native seeds whenever possible.
13. Cowbird parasitism should be controlled through maintenance and reestablishment of large, contiguous blocks of Shrubsteppe habitat and reduction of habitat fragmentation.
14. Eliminate large-scale chaining and chemical control of Shrubsteppe habitats and eliminate large scale establishment of nonnative grasses in disturbed areas.
15. Use small-scale chemical and mechanical control methods to enhance Brewer's Sparrow habitats.

Pesticide Use

1. Avoid broad-scale use of pesticides during the nesting and brood-rearing season (mid-May through July).
2. If used, avoid persistent pesticides and those with high bioaccumulation potential.
3. Avoid use of pesticides in years of low food abundance.

Implementation Opportunities

1. Increase cooperation between state and federal agencies and private organizations regarding Shrubsteppe inventory, conservation, and management.
2. Increase awareness of public regarding Shrubsteppe habitat importance to birds.

EVALUATION OF ASSUMPTIONS: RESEARCH AND MONITORING

RECOMMENDED RESEARCH

1. Continue to annually monitor Brewer's Sparrow populations through the Breeding Bird Survey. Annually analyze 5-year trend to determine stability of population.
2. Determine effects of various habitat altering activities (e.g., grazing, oil exploration, wildfire, prescribed fire, chaining) on Brewer's Sparrow and their habitats and determine how such activities might be used to enhance Brewer's Sparrow habitats.
3. Determine best methods for controlling cheatgrass invasions and reestablishing native Shrubsteppe habitats (Rotenberry 1998). Include research use of prescribed fire and revegetation (e.g., best

mixture of native seeds, broadcast and drilling methods) after prescribed and wild fire.

4. Determine the effects of commonly used pesticides on Brewer's Sparrows and their prey.
5. Determine best methods of evaluating and monitoring quality and quantity of Shrubsteppe habitats at different scales through time (e.g., combination of ground-based and remote sensing change detection).

OUTREACH NEEDS

The recent focus on status of Sage-grouse has elevated awareness of the importance of Shrubsteppe habitats to birds. However, further outreach efforts need to focus on the importance of managing Shrubsteppe habitats for a diversity of bird species. Most Shrubsteppe habitats are under federal and state land management in Utah; thus land managers should be a primary target of outreach efforts. These efforts should include incorporation of bird diversity into Sage-grouse management plans and Conservation Agreements.

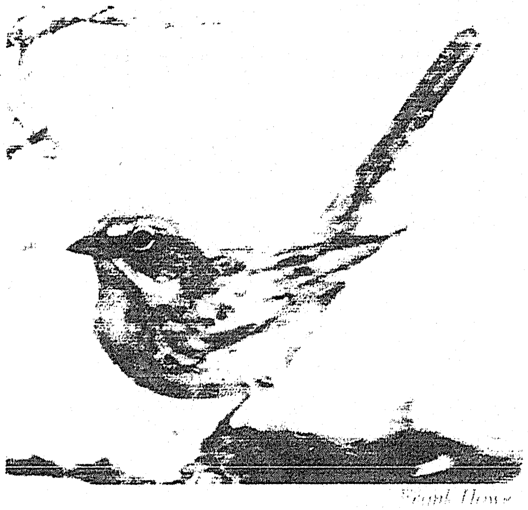
Public awareness of the importance and uniqueness of Shrubsteppe habitats is extremely limited. Preparation of a Shrubsteppe bird Wildlife Notebook suitable for classroom use would provide a vehicle for increased public awareness. This notebook would feature Priority Species (Sage-grouse, Sharp-tailed Grouse, Sage Sparrow, Brewer's Sparrow) and other associated Shrubsteppe birds.

2002

SAGE SPARROW

(Amphispiza belli nevadensis)

Conservation Priority Score of 32



ASSOCIATED SPECIES: Shrubsteppe-obligate species (Braun et al. 1976): routinely observed in similar habitats with Sage Thrasher, Black-throated Sparrow, and Brewer's Sparrow. Associated with species often occurring in shrub steppe but are not restricted to it; Northern Mockingbird, Northern Shrike, Green-tailed Towhee, and Vesper Sparrow. Other species that may benefit from specific Shrubsteppe conservation or management strategies are Ferruginous Hawk, Golden Eagle, Prairie Falcon, Burrowing Owl, Common Nighthawk, and Ash-throated Flycatcher.

DISTRIBUTION: (interior-Great Basin environs, including western Washington to Wyoming south to Arizona and Texas, eastern California, Utah, Nevada, i.e., bulk of North American distribution). Other subspecies include: *A. b. belli* (coastal California); *A. b. clementaea* (San Clemente Island, California); *A. b. cinerea* (Baja California); *A. b. canescens* (interior California). Oberholser (1946) described *A. b. campicola* for southern and central Idaho, eastern Washington, eastern Oregon, Nevada, and western Utah, but this subspecies remains unrecognized (A.O.U. 1957, 1983, Behle 1985).

Breeding Range: western North America, including; west-central Washington, Oregon east of the Cascade Mountains, southeastern. and extreme southwestern. Idaho, portions of central, western and northeastern (locally) Wyoming, all but southern Nevada, mesas of western Colorado and in the San Louis Valley in south-central Colorado, northwestern New Mexico, and northeastern Arizona. In California breeds in extreme northeast south to Sierra County, Mono County south around the western rim of the Mojave Desert to the Upper Kern River Basin, foothills of the western Sierra Nevada from El Dorado County south to Mariposa County, San Clemente Island, western California along inner Coast Range from Trinity and Shasta Counties, south to coastal Marin County, south through San Joaquin Valley to southern California west of the eastern deserts, and south through Baja California (except east coast) to about 26°N. Rare during the breeding season in western Washington, western Oregon, Montana and British Columbia. Found locally throughout Utah (Rising and Beadle 1996, Martin and Carlson 1998).

Winter Range: Found throughout year in southern California and Baja areas where overlaps with breeding range except, in northernmost portion of inner Coast Range and in east-central California where species does not typically winter. Otherwise winters locally from southern Nevada, southwestern Utah, all but northeastern Arizona, west-central and southeastern New Mexico; east to western Texas; south to central Chihuahua, northwest Sonora to Kino Bay, southeastern California, and eastern Baja California Norte (Martin and Carlson 1998, Russell and Monson 1998). Also winters near Pyramid Lake and Fallon in western Nevada. Rare in winter north to southern Oregon.

Utah: Uncommon permanent resident statewide to 2400 m (8000 ft) elevation; more common in migration; common winter resident in southern Utah (Behle et al. 1985). Migratory populations (summer breeders) occur from late 2 March - 30 September (Woodbury et al. 1949).

ECOLOGY:

Migration—Considered a short-distance migrant; 3 non-migratory subspecies (*A. b. belli, A. b. clementaea, A. b. cinerea*); 2 migratory subspecies (*A. b. canescens and A. b. nevadensis*). Some populations observed moving up slope after breeding early in spring. In late summer and fall they descend from the mountain valleys moving south and east toward wintering grounds (Johnson and Marten 1992).

Fall migrants of *A. b. nevadensis* peak mid-Sep in Oregon with stragglers to mid-Nov; depart western Colorado by mid-October with stragglers to early November but depart San Luis Valley, Colorado, by early October. Some *A. b. nevadensis* individuals may be resident but displacement by more northern individuals could account for year-round occurrence (Weathers 1983).

Both *A. b. nevadensis* and *A. b. canescens* on wintering grounds in September and remain until February or early March in Arizona along Colorado River Valley (Meents et al. 1982). In southern California *A. b. canescens* withdraw from higher elevations into deserts during winter. Based on specimen evidence, *A. b. nevadensis* wintering in this area arrive later and depart earlier (early October to mid March) than *A. b. canescens*. In New Mexico, first observed late September at Ojo Caliente, 1 October further south at Lake Burford and late October at Silver City. Wintering birds recorded in northern Mexico by October (Howell and Webb 1995).

Migratory and wintering—*A. b. nevadensis* observed in small flocks of 3 to 10 individuals, frequently in sagebrush (*Artemisia* spp.) or foraging between widely spaced creosote bush (*Larrea tridentata*) in desert scrub (Weathers 1983). Often flocks with other species; Black-throated, White-crowned, Vesper and/or Brewer's sparrows (Cody 1971).

Breeding—Some populations move in pairs all year, but the pairs are not always the same individuals throughout year or from one year to the next. In southern latitudes, males singing on territory as early as late-January or early-February. Some *A. b. nevadensis* arrive on the breeding grounds already paired which is unusual for most migratory songbirds (Rich 1980a). No information on actual pairing time; males are singing on territory by late March.

A. b. nevadensis recorded as arriving on breeding grounds on the Eastern Sierra escarpment around Mono Lake, Sagehen Summit, and Cedar Hill, California, by mid-March; early May, with some as early as mid-March; in western Colorado and late April in San Luis Valley, Colorado; mid-March in Idaho (Rich 1980a); early March in northern Nevada (Alcorn 1988); late February in Oregon peaking in mid-March; and early March in Utah (Green 1981). *A. b. nevadensis* latest arrival observed early April near the Mexican border and late April in Organ Mountains, New Mexico. Nesting activities observed in mid-March for *A. b. nevadensis* in Benton County, Washington and Morrow County, Oregon. Few nests are built after mid-June.

Nesting Habitat—Nests mainly in shrubs, but also in bunch grass (Poaceae) and occasionally on the ground under a shrub. Microhabitat preference is probably based more on structure and density of shrubs rather than specific shrub species. Winter and Best (1985) found that shrub height and nest microclimate may influence location of nests. Nests built in grasses on ground under small shrubs, provided better cover than in the shrubs themselves. Ground nests provided more warmth, beneficial early in breeding season only. Sage Sparrows prefer taller shrubs with larger canopies providing more cover (Petersen and Best 1985). Nests placed in the canopies would benefit from increased air circulation and convective heat loss (Winter and Best 1985). Populations of Sage Sparrow subspecies that use fewer shrub species nest in habitats with less shrub diversity, i.e., cold northern desert (Great Basin) v.s. hot desert (Mojave)(Wiens 1985).

Shrubs used depend on geographical location. *A. b. nevadensis* uses big sagebrush (*A. tridentata*) in Idaho (Petersen and Best 1985). Additionally, bitterbrush (*Purshia tridentata*), rabbitbrush (*Chrysothamnus* spp.), greasewood (*Sarcobatus vermiculatus*), tumbleweed (*Salsola iberica*), or bunch grasses are used in Oregon and Nevada (Wiens and Rotenberry 1981, Wiens 1985) and Bonneville Basin, UT (Martin and Carlson 1998). One study in the Uinta Basin, UT, observed 11 nests over 2 yr: above-ground shrub nests included 3 in rabbitbrush, 2 in hopsage (*Grayis spinosa*), 1 in saltbush (*Atriplex* spp.), and 1 in big sagebrush; the remaining ground-nests were in depressions beneath rabbitbrush (Green 1981).

Populations in other locations may use brittlebush (*Encelia farinosa*), black sage (*Salvia mellifera*), California buckwheat (*Eriogonum fasciculatum*), California sagebrush (*Artemisia californica*), bush mallow (*Malocothamnus fasciculatum*). Chamise (*Adenostoma fasciculatum*), white sage (*Salvia apiana*), valley cholla (*Opuntia parryi*), ceanothus (*Ceanothus* spp.), or willow (*Salix* spp.)(Martin and Carlson 1998).

Nest shrub is generally higher than average height of surrounding vegetation (Green 1981, Petersen and Best 1985). Where sagebrush coverage is sparse, nest sites are selected where shrubs clumped. The birds also prefer shrubs with at least 75% live material. Nests are typically placed away from the southwest side of the shrubs, perhaps to avoid strong southwesterly winds or afternoon solar radiation. Most nests are placed in densest portion of nest site vegetation profile (Rich 1980b, Petersen and Best 1985). Nests are built nearer main stem than the edge of shrub (Reynolds 1981).

In the Uinta Basin, UT, shrub nests were 26 cm, ranging from 24-27 cm (N=7), above ground and primarily in the northeast quadrant of the shrub (Green 1981).

Nest—Nest is open cup. Outer shell of small twigs or coarse grasses, lined with fine grasses and weed bark and completed with inner lining of softer material such as feathers, tufts of wool, rabbit fur, and cow hair (Martin and Carlson 1998), although in Utah only dried grasses and forbs were used (Green 1981). In Utah, nests were constructed in 1-2 da entirely by female (Green 1981).

Clutch Size, Incubation, and Fledging—Most *A. b. nevadensis* attempt 2 and occasionally 3 clutches each year. New nests are built for subsequent clutches.

Mean clutch size for *A. b. nevadensis* is 3.28 eggs (SD ± 0.61, range 1-4, n = 61; unpublished data, Western Foundation of Vertebrate Zoology).

Egg dates for *A. b. nevadensis* from 6 April-16 June. Incubation about 12-16 days from completion of clutch. In one instance, the first egg was laid 8 days after nest was complete (Martin and Carlson 1998). Females of all subspecies lay usually one per day until clutch complete. Although adults remain in general proximity they are inattentive to nest. No information on replacement of individual eggs. When a clutch is depredated, the nest is abandoned and another built nearby.

First chick dates for *A. b. nevadensis* 6 April. Young typically fledge in about 9-11 days. By day 2 young can lift heads with mouths open wide for food This response occurs when shrub moved slightly and occurs only during first few days. Eyes open at 4-5 days. After eyes open, young do not respond to shrub movements. Lifts heads only to feed when adult at nest itself. When shrub disturbed they huddle down into nest. Begging calls begin about 6-7 days. Young are usually in nest for 9-10 days (Petersen and Best 1986).

Adult feeding of young begins shortly after hatching and continues for duration of nestling period and for 2+ wk after nest departure. Both parents feed young. Nestling food items are small with 73% < 0.05 cm (Petersen and Best 1986). Nestlings fed a broad range of arthropods with 59% of diet consisting of spiders, butterflies and moths (Lepidoptera), true bugs (Hemiptera), and leafhoppers (Homoptera). Flies (Diptera), grasshoppers (Orthoptera), bees and wasps (Hymenoptera), beetles (Coleoptera), and psocids

(Psocoptera) also fed to nestlings (Petersen and Best 1986). As nestlings grow, they are fed fewer spiders, a pattern found in Brewer's and Vesper sparrows (Best 1972). Diet composition does not differ significantly among broods of 2, 3, and 4. Nestlings receive larger food items as they grow in all brood sizes, especially with broods of 4 (Petersen and Best 1986). In a prescribed mosaic burn, vegetation altered by fire did not significantly affect composition of nestling diet or mean size of food items (Petersen and Best 1986).

Nesting Success—In *A. b. nevadensis*, mean number of young/nest is 2.6 (SD ± 0.7, range 1-3, n = 11 (Reynolds 1981). The mean number of young fledged/successful nest in *A. b. nevadensis* is 1.3 (SD ± 1.3, range 0-3, n = 15) (Reynolds 1981). Reynolds observed high fledgling success during the first year of study but Loggerhead Shrikes depredated nearly all nests the second year.

Breeding Parameters—Assumed that both sexes attempt to breed at 1 year. Normally breeds each year.

Diet and foraging—Sage Sparrows are categorized as a ground-foraging omnivore during the breeding season, and a ground-gleaning granivore during the nonbreeding period (DeGraaf et al. 1985, Polis 1991). Foods taken during breeding season include adult and larval insects, spiders, seeds, small fruits, and succulent vegetation. Fall, winter, and early spring foods include small seeds, plant material, and insects when available.

Primarily forages opportunistically on the ground picking up seeds and miscellaneous prey, usually near or under edges of shrubs within sage scrub or chaparral, or gleaning prey from lower main stems of shrubs, occasionally from leaves. Gleans arthropods from low annuals and lower stems and leaves of shrubs. Opportunistically exploits temporarily abundant foods (Wiens and Rotenberry 1979).

Diet varies depending on the season. Seeds including grasses (Poaceae), pigweeds (Chenopodiaceae), and mustards (Brassicaceae) are heavily ingested in April, then again in July and August (Rotenberry 1980). At the Arid Lands Ecology Reserve, Benton County, WA, breeding adults in sagebrush-bunch grass (*Agropyron* spp.) take a wide variety of arthropods in May and June, including coleopterans (Curculionidae, Tenebrionidae, Scarabaeidae, and Carabidae); hymenopterans, lepidopteran larvae, orthopterans (Wiens and Rotenberry 1979, Rotenberry 1980). Larger prey items were consumed early in the season and gradually declined in size, April-August (Rotenberry 1980).

Diet shifted from 44% animal and 56% seed and plant in fall to 13% animal and 87% seed and plant material in wintering A. b. nevadensis and A. b. canescens along the lower Colorado River. Fall insects came from beetles (Coleoptera), true bugs (Hemiptera), leafhoppers (Homoptera), grasshoppers (Orthoptera), and ants (Formicidae). Winter insects in the diet was similar to Fall except it did not include grasshoppers. The only identifiable seeds were from the pigweed family (Meents et al. 1982).

Predation—Both young and eggs are removed from nests by Common Ravens typically taking calling nestlings when adults are away from nests. Unsuccessful and successful attacks on adult Sage Sparrows by Loggerhead Shrikes have been observed (Reynolds 1979). An adult was chased by a Merlin (Martin and Carlson 1998). Bones were found in a Great Horned Owl pellet in Lincoln County, Nevada. Adults chip loudly when Greater Roadrunner near nest. When approached while on nest at beginning of incubation period, female usually quietly slips off and runs on ground.

Lifespan and Survivorship—Few data on survivorship. A 6-year old A. b. nevadensis was found in one study by Wiens (1985). For A. b. belli, a 7-year and 2.5-year old color-banded males were recorded. Life spans of 3 year is not uncommon for males and 2 year for females (Martin and Carlson 1998).

HABITAT REQUIREMENTS: Considered obligate shrub steppe species by Braun et al. (1976), Rotenberry and Wiens (1980), Reynolds (1981), and Wiens and Rotenberry (1981).

Breeding Sage Sparrows prefer semi-open habitats with evenly-spaced shrubs 1-2 m high. Vertical structure, habitat patchiness, and vegetation density may be more important in habitat selection than specific shrub species, however; Sage Sparrows are closely associated with big sagebrush throughout most of their distribution (Rich 1978, Rotenberry and Wiens 1978, Wiens and Rotenberry 1981, Smith et al. 1984, Wiens 1985). Sage Sparrows are often missing from what appears to be suitable habitat, so other unknown habitat characteristics may be important (Rich 1978).

A. b. nevadensis prefers big sagebrush whether pure stands or interspersed with bitterbrush, saltbush, shadscale (*Atriplex confertifolia*), rabbitbrush, or greasewood. Rarely in mixed sagebrush-juniper (*Juniperus* spp.), except in ecotones adjacent to shrub steppe habitat. Usually breeds below 1700 m but has been found over 2400 m (Rich 1980a, Wiens and Rotenberry 1981, Petersen and Best 1985). Found breeding in creosote bush during wet years but not during dry years (Hill 1980).

In the northern portion of wintering range *A. b. nevadensis* favors big sagebrush. Further south, fairly common to uncommon during winter in desert washes, big sagebrush, creosote bush (*Larrea tridentata*), sparse cactus scrub, arid Grasslands and arboreal yucca (*Yucca* spp.) mixed with greasewood (Russell and Monson 1998). Along the Colorado River, honey mesquite (*Prosopis glandulosa*) with high densities of inkweed (*Suaeda torreqana*) had higher densities of Sage Sparrows than honey mesquite without inkweed (Meents et al. 1982).

HABITAT AND/OR POPULATION OBJECTIVES:

To the knowledge of the authors, no habitat or population parameters have ever been established for the Sage Sparrow as a management objective. However, the following density and territory data may assist in developing local management strategies.

Density—Range-wide estimates in optimal habitats, 57-145 individuals/km^2 (Rotenberry and Wiens 1980, Wiens and Rotenberry 1981).

Territoriality—Size variable, some of the largest for any sparrow species; at higher densities, territories at minimum size, but at lower densities territory sizes are at maximum (Martin and Carlson 1998).

Territories for *A. b. nevadensis* varied in size and shape: Idaho, 4.43 ha (1.86:8) 1.06-7.06 ha (Rich 1980b); 0.81 ha (21:16)(Reynolds 1981); Nevada and Oregon, 0.65-5.81 ha (Wiens and Rotenberry 1985); and Utah, 1.53 ha (0.23:7) 1.21-1.79 ha (Green 1981). Territories with grass and sagebrush are smaller than those in heterogeneous vegetation dominated by spinescent shrubs. Territory size did not increase significantly over the short term in areas where vegetation was removed (Wiens et al. 1986). Territory fidelity or tenacity may be a factor for the first one or two years after vegetation removal. In addition, if insular islands of vegetation are left within territories Sage Sparrows may adjust to removal without significant density changes. Where Sage Sparrows are less abundant, they appear to increase territory size with no apparent upper limit except they do not completely saturate the available habitat (Wiens and Rotenberry 1985, Wiens et al. 1985). At this point the Sage Sparrow may be selecting "optimal habitat." Habitat characteristics differed little between territories shifted from one year to the next, but subsequent territories were on the average significantly larger than previous ones (Petersen and Best 1987a, 1987b). Furthermore, larger territories were positively correlated with a greater number of fledglings.

Males occupy breeding territories that are essentially non-overlapping. However, territory boundaries can change slightly from day to day. Territory shifting seems adaptive to increasing territory size (Petersen and Best 1987b). Male establishes territory in spring by singing from perches with occasional conflicts. Fighting, visual display, or chasing occurs in different frequencies in different populations (Wiens 1982, Rich 1980a). Intensity of territorial defense may be affected by the differences in population densities, with more defense occurring at higher densities.

IMPLEMENTATION RECOMMENDATIONS AND OPPORTUNITIES

MANAGEMENT ISSUES WITH CONSERVATION RECOMMENDATIONS:

Habitat Degradation—Due to the Sage Sparrow's close association with sagesteppe habitats (Great Basin specifically but including other shrub-dominated ecosystems) throughout its range (Wiens 1985) and the broad scale efforts to control big sagebrush via chemical and mechanical methods, conversion of native mixed shrub-grass communities to exotic annuals largely through increasing intensity and frequency of rangeland fires, livestock grazing disturbances, and natural shrub die-off, overall sparrow distribution has most likely been altered from pristine conditions (Young et al. 1975, Braun et al. 1976, Nelson et al. 1989).

Alteration to native vegetation by removal of shrubs, brought no immediate change in Sage Sparrow densities in the first year after removal, possibly due to lag time related to site fidelity or tenacity (a common problem in short-term avian studies). During the second and third years following sagebrush treatment, significant declines were observed (Wiens 1985, Wiens and Rotenberry 1985, Rogers et al. 1988).

Compounding any consideration of direct habitat impacts is fragmentation (urbanization, agricultural conversion, road and power line right-of-ways, etc.) with typically associated increases in brown-headed cowbird brood parasitism. Associated with increasing human densities increases likelihood of secondary predation by feral cats. The introduction of pigs, goats, and other grazing animals on San Clemente Island, CA, has significantly impacted habitat, which in turn has greatly reduced local Sage Sparrow numbers (Everatt et al. 1994). Only with removal of exotic animals and the subsequent recovery of native habitats have A. b. clementaea numbers stabilized and increased. However, fragmentation, brood parasitism, and predation are only qualitatively recognized and further study is necessary to quantify these threats and develop management actions.

Fire Management and Exotic Weed Invasion—Long-term fire suppression in some locations alters the pattern of natural plant succession, allowing communities to grow into dense stands, which probably reduce available Sage Sparrow habitat for breeding.

Increasing frequency and intensity of range fires in the Great Basin poses a significant threat to native grasses and shrubs. Historically fires were infrequent and perennial grasses and shrubs were not adversely affected. With increased fire frequency, native plants are killed and seed-reservoirs of grasses and shrubs are depleted and replaced with exotic annuals, such as cheatgrass (*Bromus tectorum*). Sage Sparrows abandon former habitats once invaded by cheatgrass (Wiens 1985, Rogers et al. 1988).

Brown-headed Cowbird Parasitism—Nests have been parasitized by Brown-headed Cowbirds in Idaho (Rich 1978, Reynolds 1981). Some parasitized nests were abandoned by Sage Sparrows (Reynolds 1981, Friedmann and Kiff 1985). Sage Sparrows have fledged or attempted to raise cowbirds (Gaines 1988). Most records of parasitism are correlated with human disturbance such as removal of big sagebrush for crops or increase Grassland for livestock grazing. Cowbirds rarely intrude into large tracts of big sagebrush.

Management Protection (Administrative) Actions—*A. b. clementeae* was listed as "threatened" by USFWS in 1977. A recovery plan developed by the USFWS in 1984 recommended protection, enhancement, and restoration of plant populations by removal of feral animals and prevention of their reintroduction, replanting native plants, and controlling erosion where necessary (Everatt et al. 1994). *A. b. belli* is listed as Species of Special Concern in California, which means that the species may be declining but supporting biological evidence has not been published.

Management Actions—Conservation management strategies for the Sage Sparrow are just being considered and developed by the Western Working Group of Partners in Flight. In the interim, general recommendations are provided by Braun et al. (1976).

Implementation Opportunities

One of the major floristic characteristics of the Great Basin is the large expanses of seemingly continuous biotic communities, whether Shrubsteppe (greasewood, shadscale, or sagebrush) or pygmy forests (Pinyon-Juniper or juniper). The loss of these habitats is no less important than pristine Wetlands, riparian corridors, or high-elevation forests. Land and resource managers should endeavor to conserve the ecological processes and functions, including large manageable tracts of shrub habitats essential to the Sage Sparrow and a variety of similar species which it represents.

EVALUATION OF ASSUMPTIONS: RESEARCH AND MONITORING

RECOMMENDED RESEARCH

A cursory review of the literature gives the impression that Sage Sparrows have been studied intensively. Many of the direct observations are anecdotal or derived from community-based studies where Sage Sparrows were component members of a larger investigation. Even in studies where the Sage Sparrow was a focal species, more questions were raised than resolved (Wiens 1985). Because the Sage Sparrow is closely associated with big sagebrush and other similar shrub habitats (a habitat generally considered poorly studied within North American ornithology), its presence or absence, density and distribution, and productivity could be used as an indicator of habitat quality. Shrub vegetation is difficult to measure from year-to-year due to subtle changes that are not as readily detectable as in grasses or forbs.

Research should focus on:

1. Species-specific life history and ecology in long-term contexts (i.e., longer than 1-2 years, due to lag time related to site fidelity or tenacity), inclusive of migrating, breeding, and wintering periods.
2. Improved methods for measuring shrub vegetation parameters useful to avian studies.
3. Options for management of Shrubsteppe bird species for long-term conservation, which includes determination of the significance of human-related and natural changes to Sage Sparrow population parameters and their habitats.

OUTREACH NEEDS

Increased awareness through public education is critical to promoting public support for the Utah Partners in Flight habitat and species conservation objectives. Much of the public at large is unaware that there is even an issue regarding avian conservation. At a minimum, it is suggested for the Sage Sparrow (and all the Utah conservation priority species), information fact sheets (front and back 8.5x11 in card stock) be prepared similar to those already prepared for some Utah species (i.e., Bald Eagle, Utah Prairie Dog, etc.). A picture of the species, distribution map, and general life history information should be included sufficient for classroom use.

The recent focus on status of Sage-grouse has elevated awareness of the importance of Shrubsteppe habitats to birds. However, further outreach efforts need to focus on the importance of managing Shrubsteppe habitats for a diversity of bird species. Most Shrubsteppe habitats are under federal and state land management in Utah; thus land managers should be a primary target of outreach efforts. These efforts should include incorporation of bird diversity into Sage-grouse management plans and Conservation Agreements.

Public awareness of the importance and uniqueness of Shrubsteppe habitats is extremely limited. Preparation of a Shrubsteppe bird Wildlife Notebook suitable for classroom use would provide a vehicle for increased public awareness. This notebook would feature Priority Species (Sage-grouse, Sage Sparrow, Brewer's Sparrow) and other associated Shrubsteppe birds.

The Condor 98:75–81

NESTING HABITS OF THE LOGGERHEAD SHRIKE IN SAGEBRUSH[1]

CHRISTOPHER P. WOODS[2]
Department of Biology, Boise State University, 1910 University Drive, Boise, ID 83725

TOM J. CADE
The Peregrine Fund, Inc., 5666 W. Flying Hawk Lane, Boise, ID 83709

Abstract. The Loggerhead Shrike (*Lanius ludovicianus*) is a widely distributed member of Idaho's sagebrush (*Artemesia tridentata*) rangeland avifauna. During 1991–1993, we studied the breeding ecology and nesting locations of the species in this semi-arid habitat. Most loggerhead nests (65%) were constructed in sagebrush, although bitterbrush (*Purshia tridentata*) and greasewood (*Sarcobatus vermiculatus*) were also used frequently. Height of nest shrubs averaged 162 cm (range: 89–297 cm), and the mean height of nests was 79 cm (range: 33–160 cm). Nest variables we measured did not differentiate successful from unsuccessful nests. Significant differences, however, distinguished the three primary nest shrubs: sagebrush shrubs used for nesting tended to be smaller than bitterbrush or greasewood shrubs. Nonetheless, nest height was nearly identical regardless of shrub species, and nest success was independent of nest shrub. The low nesting heights for the loggerhead in this sagebrush-scrub habitat represent a notable departure from nesting heights in many parts of the species range, although they may be typical for the species where it occurs in the southwest. The preservation of Idaho's sagebrush rangelands will be important to the long-term survival of this species in the state.

Key words: Lanius; Artemesia; *nest success; nest height; shrub-steppe.*

INTRODUCTION

Shrikes (*Lanius* spp.) occupy a distinctive position in avian communities: they are passerines that prey upon reptiles, mammals, and other birds, as well as a wide array of invertebrates. The Loggerhead Shrike (*L. ludovicianus*), the smaller and more widespread of the two North American species, was relatively common across most of the continent early in this century (Miller 1931). In recent years, interest in the loggerhead shrike has increased as dramatic population declines have been noted in many geographic regions (Cade and Woods, unpubl.). Studies in the United States have focused on resident or migratory Loggerhead Shrike populations in the East (Siegel 1980, Milburn 1981, Luukkonen 1987, Gawlik 1988, Novak 1989), Midwest (Graber et al. 1974, Kridelbaugh 1982, Brooks 1988, Haas 1990), prairie states (Porter et al. 1975, Tyler 1992), and California (Craig 1978, Scott and Morrison 1990).

Fraser and Luukkonen (1986) speculated that, prior to European settlers, the Loggerhead Shrike may have been rare in the eastern United States and occurred in its highest numbers in brushlands and deserts of the southwest, habitats which are also found in Idaho, Oregon, and Washington. Supporting that contention, Jewett et al. (1953) and Gabrielson and Jewett (1940) found the loggerhead to be common in sagebrush habitat in eastern Oregon and Washington, respectively, early in the 1900s. Furthermore, early accounts of Idaho's avifauna indicate shrikes may also have been common here in the late 1800s and 1900s (Stearns 1930, Burleigh 1972). Southwestern loggerhead populations, however, have been largely unstudied, and the nesting habits of these shrikes are relatively unknown.

We studied ecological characteristics of Loggerhead Shrikes (*L. l. gambeli*) breeding in southwest Idaho's sagebrush (*Artemesia tridentata*) rangelands, and we report here on nesting habitats and nesting sites of the loggerhead in that semi-arid habitat during the 1991–1993 breeding seasons.

METHODS

This study was conducted primarily on three study sites on the Snake River Plain and along

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[1] Received 13 April 1995. Accepted 5 September 1995.

[2] Present address: Department of Biology, University of Regina, Regina, Saskatchewan, S4S OA2, Canada.

TABLE 1. Characteristics of nests and nest shrubs used by Loggerhead Shrikes in southwest Idaho from 1991 through 1993. All measurements are in cm, and expressed as mean ± SD.

	n	Shrub height	Shrub width	Nest height	Nest to top	Nest to edge	% Nest success
Shrub[a]							
Sagebrush	106	151 ± 36^{1}	202 ± 52^{1}	80 ± 24^{1}	52 ± 20^{1}	38 ± 14^{1}	59%
Greasewood	20	173 ± 30^{1,2}	268 ± 49^{2}	78 ± 18^{1}	65 ± 13^{2}	63 ± 20^{2}	53%
Bitterbrush	33	190 ± 48^{2}	283 ± 76^{2}	79 ± 29^{1}	68 ± 24^{2}	61 ± 26^{2}	59%
Total[b]	162	162 ± 41	231 ± 71	79 ± 24	57 ± 21	47 ± 22	59%
Outcome[c]							
Successful	89	166 ± 45	226 ± 76	82 ± 27	59 ± 23	47 ± 20	
Failed	63	157 ± 35	237 ± 68	76 ± 20	57 ± 20	49 ± 24	
Renest Difference[d]	26	−5 ± 48	14 ± 110	−16 ± 29*	4 ± 28	9 ± 27	

[a] Nest shrubs with significantly different means (Tukey's test; $P < 0.05$) are indicated by different superscripts.

[b] Includes two nests in four-wing saltbush and one in Russian thistle.

[c] MANOVA did not indicate a significant difference in any category ($P > 0.1$).

[d] All measurements presented as mean of first minus second nest measurement. Asterisk indicates measurement with a significant difference ($P < 0.05$) between first and second nests.

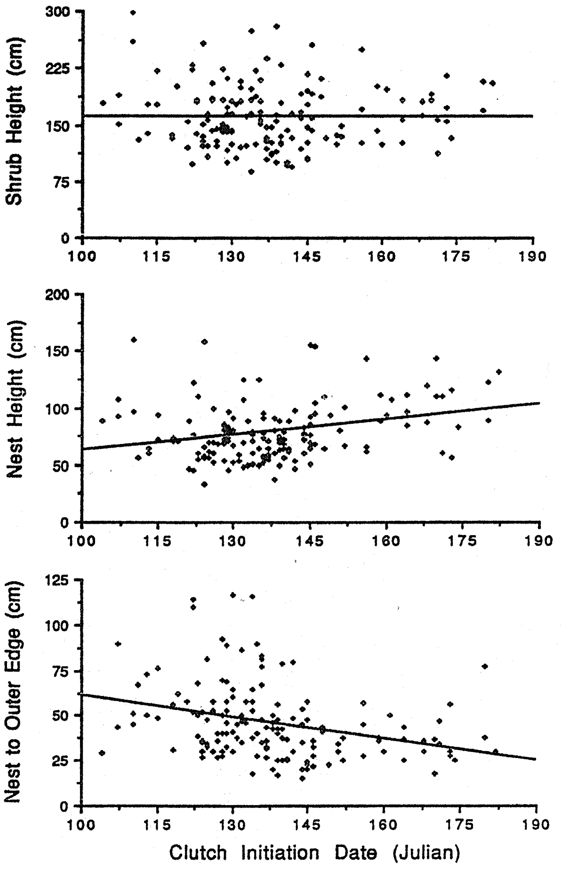

FIGURE 1. Relation between particular nest site variables and clutch initiation date in Loggerhead Shrike nests in southwest Idaho in 1991 through 1993. Date of clutch initiation was corrected by 11 days in 1992 and 1 day in 1993 so median lay dates were equal in all years.

species, and nest success, which varied from 53% to 59% among shrub species, was independent of nest shrub ($\chi^2_2 = 0.15$, $P > 0.1$). The size of the nest shrub did not vary with nesting date (shrub height: $r = -0.001$, $P > 0.1$; shrub width: $r = -0.143$, $P = 0.081$; Fig. 1). In contrast, nest height was positively correlated with clutch initiation date ($r = 0.289$, $P < 0.001$), while the distance from the nest to the top of the shrub above the nest, and the distance from the nest to the shrub's nearest outer edge, decreased as the breeding season progressed ($r = -0.187$, $P = 0.020$, and $r = -0.299$, $P < 0.001$, respectively; Fig. 1).

There were no significant differences in nest shrub height or width, distance to the top of the shrub, or minimum distance from nest to outer perimeter of shrub between first and second nest attempts ($P > 0.1$ in all cases; Table 1). Second nests were, however, significantly higher than first ones ($t_{25} = 2.78$, $P = 0.010$). Nearly all renests were found following initial failure; we found a second nesting attempt following successful breeding on only two occasions, once in 1991 and again in 1993. Shrikes constructed second nests, and initiated second clutches, more quickly than initial nest attempts. The period from initial nest construction to egg-laying in early nests was 13.1 ± 3.4 days (7; 9–18). In contrast, the mean time from first nest failure to second clutch initiation was only 8.8 ± 3.5 days (20; 3–18). Note that this time includes the selection of an alternative nest site, as well as construction ulate that the loggerhead bred in the state prior to man's alteration of the plain, and that a contraction in both range and numbers has occurred as appropriate habitat has been lost. Furthermore, as Idaho's human population continues to grow, additional loss of sagebrush is likely. Habitat loss has been correlated with shrike declines elsewhere (Lymn and Temple 1991, Gawlik and Bildstein 1993, Prescott and Collister 1993), and an emphasis should be placed on preservation of Idaho's existing sagebrush-scrub desert to help maintain this shrike population in the future, especially as these shrikes appear to have a limited ability to occupy human-modified landscapes.

ACKNOWLEDGMENTS

The field assistance of Colleen Sweeney, Beth Telford, Bev Witte, and Beth Woods were invaluable to this study. The study benefitted from the cooperation of Mike Black, who allowed us free access to his ranch, where some shrikes nested. We are indebted to Al Dufty, Jim Munger, John Rotenberry, and Rick Williams, all of whose advice and support improved the study considerably. This manuscript further benefitted from comments by Reuven Yosef and one anonymous reviewer. This study was funded in part through the support of the Raptor Research Center at Boise State University and a Grant-in-Aid of Research from Sigma Xi, the Scientific Research Society. This paper is dedicated to Frank Pitelka on the occasion of his 80th birthday.

LITERATURE CITED

AUSTIN, G. T. 1974. Nesting success of the Cactus Wren in relation to nest orientation. Condor 76: 216–217.

AUSTIN, G. T. 1976. Behavioral adaptations of the Verdin to the desert. Auk 93:245–262.

BRAUN, C. E., M. F. BAKER, R. L. ENG, J. S. GASHWILER, AND M. H. SCHROEDER. 1976. Conservation committee report on effects of alteration of sagebrush communities on the associated avifauna. Wilson Bull. 88:165–171.

BROOKS, B. L. 1988. The breeding distribution, population dynamics, and habitat availability and suitability of an upper midwest Loggerhead Shrike population. M.Sc.thesis, Univ. of Wisconsin, Madison, WI.

BURLEIGH, T. D. 1972. Birds of Idaho. Caxton Printers, Caldwell, ID.

CRAIG, R. B. 1978. An analysis of the predatory behavior of the Loggerhead Shrike. Auk 95:221–234.

FRASER, J. D., AND D. R. LUUKKONEN. 1986. The Loggerhead Shrike, p. 933–941. *In* Audubon Wildlife Report 1986. National Audubon Society, New York.

GABRIELSON, I. N., AND S. G. JEWETT. 1940. Birds of Oregon. Oregon State Monographs, Studies in Zoology, No. 2. Oregon State College, Corvallis, OR.

GAWLIK, D. E. 1988. Reproductive success and nesting habitat of Loggerhead Shrikes, and relative abundance, habitat use, and perch use of Loggerhead Shrikes and American Kestrels in South Carolina. M.Sc.thesis, Winthrop College, Rock Hill, SC.

GAWLIK, D. E., AND K. L. BILDSTEIN. 1990. Reproductive success and nesting habitat of Loggerhead Shrikes in north-central South Carolina. Wilson Bull. 102:37–48.

GAWLIK, D. E., AND K. L. BILDSTEIN. 1993. Seasonal habitat use and abundance of Loggerhead Shrikes in South Carolina. J. Wildl. Manage. 57:352–357.

GRABER, R. R., J. W. GRABER, AND E. L. KIRK. 1974. Illinois birds: Laniidae. Illinois Nat. Hist. Biol. Surv. Notes 83.

HAAS, C. A. 1990. Breeding ecology and site fidelity of American Robins, Brown Thrashers, and Loggerhead Shrikes in shelterbelts in North Dakota. Ph.D.diss, Cornell Univ., Ithaca, NY.

JEWETT, S. G., W. P. TAYLOR, W. T. SHAW, AND J. W. ALDRICH. 1953. Birds of Washington State. Univ. of Washington Press, Seattle, WA.

KRIDELBAUGH, A. L. 1982. An ecological study of Loggerhead Shrikes in central Missouri. M.Sc.thesis. Univ. of Missouri, Columbia, MO.

LUUKKONEN, D. R. 1987. Status and breeding ecology of the Loggerhead Shrike in Virginia. M.Sc.thesis, Virginia Polytech. Inst. and State Univ., Blacksburg, VA.

LYMN, N., AND S. A. TEMPLE. 1981. Land-use changes in the Gulf Coast region: links to declines in midwestern Loggerhead Shrike populations. Pass. Pigeon 53:315–325.

MILBURN, T. 1981. Status and distribution of the Loggerhead Shrike *Lanius ludovicianus* in the northeastern United States. Report to U.S. Dept. Inter., Fish and Wildl. Serv., Off. Endan. Species.

MILLER, A. H. 1931. Systematic revision and natural history of the American shrikes (*Lanius*). Univ. Calif. Publ. Zool. 38:11–242.

NICKELL, W. P. 1958. Variations in engineering features of the nets of several species of birds in relation to nest sites and nesting materials. Butler Univ. Botanical Studies 13:121–140.

NOVAK, P. G. 1989. Breeding ecology and status of the Loggerhead Shrike (*Lanius ludovicianus*) in New York State. M.Sc.thesis, Cornell Univ., Ithaca, NY.

PORTER, D. K., M. S. STRONG, J. B. GIEZENTANNER, AND R. A. RYDER. 1975. Nest ecology, productivity and growth of the Loggerhead Shrike on the shortgrass prairie. Southwest. Nat. 19:429–436.

PRESCOTT, D.R.C., AND D. M. COLLISTER. 1993. Characteristics of occupied and unoccupied Loggerhead Shrike territories in southeast Alberta. J. Wildl. Manage. 57:346–352.

RICH, T. 1980. Nest placement in Sage Thrashers, Sage Sparrows and Brewer's Sparrows. Wilson Bull. 92:362–368.

SAS INSTITUTE. 1985. SAS user's guide: statistics, 5th ed. Cary, NC.

SCOTT, T. A., AND M. L. MORRISON. 1990. Natural history and management of the San Clemente Loggerhead Shrike. Proc. W. Found. Vert. Zool. 4:23–57.

Siegel, M. S. 1980. The nesting ecology and population dynamics of the Loggerhead Shrike in the blackbelt of Alabama. M.Sc.thesis, Univ. of Alabama, Tuscaloosa, AL.

Soil Conservation Service. 1986. Map. Potential natural vegetation map of Idaho.

Stearns, H. T. 1930. Guide to Craters of the Moon National Monument, Idaho. Reprinted from: Bull. of the Idaho Bur. of Mines and Geol., Vol. 13, July, 1928. Caxton Printers, Ltd., Caldwell, ID.

Tyler, J. D. 1992. Nesting ecology of the Loggerhead Shrike in southwestern Oklahoma. Wilson Bull. 104:95–104.

Vale, T. R. 1975. Presettlement vegetation in the sagebrush-grass area of the intermountain west. J. Range Mange. 28:32–36.

Woods, C. P. 1993. Variation in Loggerhead Shrike nest composition between two shrub species in southwest Idaho. J. Field Ornith. 64:352–357.

Woods, C. P. 1994. The Loggerhead Shrike in southwest Idaho. M.Sc.thesis, Boise State University, Boise, ID.

Yensen, D. L. 1980. A grazing history of southwestern Idaho with emphasis on the Birds of Prey Study Area. Prog. Rep., U.S. Dept. Inter., Bureau Land Manage., Boise, ID.

Yosef, R. 1993. Influence of observation posts on territory size of Northern Shrikes. Wilson Bull. 195:180–183.