Effects of Fire on Wildlife in Southwestern Lowland Habitats

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The purpose of this paper is to review and synthesize information about the responses of wildlife to fire in the grasslands and shrublands of Arizona and New Mexico. We begin by describing the effects of one particular fire at a site we have been studying intermittently since 1974. We intend this as more than personal indulgence. Rather, the wildlife responses to this fire, and to others we have studied at the same site, lead us to certain conclusions that we next will test for lowland southwestern habitats generally.

On July 14, 1987, lightning ignited a wildfire on the Appleton-Whitten Research Ranch, a National Audubon Society sanctuary in the grasslands and oak savannahs of southeastern Arizona. Summer rains were late that year, and in fact they had not yet begun to any significant extent. Atmospheric conditions were hot, dry, and very windy. There was a great deal to burn, since the area had not been grazed by livestock since 1968 (Bock et al. 1984). U. S. Forest Service fire crews, with the assistance of local volunteers, were able to contain the burn by the evening of July 15, largely along firebreaks such as the sanctuary’s boundaries with adjacent operating cattle ranches. By this time, the fire had completely blackened the grasses, shrubs, and scattered mesquite on about one-third of the sanctuary’s 3200 ha.

Summer rains began four days after the fire, followed quickly by dramatic vegetation and wildlife responses. Because we had worked on the sanctuary prior to this burn, on sites spared as well as those combusted by the fire (Bock et al. 1986), we were in a position to accurately measure its effects upon plant and animal populations. At the end of the 1987 growing season, burned study plots supported about one-third the grass cover, less than half the shrubs, and over twice the herb cover as that found on nearby unburned areas. Insect populations were reduced, and seed production was dramatically higher on the burn. Certain animal species requiring heavy cover, such as the cottonrat (Sigmodon hispidus), Botteri’s sparrow (Aimophila botterii), Cassin’s Sparrow (A. cassini), and grasshopper sparrow (Ammodramus savannarum), disappeared entirely from the burned areas. Overall, the wildlife response was positive. Mule deer (Odocoileus hemionus) and pronghorn (Antilocapra americana) used burned areas more frequently than comparable unburned sites through the fall and winter of 1987-88. Mourning doves (Zenaida macroura) concentrated on the burn in that first post-fire year in densities nearly fifteen times higher than those typical of unburned grassland. Birds collectively were more than five times more abundant on the burn than they were either on adjacent control plots, or on our burned plots in years prior to the fire.

By the end of the 1988 growing season, grass cover on burned plots was nearly 75% that on control areas, while herb cover remained more than two times greater on the burn. Shrubs were reduced, due largely to near total fire-kill of burro weed (Haplopappus tenuisectus). Cassin’s, Botteri’s, and grasshopper sparrows had begun to re-colonize the burn, but in numbers far lower than nested in unburned sites. Horned larks (Eremophila alpestris), lark sparrows (Chondestes grammacus), and mourning doves were the most abundant birds breeding on the burn. Rodents remained virtually absent from burned plots.

Results of this 1987 fire, and others we have studied previously at the Research Ranch (Bock et al. 1976; Bock and Bock 1978, 1987), suggest the following about fire and wildlife in grasslands on the sanctuary:

1. Fire has been a natural and powerful evolutionary force shaping this ecosystem. Most species are fire tolerant; some are fire-dependent.

2. Livestock grazing interrupts the fire regime in this ecosys-
tem, making it difficult to evaluate the natural effects of fire on vegetation and wildlife, or even to use fire as an effective management tool.

3. The sanctuary's grasslands recover quickly from the effects of fire, with grasses and herbs returning to pre-burn conditions in three to four years.

4. Prescribed fire should be used when possible to create rather fine-scale mosaics of stands at various stages of post-fire ecological succession. Probably no site should be re-burned at least until its grasses and herbs have recovered to pre-burn conditions. Some wildlife species require heavy cover typical of areas protected from fire as well as grazing; others thrive on increased resources temporarily available following a recent burn; still other species need both sorts of habitats simultaneously.

A necessary first step in reviewing the effects of burning on wildlife across the full range of lowland southwestern ecosystems is to group the landscape into major habitat types. The excellent volume edited by D. Brown (1982) provides a logical starting point. However, this treatise divides the study area into more biotic communities than we can reasonably consider. Furthermore, as noted by Lowe and Brown (1982), many of these communities grade into one another both spatially and through time. Of particular relevance to the present review, fire may have played a major role in the past in maintaining many southwestern sites as grassland, where today they are dominated largely by desertscrub.

The landscape units we have chosen to examine correspond closely to those broad biogeographic provinces recognized by Lowe and Brown (1982, fig. 3). These are units with generally distinct floral and faunal histories (e.g. Axelrod 1985, Cronquist et al. 1972, Lowe and Brown 1982), among which fire may differ in historical importance and management applicability. These habitat associations are:

1. Great Basin shrubsteppe,
2. Interior chaparral,
3. Madrean evergreen woodland,
4. Chihuahuan shrubsteppe (including Chihuahuan desertscrub, semidesert grassland, and plains grassland), and
5. Sonoran and Mojave desertscrub.

For each of these landscape units we shall review 1) something of their distribution, composition, and history, especially as relates to fire, 2) known effects of fire on vegetation, and 3) wildlife responses to fire, based upon actual studies, or as can be projected from what is known of the species' habitat requirements. Severson and Rinne (this volume) consider similar information about fire in higher-elevation southwestern forests and woodlands.

**Great Basin Shrubsteppe**

The most common vegetation throughout the Intermountain West is Great Basin shrubsteppe, with its most conspicuous shrub being big sagebrush (*Artemisia tridentata* L.). Particularly in the northern Great Basin, an understory of cool-season perennial grasses is (or was) important. Dominant species are in the genera *Agropyron*, *Festuca*, and *Stipa* (Wright et al. 1979). The exotic annual cheatgrass (*Bromus tectorum* L.) has spread widely within historic times (Mack 1981), as the native perennial grasses are (or was) important in southern Great Basin shrubsteppe before the depredations of domestic grazers, though likely not to the same extent as farther north.

Humphrey (1974) felt that Great Basin shrubsteppe burned more frequently than any other North American desert. Furthermore, because big sagebrush does not re-sprout following fire, burns in this habitat have the potential to drastically change the structure and composition of these plant communities. However, this presumes the presence of sufficient grass cover to carry fire from one shrub to the next, as well as to provide a source of grass propagules to colonize and spread following the burn.

Whatever the prehistoric importance of fire in Great Basin shrubsteppe, the Southwest can figure only...
peripherally in this debate, since the habitat itself is only marginally present in the region. Sagebrush-dominated shrubsteppe occurs in the Arizona Strip north of the Grand Canyon, and in the Painted Desert region of northeastern Arizona and northwestern New Mexico (Turner 1982a).

Fire and Vegetation

Fire has been little-studied in Great Basin shrubsteppe of the Southwest. The Bureau of Land Management has been experimenting with prescribed burns in its Arizona Strip District (F. L. Leavitt, pers. comm.). Despite an initial scarcity of grass cover, a very high percent kill of sagebrush was achieved, with little shrub reinvasion in the first few post-fire years. Burns were seeded by broadcast or drilling, with an especially good establishment of sand dropseed [Sporobolus cryptandra (Torr.) Gray] and sideoats grama [Bouteloua curtipendula (Michx.) Torr.]. These managers are to be commended for using native grasses in their re-seeding program.

Elsewhere in the Great Basin, fire has been used extensively as a land management tool. Burning generally has proven the most economical and effective means of reducing big sagebrush (e.g. Wambolt and Payne 1986). Native grasses and forbs can be damaged by hot or too frequent fires, but they are generally tolerant of spring and fall burns. Reduced shrub cover at least potentially provides the opportunity for increases in grass and forb species (see Wright et al., 1979, for details).

Fire and Wildlife

We are unaware of any studies specifically designed to test the effects of burning on wildlife in Arizona or New Mexico sagebrush habitats. However, BLM personnel have seen large influxes of mule deer following prescribed burning and seeding in the Arizona Strip. Presumably this resulted from "the large increase in forbs and grasses and the re-sprouting of browse species" (F. L. Leavitt, pers. comm.). Hobbs and Spowart (1984) found that prescribed burning elevated the nutritional quality of mule deer and mountain sheep (Ovis canadensis) diets in winter, but not in spring, in Colorado sagebrush-grassland. These changes did not result from increased nutritional value of individual plants, but rather from increased availability of grasses in the burned areas.

While burning may cause temporary increases in certain high-quality grass and herb forage, it is important to remember that many Great Basin shrub species provide essential winter deer browse. Most of these species, such as sagebrush and antelope bitterbrush [Purshia tridentata (Pursch.) DC.], are damaged or eliminated by fire (Severson and Medina 1983). Pronghorn antelope similarly may benefit from increased forb cover following fire, but only if sufficient shrub browse is spared for the winter (Wright and Bailey 1982). A mosaic of burned and unburned stands probably provides the best overall big game habitat, although this needs to be tested specifically for southwestern sagebrush stands.

Because burning potentially can convert Great Basin shrubsteppe into pure and relatively persistent grassland, it can have a dramatic impact on those wildlife species sensitive to structural habitat features. This has been particularly well-studied with regard to songbirds, many of which require shrubs for nesting or as song perches. Although Great Basin shrubsteppe birds have rather wide ecological tolerances (Wiens and Rotenberry 1981), nevertheless certain species will be drastically affected by changes in shrub and/or grass cover. Sage thrashers (Oreoscoptes montanus) and sage sparrows (Amphispiza belli) would be eliminated by a complete sagebrush kill, whereas horned larks (Eremophila alpestris) and vesper sparrows (Poecetes gramineus) are likely to increase (Peterson and Best 1988). A fire in ungrazed sage-grassland in southcentral Montana completely eliminated all sagebrush, including its stems and branches. Only western meadowlarks (Sturnella neglecta) nested on the burn in the first three years following the fire (Bock and Bock 1987), whereas adjacent unburned sites supported meadowlarks plus four additional species: lark sparrow (Chondestes grammacus), lark bunting (Calamospiza melanocorys), grasshopper sparrow, and Brewer's sparrow (Spizella breweri).

Conclusions

Prescription burning can have beneficial impacts on wildlife in Great Basin shrubsteppe, as long as it is used to create rather fine-scale mosaics of stands in various stages of post-fire succession. Fire can increase the quality and quantity of grasses and forbs, but it also reduces cover and browse. We suspect that mosaics of burned and unburned shrubsteppe represent the condition of most Great Basin lowlands prehistorically, when the native grassland understory was relatively intact.

Interior Chaparral

There are no North American ecosystems better characterized as evolving under the influence of fire than chaparral (e.g., Conrad and Oechel 1982, Sweeney 1956). Interior chaparral is best developed in a band south of the Mogollon Rim, from northwestern Arizona into extreme southwestern New Mexico (Carmichael et al. 1978, Pase and Brown 1982). Shrub predominate on unburned sites, often forming a nearly complete canopy. Lightning-caused wildfires are common (Boland 1982), and top-kill of shrubs can be very high, but the shrub species
are adapted to recover and re-colonize quickly. Dominant species such as shrub live oak (*Quercus turbinella* Greene) and mountain mahogany (*Cercocarpus* spp.) re-sprout from root crowns (Pase and Granfelt 1977), while others such as manzanita (*Arctostaphylos* spp.) and ceanothus (*Ceanothus* spp.) reproduce from large and persistent seed banks that germinate following fire. Grasses and herbs are scarce where shrubs predominate, but they can become temporarily abundant after fires (Pase and Brown 1982).

Historically, management efforts in Arizona chaparral have been directed at reducing shrub cover and increasing grasses, principally to improve water yield and livestock production (Brown et al. 1974, Hibbert et al. 1974). Prescribed burning has been an important tool in these chaparral “conversion” efforts. However, the shrubs recover within 5 to 10 years following burning (Pase and Granfelt 1977). Therefore, other control methods such as root plowing and herbicide treatment usually also have been used (Baldwin 1968, Hibbert et al. 1974), along with seeding of African lovegrasses (*Eragrostis* spp.).

**Fire and Vegetation**

Fire in Arizona chaparral appears not to result in long-term increases in grass and herb cover, except when coupled with repeated re-burns, or when followed by application of herbicides and seeding with exotic grasses (Pase and Pond 1964, Pase and Knipe 1977). Unseeded chaparral stands recover in 5 to 10 years, though they may not carry fire for as long as 20 years (Cable 1957, Hibbert et al. 1974, Pase and Pond 1964). Pase and Knipe (1977) studied the effect of a winter prescribed burn on a converted chaparral watershed in the Tonto National Forest. The area had been previously burned, treated with herbicide, and seeded with a mixture of African lovegrasses. The winter fire did not reduce subsequent production by the exotic grasses, but it did increase abundance of a dominant annual herb, *Ipomoea coccinea* L. Because of the complex pre-burn unnatural treatment of the area, and the fact that a winter burn is outside the natural fire season, it is difficult to determine what this study tells us about the natural role of fire in interior chaparral. Mayland (1967) found increased nitrogen availability in soils under scrub oak and mountain mahogany chaparral that was treated with herbicide and then control-burned in the fall.

**Fire and Wildlife**

There is general agreement that prescribed fire will be most beneficial to wildlife in interior chaparral if it is used to create relatively small openings, especially in areas with heavy cover of the less palatable shrubs (Brown et al. 1974, Hibbert et al. 1974, Severson and Medina 1983). Chemical treatment has the undesirable effect of eliminating palatable and well as unpalatable browse (Severson and Medina 1983). At least in southeastern Arizona, stands dominated by exotic African lovegrasses support a greatly diminished variety and abundance of wildlife compared to native grasslands (Bock et al. 1986). Burning outside the natural fire season could select against native plants and animals that have lived in this region since the last glaciation.

Few studies have considered the responses of individual wildlife species to burning of interior chaparral. More such work has been conducted in California chaparral, and we have attempted to extrapolate from these results to the southwestern situation where it is appropriate.

Szar (1981) compared bird populations in Arizona chaparral with those of adjacent grassland and riparian habitats created over the preceding 20 years by repeated burning, seeding with African lovegrasses, and herbicide applications. Because the riparian zone, created by increased water yield in the manipulated area, was small relative to grassland, the overall effect of the conversion was a substantial loss of breeding birds. Densities ranged from a high of 321 pairs per 40 ha in riparian, to 103 and 24 pairs in chaparral and grassland, respectively. Species most abundant in untreated chaparral included Gambel’s quail (*Callipepla gambelii*), Bewick’s wren (*Thryomanes bewickii*), bridled titmouse (*Parus wollweberi*), and rufous-sided towhee (*Pipilo erythrophthalmus*). Rock wren (*Salpinctes obsoletus*) and rufous-crowned sparrow (*Aimophila ruficeps*) were restricted to grassland. Eight species, including especially Bell’s vireo (*Vireo bellii*), yellow warbler (*Dendroica petechia*), and Scott’s oriole (*Icterus parisorum*), were most abundant in the riparian site. Breeding birds may have been uncommon in the converted grassland because of the dominance of the African exotics (Bock et al. 1986).

Lawrence (1966) compared breeding birds of burned vs. unburned chaparral in the Sierra Nevada foothills in California. Total bird numbers were little-changed by the fire, but species’ relative abundances were substantially affected through three post-fire years. Brushland species, such as the California quail (*Callipepla californica*), Bewick’s wren, and brown towhee (*Pipilo fuscus*) declined, while grassland birds such as the mourning dove and western meadowlark increased.

We can find no data on rodent responses to fire in Arizona chaparral. Several studies in California brushlands show that fire has a dramatic, if short-term, effect on this group of mammals. Fires probably cause little direct rodent mortality, since most species survive in underground burrows. However, woodrats (*Neotoma* spp.) living in above-ground nests are killed in large numbers (Chew et al. 1959). Fires initially appear to
cause all rodents to abandon chaparral (Cook 1958, Wirtz 1982). However, recolonization is rapid, especially for those species preferring open or grassland habitats, such as voles (Microtus spp.), pocketmice (Peromyscus spp.), and harvest mice (Reithrodontomys spp.). Shrub specialists such as *Peromyscus truei* and *P. californicus* were unable to occupy burned California chaparral, due to lack of suitable habitat (Lawrence 1966).

Prescribed burning potentially can improve interior chaparral for deer, if it results in mosaics of stands in various stages of post-fire re-growth (Hibbert et al. 1974, Severson and Medina 1983). Young shoots and herbs could provide higher quality forage, while older stands may provide some cover. Excessive brush control, whether through fire or chemical treatment, could reduce habitat suitability for either white-tailed deer (*Odocoileus virginianus*) or mule deer (McCulloch 1972). Fire has been used to improve California chaparral for mule deer (Biswell 1969, Taber and Dasman 1958). In northern California, mule deer herds increased 300%, and animals were in better condition, following initiation of a prescribed burning program (Thornton 1982). Large wildfires were excluded, while controlled burns were conducted on 8 to 10 ha parcels planned for a 25 to 30-year rotation. Perennial grass fuel-breaks also were burned, on a planned 10-year rotation. Deer herd improvements were attributed to (1) creation of desirable cover/orage ratios, (2) increased nutrient quality of shrub sprouts for about three post-fire years, and (3) increased quantity of grass forage on the fuel break sites for at least three post-fire years.

**Conclusions**

From a wildlife standpoint we question the desirability of herbicide treatments, seeding with exotic grasses, or burning outside the natural fire season, in chaparral ecosystems. All have the potential to reduce the wildlife value of such areas (to say nothing of their impacts on the native flora), and especially if applied on too broad a geographic scale. An exception might be to use chemical treatments to create fire-breaks for management of small-scale prescribed burns. Such small burns, if repeated on perhaps a 10 to 20 year rotation, almost certainly will increase the abundance and variety of all kinds of wildlife in interior chaparral. Reseeding, if necessary and practical, should involve the use of indigenous plants.

**Madrean Evergreen Woodland**

This ecosystem type is centered in the Sierra Madre of Mexico, but it extends northward into the foothills and mountains of southeastern Arizona, southwestern New Mexico, and Trans-Pecos Texas (D. Brown 1982). At higher elevations it grades into pine-oak woodland, while at lower elevations it is open and savannah-like. This lower-elevation savannah, or encina, is the focus of this review.

Dominant trees are oaks, most particularly Emory oak (*Quercus emoryi* Torr.), Arizona white oak (*Q. arizonica* Sarg.), Mexican blue oak (*Q. oblongifolia* Torr.), and gray oak (*Q. grisea* Liebm.). Alligator bark and one-seed juniper (*Juniperus deppeana* Steud and *J. monosperma* (Engelm.) Sarg.) and Mexican pinyon (*Pinus cembroides* Zucc.) also are locally common (D. Brown 1982). Perennial grasses can be very well developed, especially on level, lowland, and ungrazed sites. Grass species diversity appears to be very high, some common species being side oats grama, plains lovegrass (*Eragrostis intermedia* Hitch.), Texas bluestem (*Andropogon cirratus* Hack.), and muhlys (*Muhlenbergia* spp.). There is much debate about the prehistoric role of fire in southwestern oak savannah (Hastings and Turner 1965, Severson and Medina 1983), and not much evidence upon which to rely. Bahre (1985) reports on 33 fires in southeastern Arizona that were described in local newspapers between 1859 and 1890; some of these certainly involved the encinal. Humphrey (1987) visited and photographed U.S.-Mexico border monuments in 1983-84, and then compared his photos with those taken in 1892-93. Many upper elevation evergreen woodland sites are more heavily wooded today than they were in the 1890's, which Humphrey attributes at least partially to fire suppression policies in place over the past 90 years. However, the lower elevation encinal sites appeared relatively little-changed.

**Fire and Vegetation**

Johnson et al. (1962) studied juniper and oak mortality following a hot June wildfire in southeastern Arizona. Mortality of Emory oak and Arizona white oak was 10% to 20% on the burn, but these results were difficult to interpret because of apparent drought-related oak mortality on the adjacent control area. Fire killed nearly 80% of one-seed juniper in all size classes, but only 32% of alligator junipers less than 3 inches in diameter, and only 23% of larger alligator junipers. Basal sprouting of burned alligator juniper was 42%, while only 10% of one-seed juniper sprouted after the fire.

Gawith (1987) studied the effects of a May prescription fire on alligator juniper invading grassland on Fort Huachuca. Fuel levels were high because the area had been ungrazed for about 30 years. Fire intensity ranged up to 2800 kilowatts per meter of fire front. Only 13% of burned trees died, but mortality was highest in small size classes. Gawith concluded that repeated burning would prevent alligator juniper invasion into ungrazed encinal grasslands.
Bock and Bock (1987) studied the effects of small (600 m²) prescribed burns in oak savannah on the Appleton-Whittell Research Ranch, west of the Huachuca Mountains. Burns took place on 25 May 1984, when temperature was 33°C, relative humidity 17%, and winds generally calm. The fires were cool and of short duration. Scattered mature Emory and Arizona white oak were unaffected by burning, and oak seedling densities did not differ on burn vs. control plots through two post-fire growing seasons. Shrub densities were not affected, but heights of two species, *Mimosa biuncifera* Benth. and *M. dysocarpa* Benth., were significantly reduced on burned plots through two post-fire years. Perennial grass cover was reduced on burned plots by about 27% through one year, but this fire effect disappeared after two years. Herb cover nearly doubled on burned plots relative to controls in the first year, but again this difference disappeared by the end of the second post-fire year.

Our study plots were burned again by the extremely hot wildfire described in the introduction to this review. Our impression is that more woody plant mortality occurred as a result of this more intense fire, but such a conclusion must await data collection through more post-fire years.

**Fire and Wildlife**

Very little is known about wildlife responses to fire in low-elevation encinal. Barsch (1977; cited in Severson and Medina 1983) found that white-tailed deer benefitted from a wildfire that stimulated desirable browse in upper encinal. R. Brown (1982) found that Montezuma quail (*Cyrtonyx montezumae*) in southeastern Arizona are dependent upon dense perennial grasses as escape cover. These quail disappeared from heavily grazed sites, even though food supplies remained abundant. Presumably large-scale fires could have a similar negative impact on Montezuma quail.

We studied the responses of birds, rodents, and vegetation to a human caused February wildfire in open oak savannah on the Research Ranch, six years after livestock removal (Bock et al. 1976). Because no pre-burn data were collected, the legitimacy of our conclusions depends on the validity of comparisons between burned and adjacent unburned areas. There was no sign of bird mortality on this Research Ranch burn, although up to half the leaves on many oaks were scorched and killed. Grass cover was reduced through two post-fire summers, but there was little change in herb cover. Seed production was much higher on the burn in the first post-fire year.

Birds collectively were about 18% more abundant on the burn over 18 post-fire months. Two seed-eating species, mourning dove and chipping sparrow (*Spizella passerina*), were largely responsible for this difference. The grasshopper sparrow is dependent on heavy grass cover in southeastern Arizona (Bock and Webb 1984), and it disappeared entirely from the burned area for the duration of the study. Rodents were about 40% less common on the burn than on the adjacent control, and no species was trapped significantly more often on the burn. White-throat woodrats (*Neotoma albigula*), whose nests were burned away, and least cottonrats (*Sigmomax minimus*), grazing rodents typical of dense grass, were virtually absent from the burn through two post-fire summers.

**Conclusions**

Much more research is needed into the responses of vegetation and wildlife to prescribed burning in lowland encinal. Managers planning to burn this habitat should try to notify the research community early enough to permit pre-burn sampling.

The impact of fire in encinal is likely to depend heavily on tree densities, ground fuel levels, and atmospheric conditions. Cool fires should kill few trees, but they may temporarily reduce grass cover, stimulate herbs, and increase seed production. Reduced ground cover should favor songbirds over rodents.

**Sonoran Desertscrub**

This diverse biome occupies most of lowland southwestern Arizona, grading to a small and uncertain degree into Mojave desertscrub in the northwestern part of the state (Humphrey 1974, Shreve 1942, Turner 1982b, Turner and Brown 1982). The vegetation is complex, both within and between localities. Creosote-bush (*Larrea tridentata* (DC) Coville) and white bur sedge (*Ambrosia dumosa* (Gray) Payne) are dominant shrubs in lowland sites. Sonoran uplands are characterized by a wide variety of trees and large cacti, including ironwood (*Olneya tesota* Gray), palo verde (*Cercidium spp.*), mesquites (*Prosopis spp.*), and saguaro (*Carnegia gigantea* (Engelm.) Britt. & Rose).

There is much debate about whether a true “desert grassland” is (or ever was) an important part of the Sonoran Desert in Arizona. Even more controversial is the role that fire, lightning or human-caused, may have played in reducing woody vegetation or maintaining such a grassland prior to European colonization (Bahre 1985, Dobyns 1981, Hastings and Turner 1965). Certainly it is gone today, except for swales and lowlands occupied by big galleta (*Hilaria rigida* (Thurb.) Benth.) or tobobosgrass (*H. mutica* (Buckl.) Benth.).

Humphrey (1974, 1987) considered fires to be rare and generally unimportant in the Sonoran and Mojave deserts, except in *Hilaria* stands or in places where the desert intergrades with semidesert grassland. Wright and Bailey (1982) do not
even include a chapter on the Mojave or Sonoran deserts in their general treatise on fire ecology. Nevertheless, fires do occur in these deserts today, following years when exceptional rains produce a heavy ground cover of annual herbs and grasses (Cave and Patten 1984, McLaughlin and Bowers 1982, Pase and Granfelt 1977, Roundy 1986). Many of these annuals are introduced, however, so it remains unclear how often native annuals could have carried fire (Rogers and Steele 1980, Roundy 1986).

In the absence of dendrochronological data, Rogers and Steele (1980) attempted to use degree of fire adaptation in perennial plants as evidence for historical fire frequency in the Sonoran Desert. They concluded that "...positive adaptations are common, but are weakly developed" (p. 15), that post-fire recovery time is very long (up to 20 years), and that fire management should be generally conservative.

Fire and Vegetation

Wildfires and controlled burns in upland Sonoran Desert sites have caused substantial mortality of woody plants and cacti. Fire reduced density and cover of perennial plants by 91% and 84%, respectively, on the Granite Burn near Florence, Arizona (McLaughlin and Bowers 1982). Mortality of bur sage was 92%, creosote-bush 61%, and palo verde 63%. Mortality of saguaros was 31% after 19 post-fire months, and consisted mostly of individuals <2 m tall; but saguaro death on this same area rose to 68% after 54 months, and included many large plants (Rogers 1985).

It is not clear whether Sonoran Desert fires necessarily even increase ground cover. Cave and Patten (1984) found that fire reduced annual plant density but increased biomass. This seeming paradox is largely explained by fire-caused reductions in dense red brome (Bromus rubens L.), coupled with increased biomass of schismus grass (Schismus arabis Nees.). Both these annual grasses are exotic. Herbs, native and introduced, showed mixed responses to the fires.

Fire and Wildlife

We have found no data on wildlife responses to fire in Sonoran deserts. Some projections can be made, based upon general knowledge about the habitat and food requirements of particular groups. The herpetofauna and avifauna of Sonoran Desert uplands are very rich (Phillips et al. 1964, Turner and Brown 1982, Tweit and Tweit 1986). Because most species depend upon the trees, shrubs, and cacti at least for cover and breeding sites, fire-caused mortality of these plants would have a highly negative impact on these groups. As an example, some of the best-known and most characteristic Sonoran Desert birds nest in cavities in saguaros. Fire-kill of this cactus would do great harm to these species, including flicker (Colaptes auratus), Gila woodpecker (Melanerpes uropygialis), brown-crested flycatcher (Myiarchus tyrannulus), and elf owl (Micrathene whitneyi). Many rodents would respond positively to increases in seeds produced by the desert annuals, but at least today these events are triggered largely by rainfall rather than fire. Mule deer occupy Sonoran Desert (Ordway and Krausman 1986), but seem to prefer areas with substantial cover. Desert bighorn (Ovis canadensis nelsoni) occupy only a small part of their original southwestern range. Bighorn sheep generally are thought to prefer grasses, and fire might be used on a limited scale to increase their forage (Monson and Sumner 1980).

Conclusions

Fire historically may have helped maintain an herb/grassland in parts of the Sonoran Desert now dominated almost exclusively by shrubs, trees, and cacti. Most evidence suggests that fires under present conditions are highly destructive of this perennial vegetation, without necessarily increasing cover of herbs and grasses, annual or perennial. More research is needed on this subject, including measuring responses of wildlife to wildfire. In the meantime, we can see little benefit and much potential harm coming to wildlife as a result of fire, wild or prescribed, in Sonoran deserts.

Chihuahuan Shrubsteppe

The biome we choose to call Chihuahuan shrubsteppe includes three community types: Chihuahuan deserts, semidesert grassland, and plains grassland (D. Brown 1982). There are three reasons why we have combined them for the present review. First, as noted by Lowe and Brown (1982:16), the boundaries between them are difficult to discern, at least under present circumstances. Second, these communities have a common, essentially Chihuahuan, ancestry (Axelrod 1985). Finally, fire and fire-exclusion have strongly influenced their composition and distribution.

Before recent desertification, true Chihuahuan deserts occurred largely in Mexico, extending into the U.S. only in southwestern New Mexico, up the Rio Grande Valley, and into a small part of extreme southeastern Arizona (D. Brown 1982, Schmidt 1979). Lowland plains, usually of limestone origin, were dominated by creosote-bush, tarbush (Flourensia cernua DC), whitethorn acacia (Acacia constricta Benth.), or honey mesquite [Prosopis juliflora var. glandulosa (Torrey) Cockerell.]. Higher slopes supported mixtures of succulents, especially of Agave and Yucca.

Semidesert grasslands occupied lowland sites in southern New Mex-
mimosa, acacia, by blue grama and other perennials. O f this can be attributed to the devas­

closure has not stopped the loss of black grama at the Jornada Experi­

ting impacts of livestock grazing. However, subsequent livestock ex­
closure has not stopped the loss of black grama at the Jornada Experi­
mental Range in southern Arizona (Martin 1986). Once relatively pure stands of black grama and other native grasses predominated. Today, burro weed [Haplopappus tenutectus (Greene) Blake], cholla (Opuntia fulgida Engelm.), and mesquite [Prosopis juli­flora (Swartz) DC] are abundant. Na­tive grasses have largely been re­
placed by the African exotic lovegrass, Eragrostis lehmanniana Nees. These changes can be attrib­
uted to the combined effects of con­
tinued livestock grazing, early fire exclusion, and introduction of the ex­otic. At least on the Research Ranch, stands of African lovegrasses are biologically sterile compared to adjacent stands of native vegetation (Bock et al. 1986).

Humphrey (1974, 1987) believes that fire played a major historical role in controlling woody plants in semidesert grassland. Fires were equally common and important in plains grassland (Jackson 1965, Vogl 1974), and shrub invasions have oc­
curred here as well following over­
grazing and fire suppression. Paraadoxically, the dominance of plains grassland in higher elevation sites in southeastern Arizona may be an arti­fact of livestock grazing. Bison (Bison bison) have not occurred in south­
eastern Arizona for at least the past 10,000 to 12,000 years (McDonald 1981; P.S. Martin, pers. comm.), so that cattle were a truly exotic force when they were introduced. Blue grama is perhaps the quintessential plains grassland species, and it domi­nates grazed areas today in parts of southeastern Arizona (Bock et al. 1984, Bock and Bock 1986). Yet on livestock exclosures, taller grasses such as plains lovegrass, wolftail (Lycurus pleoides H.B.K.), and certain bluestems (Andropogon spp.) become common, eventually at the expense of blue grama. However such grass­
lands are to be classified, we believe they too evolved with and were maintained by fire prehistorically.

Fire and Vegetation

The impact of fire on Chihuahuan shrubsteppe vegetation has been well studied and thoroughly reviewed (Martin 1975, Pase and Granfelt 1977, Wright 1980). While it may have broader wildlife applications, the ob­jective of most prescribed burning has been to reduce woody plants and stimulate livestock forage. The effi­cacy of these attempts seems rather site-specific, depending upon such things as precipitation patterns, stand species composition, and range condition.

Black grama can be severely dam­aged by fire in lowland sites (Cable 1965), though this grass also has been stimulated by burns (e.g. Ahlstrand 1982). Most plains and semidesert grasses recover from fire in one to three years, and some may be en­couraged by it (Bock and Bock 1987, Wright 1980). Tobosagrass may ei­ther be stimulated or temporarily re­duced by fire, depending upon pre­
cipitation following the burn (Neuen­schwander et al. 1978).

Shrubs such as burro weed and creosote-bush are killed by pre­scribed burning, if fuel levels are suf­ficient to carry fire (Cable 1973). Small mesquite are killed by fire, but individuals over 5 cm diameter are relatively invulnerable. Frequent fire in undisturbed grasslands histori­cally may have kept large areas free of mesquite. Yet recent invasions into disturbed sites can be reversed only by more drastic measures such as root plowing and chemical treatment (Martin 1975), or perhaps by dra­matic natural events such as rapidly repeated wildfires, extremely low winter temperatures, or a combina­tion of both.
Wright (1980) concluded that fire should not be used on black grama ranges, but that it can be an effective tool for reducing certain shrubs on mixed grama areas and in tobosa-grass. However, effectiveness of prescribed burning depends upon good grass cover.

Relatively little is known about the response of herds to fire in Chihuahuan shrubsteppe (Wright 1980), yet this component may be a critical one for certain wildlife. Relatively cool prescribed burns in an ungrazed Mimosa-Bouteloua-Eragrostis intermedia community on the Research Ranch had little impact upon grasses or forbs (Bock and Bock 1987); however, hot wildfires, such as the one described in the introduction to this review, stimulated herb production (see also Bock et al. 1976).

Fire and Wildlife

Compared to vegetation and livestock, the responses of wildlife to burning Chihuahuan shrubsteppe have been surprisingly little-studied. Much of what we can say about wildlife here must be based upon general knowledge of species’ food and habitat requirements.

Wildfires at the Research Ranch have had mixed impacts upon passerine birds (Bock et al. 1976, Bock and Bock 1988). Common winter species such as vesper sparrows and chipping sparrows concentrate on fresh burns in very large numbers, probably because of increased seed production. Species preferring bare ground, such as the horned lark and lark sparrow, breed in high numbers on either grazed or recently burned sites. Three sparrows characteristically nest in unburned and ungrazed grasslands at the Research Ranch: Cassin’s (Aimophila cassinii), Botteri’s (A. botterii), and grasshopper. These species are scarce or absent on grazed sites (Bock and Bock 1988), and they also avoid burns for one or two post-fire growing seasons.

Fire-caused reductions in woody plants are not necessarily beneficial to song or gamebirds. Bock and Webb (1984) found that while Cassin’s and grasshopper sparrows selected sites with heavy grass cover, they also were associated with scattered mesquite and low shrubs used as song-perches. Renwald (1978) found that scattered mesquite and lotebush [Ziziphus obtusifolia (Hooker) Gray] were an important habitat component for songbirds nesting in tobosa-grass in central Texas. Maurer (1985) found that eleven songbird species nested in greater numbers on parts of the Santa Rita Experimental Range with numerous mesquite, while ten different species preferred relatively open grassland.

Mourning doves concentrate in very large numbers on burned portions of the Research Ranch in the first fall after a wildfire (Bock and Bock 1988). Presumably this is because of increased forb seed production (Best and Smartt 1986, Bock et al. 1976). However, Germano et al. (1983) found that mourning dove, scaled quail (Callipepla squamata), and Gambel’s quail all preferred grassland areas with partial mesquite cover more than areas completely cleared of mesquite (see also Ault and Stormer 1983). A mosaic of sites with varying mesquite densities also would maintain high reptilian diversity (Germano and Hungerford 1981).

Mule and white-tailed deer require some woody cover (e.g. Quinton et al. 1979, Steuter and Wright 1980), but could perhaps benefit from increased grasses and high-quality browse sprouts following fire in semidesert and plains grassland (Anthony and Smith 1977, Severson and Medina 1983). Forbs were a very important component of the winter diet of mule deer and pronghorn in New Mexico (Stephenson et al. 1985). More research is needed to determine if these ungulates would respond positively to fire-caused forb increases in Chihuahuan shrubsteppe.

Conclusions

As with Great Basin shrubsteppe and interior chaparral, prescribed fire will benefit wildlife in Chihuahuan shrubsteppe if it is used to create mosaics. Fire can stimulate herb, seed, and perhaps grass production. Scattered shrubs and mesquite are likely to enhance the wildlife value of most areas, compared either to dense stands of woody vegetation or to pure grasslands. Ground cover should return to pre-burn conditions in about three growing seasons.

The most serious negative impacts on wildlife in semi-arid grasslands are from livestock grazing and the spread of exotic grasses. Fire cannot solve either of these problems. However, prescribed burning can be an integral and natural part of wildlife management in these ecosystems, especially if they can be restored to something resembling their prehistoric condition.

Miscellaneous Habitats

Riparian Woodland

Southwestern riparian woodlands are major centers of wildlife diversity, especially for birds (Johnson and Jones 1977, Johnson et al. 1985). In central Arizona, a complex history of wildfire, chemical treatments, and prescribed burning in interior chaparral increased runoff and stimulated growth of riparian trees (Szaro 1981). However, fire is difficult to manage and potentially very destructive in established riparian woodlands. At the Research Ranch, wildfires have killed mature cottonwood (Populus fremontii S. Wats.), sycamore (Platanus wrightii S. Wats.), velvet ash (Fraxinus velutina Torrey) and Arizona walnut (juglans major Heller). We do not recommend prescribed
burning in these limited habitats (see also Severson and Rinne, this volume).

Sonoran Savanna Grassland

This habitat once occupied lowland plains and bottomlands in the Altar and Santa Cruz valleys of southern Arizona (D. Brown 1982). Dominant grasses were Bouteloua and Aristida species, while mesquite was an important, if scattered, over-story tree. Heavy grazing and fire suppression caused native grasses to decline and shrubs to increase within the past century. In addition, Lehmann’s lovegrass has spread into the area.

The masked bobwhite quail (Colinus virginianus ridgwayi) once occupied Sonoran savannah grassland in Arizona and Mexico. This endangered subspecies became extinct in Arizona when its habitat was degraded to desertscrub (Goodwin and Hungerford 1977, Stromberg et al. 1986). The Buenos Aires Wildlife Refuge was established in the Altar Valley as a site for re-establishment of masked bobwhite in Arizona. Prescribed burning may prove to be an important tool in this recovery effort (W. Shifflett, pers. comm.). Recent wildfires and controlled burns have significantly reduced burro weed and snake weed [Gutierrezia sarothrae (Pursh.) Britton], set back mesquite, and possibly stimulated native perennials relative to Lehmann’s lovegrass.

Sacaton Floodplains

Big sacaton (Sporobolus wrightii Munro.) is a coarse, tall, perennial bunchgrass that forms nearly monotypic stands in broad floodplains of southeastern Arizona and southwestern New Mexico. These stands have been severely degraded historically by de-watering, channelization, and erosion (Cox et al. 1983). Sacaton frequently is burned to increase its value as livestock forage (Cox 1988). Spring and fall burning have long-term negative impacts, but sacaton burned in the natural mid-summer season recovers to pre-fire standing crop in three years (Cox and Morton 1986).

Wildfires in ungrazed sacaton stands on the Research Ranch have dramatic but short-term wildlife consequences (Bock and Bock 1978, 1979). Collared peccary (Dicotyles tajacu) use mature stands for cover, while many important food plants grow on recent burns. Yellowthroats (Geothlypis trichas), blue grosbeaks (Guiraca caerulea), and Botteri’s sparrows commonly nest in mature sacaton. However, doves, quail, and many wintering sparrows are attracted to large seed crops produced by forbs in recently burned stands. Grazing cottonrats are very abundant in unburned areas, whereas burns can support large populations of seed-eating rodents such as the hispid pocket mouse (Perognathus hispidus) and Merriam’s kangaroo rat (Dipodomys merriami).

Undisturbed sacaton appears to recover from fire in two to three years. A mosaic of stands in various stages of post-fire re-growth would be maximally beneficial to wildlife.

Management Implications And Recommendations

1. From a wildlife perspective, we believe the goal of land management should be to maintain or return ecosystems to their natural state, because such habitats will support the greatest diversity of native plants and animals. By natural state we mean the general condition in which they existed since the last glaciation but prior to European colonization. This goal should take precedence over management of a habitat for one particular species, except under very unusual and restricted circumstances. Most particularly, this overall goal may be at odds with the objective of increasing forage for livestock. Prescribed fire should be used when it is a tool for returning ecosystems to their natural state. We recognize this may be a problem (1) when it is not clear what was the natural state of a particular system, or (2) when historical habitat modifications make application of fire difficult, or (3) when such ecosystem restoration is in conflict with other land uses.

Fire is a natural ecological and evolutionary force in many southwestern lowland habitats, and it is one to which most native plants and animals are variously adapted. Fires kill little wildlife directly (Bendell 1974; Wright and Bailey 1982). Burning in the natural fire season is less likely to damage vegetation than burning out of season (e.g., Cox and Morton 1986).

2. There is universal testimony to the value of using fire to create mosaics for the benefit of wildlife (e.g., Lyon et al. 1978; Pase and Granfelt 1977; Severson and Medina 1983; Severson and Rinne, this volume). Unburned stands provide essential cover, while fires can stimulate browse, herbs, and grasses. Since many species need both unburned cover and fire stimulated food, mosaics should be on an appropriately small geographic scale.

3. Based upon available information, fire is likely to be
more destructive than beneficial to wildlife and wildlife habitat in (1) Sonoran and Mojave deserts, (2) black grama ranges in the lower parts of the Chihuahuan shrubsteppe, and (3) riparian woodlands. More research is needed into the effects of fire on wildlife in Madrean evergreen woodland.

4. Prescribed fire is a potentially powerful wildlife management tool in (1) Great Basin shrublands, (2) interior chaparral, (3) semidesert and plains grassland portions of Chihuahuan shrubsteppe, (4) Sonoran savanna grassland, and (5) sacaton floodplains. In all cases, however, fire frequency should not exceed the time required for post-burn recovery of herbs and grasses. Furthermore, the landscape mosaic always should include cover provided by shrubs and trees, where they are a natural component of the ecosystem.

5. Seeding burned areas with exotic grasses is highly undesirable, since stands of these grasses are sterile places for wildlife compared to mixtures of native species.

6. A major impediment to using fire creatively in southwestern lowlands is habitat alteration by domestic grazers. Reduced fuel loads prevent fires of a natural intensity. Fire-tolerant woody vegetation has become too well established, often crowding out grasses and herbs.

7. There is a great need for further research on the specific effects of prescribed burning on wildlife in southwestern

lowlands. Fire-ecological experiments must be doubly-controlled; that is, burn and control study plots must be monitored both before and for 3-4 years after a fire. In addition to population density, it may also be important to measure species’ survivorship and reproductive success on burned vs. unburned sites (Johnson and Temple 1986, Van Horne 1983).

Given the present emphasis on the value of field experiments in ecology in general, it should be possible to attract more university-based researchers, especially students, to study fire. Burning makes a dramatic and often spectacular ecological experiment. All that should be necessary on the part of managers is to contact researchers one or two years in advance, and then to burn where and when planned.

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