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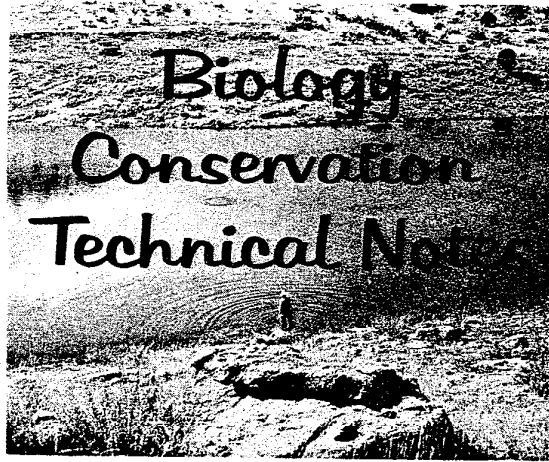


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U. S. DEPARTMENT OF AGRICULTURE NEW MEXICO SOIL CONSERVATION SERVICE

TECHNICAL NOTE No. 28

September 22, 1983

RE: DEER AND ELK HABITAT MANAGEMENT

Attached is a new reference which presents a synopsis of information on deer and elk habitat management in the southwest.

The article is based upon data gleaned from already available literature. There is a very complete section of literature cited.

Planners will find this to be a useful reference in Planning Practice 646 - Wildlife Upland Habitat Management

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Deer and Elk Habitat Management in the Southwest

by

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Abstract

Available information concerning habitat, habitat requirements, management direction, and research needs is summarized for three cervid species common to Arizona and New Mexico—elk, mule deer, and white-tailed deer. Emphasis is given to effects of timber and livestock management and prescribed fire and wild fire because of their existing and/or potential influence on deer and elk habitats.

Introduction

Mule deer,¹ white-tailed deer, and elk are among the most popular and important wildlife species in Arizona and New Mexico. Most of this popularity has derived from their value to sport hunters, but an ever increasing proportion of people simply enjoy seeing them in something that approaches a natural environment. This importance has dictated that these species be given a high level of management. One of the basic principles of big game management is to keep the population size in balance with the habitat—a premise based on the idea that habitat is the foundation on which a population exists.

Habitat management for deer and elk is proving to be quite complex, however. There are two immediate problems facing wildlife habitat managers. One that grows more troublesome and unmanageable every day is the effects of a continually growing human population on the animal's habitat (Poole 1981). This problem may be beyond the direct control of a habitat manager, but the ramifications of increasing demands for a decreasing habitat base are not—and this is putting overdue emphasis on the second problem. We lack the methodology to determine properly the relationships between an animal and its habitat. There are no established procedures, for example, by which cervid harvest can be tailored to habitat conditions (Connolly 1981). Similarly, we cannot predict what effects habitat modification will have on an animal population other than in a very general sense.

The objective of this paper is to assemble existing information on habitat relationships of mule deer, white-tailed deer, and elk applicable to Arizona and New Mexico. It is intended not to solve the problem of developing animal

/habitat interaction procedures but to facilitate their development by summarizing existing information. This report is intended to serve three general purposes. One will be to provide new habitat managers with a description of Southwestern habitat needs, management problems, and possible solutions. Another purpose will be to provide a summary and literature base for all resource managers that may help them deal with specific habitat management problems. Although this paper was written primarily for managers, its third purpose is to serve researchers by identifying those habitat interactions for which information is lacking, and by suggesting research direction that will provide the additional data necessary to formulate functional habitat management guidelines.

The Habitats

We have divided Arizona and New Mexico into three broad geographic areas based on biotic communities and animal use. These three regions, arbitrarily called the South, Mountains and Associated Areas, and the North, each contain a unique combination of biotic communities and different proportions of mule deer, white-tailed deer, and elk populations. The boundaries between the regions (Fig. 1) are based on the biotic communities included in each, which follow Brown and Lowe (1980). Detailed descriptions of each biotic community are provided in a separate paper.² Approximate percent composition of the biotic communities within each region is given in Table 1. General soil descriptions have been provided by Short (1979).

The South

The South is composed mostly of deserts and Semidesert Grasslands. Mule and white-tailed deer are present in significant numbers, but not uniformly across the region. Elk are absent. The biotic communities included in this region are the Mohave, Sonoran, and Chihuahuan Desertscrub; the Semidesert Grassland; and the Madrean Evergreen Woodlands. Isolated elements of the Montane Conifer Forest and Plains Grassland occupy scattered areas in southeastern Arizona and southwestern New Mexico. The Sonoran Desertscrub includes two subdivisions which are pertinent to this discussion, the Lower Colorado River and Arizona Upland.

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Research was conducted at the Station's Research Work Unit in Tempe, in cooperation with Arizona State University. Station headquarters is in Fort Collins, in cooperation with Colorado State University.

¹Common and scientific names of animals are listed in Appendix A and follow Cockrum (1960), Findley et al. (1975), and Decker (1978).

²Brown, David E., Charles P. Puse, and Raymond M. Turner. Biotic communities of the Southwest. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colo. [In preparation].

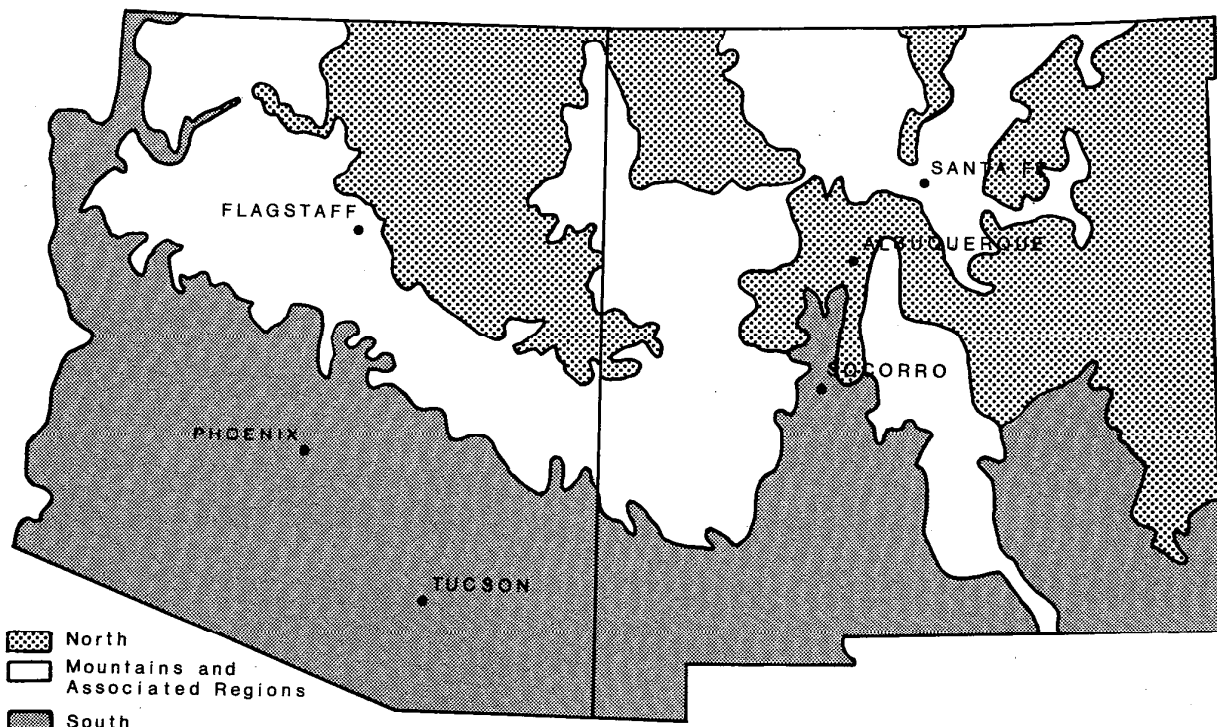


Fig. 1. The three generalized geographic regions of the Southwest.

Table 1. Approximate percent composition (to nearest 0.5% of total land area) of biotic communities in Arizona and New Mexico (modified from Short 1979).

| Region Community | Arizona | New Mexico |
|---------------------------------------|---------|------------|
| South | | |
| Mohave Desertscrub | 4.5 | 0 |
| Sonoran Desertscrub | | |
| Lower Colorado Subdivision | 17.0 | 0 |
| Arizona Uplands Subdivision | 14.0 | 0 |
| Chihuahuan Desertscrub | 1.5 | 8.5 |
| Semidesert Grasslands | 8.0 | 21.0 |
| Interior Chaparral | 4.0 | 0 |
| Madrean Evergreen Woodland | 3.0 | 1.0 |
| Montane Conifer Forest | 0.5 | 0 |
| Region total | 52.5 | 30.5 |
| Mountains and Associated Areas | | |
| Great Basin Conifer Woodland | 18.0 | 20.5 |
| Montane Conifer Forest | 7.0 | 9.0 |
| Subalpine Conifer Forest | 0.5 | 3.0 |
| Subalpine Grasslands | 0 | 0 |
| Alpine Tundra | 0 | 0 |
| Region total | 25.5 | 32.5 |
| North | | |
| Great Basin Desertscrub | 6.5 | 1.5 |
| Plains Grassland | 15.5 | 35.5 |
| Region total | 22.0 | 37.0 |

Mountains and Associated Areas

The Mountains and Associated Areas include woodlands and forests. Mule deer are distributed throughout. White-tailed deer and elk are less abundant, and their distribution is sporadic. This region extends from northwestern Arizona,

centering around and including the Kaibab Plateau south to the San Francisco Peaks, then southeast along the Mogollon Rim to the White Mountains in Arizona and the Mogollon and Black Mountains in New Mexico (Fig. 1). From this point, one lobe extends north to the Chuskas and another northeast to include the Zuni, San Mateo, Jemez, and San Juan mountains. East of the Rio Grande this region extends from the Sangre de Cristo Range in the north, southward to the Guadalupe Mountains. It includes the Sandia, Manzano, Jicarilla, Capitan, and Sacramento mountains, as well as the upper slopes of the San Andres Mountains.

The most important biotic communities occurring in this region are Great Basin Conifer Woodland, Montane Conifer Forest, and Subalpine Conifer Forest. Small relics of Alpine Tundra are present on the San Francisco Peaks and the Sangre de Cristo Range.

Various kinds of grasslands are found scattered throughout these biotic communities. Mountain meadows and subalpine grasslands occur within high elevation conifer types and Plains Grasslands are commonly associated with the Great Basin Conifer Woodland.

The North

The North includes high elevation or cold deserts and Plains Grasslands. All three cervids are conspicuously absent with only local exceptions. This region (Fig. 1) includes northeastern Arizona and northwestern New Mexico excluding the Chuska Mountains. Other areas in New Mexico, the upper Rio Grande Valley and the plains east of the Sangre de Cristo Range and Capitan Mountains, are also included.

There are two biotic communities within the region, and both are extensions of the two largest biomes in North

America—the Plains Grassland and the Great Basin Desertscrub. Extensive tracts of Great Basin Conifer Woodlands also occur throughout.

The Cervid Species

Two subspecies of mule deer are found in the two states. The one occupying the South is the desert mule deer; Rocky Mountain mule deer is found in the two northern zones. Subspecific identification is often difficult because there is an extensive zone of intergradation extending from the Mogollon Rim to the Gila River in Arizona (Hoffmeister 1962) and another broad area extending from the Black Mountains northeastward in New Mexico. Also, there are several areas in central and southern Arizona where the Rocky Mountain species from the Kaibab Plateau has been introduced (McCulloch 1968).

Another subspecies of mule deer, the burro deer, was once described as occurring along the lower Colorado River in California and Arizona (Cowan 1956) but has more recently been combined with desert mule deer (Hoffmeister 1962).

Like mule deer, two subspecies of white-tails have been recognized: Coues white-tailed deer and Texas white-tailed deer (Raught 1967).

Two subspecies of elk, Rocky Mountain and Merriam, once roamed the Southwest. Specific and subspecific differentiation of the genus *Cervus* has been complicated by extinction of both the eastern form of elk and Merriam elk of the Southwest. This has resulted in too few specimens to make adequate comparison, so separation of forms has been based largely on conjecture. Recent investigations consider Merriam elk to be only a marginally valid subspecies, not a separate species (Anderson and Barlow 1978).

Merriam elk were confined to the rugged Mogollon Rim country from the San Francisco Peaks area near Flagstaff, southeastward to Mogollon and Black mountains in New Mexico. They were present on the Sacramento and Guadalupe mountains in southern New Mexico east of the Rio Grande. Rocky Mountain elk reached their southernmost extension in the Jemez, San Juan, and Sangre de Cristo mountains (Bailey 1931, Gates 1967). Both groups, however, were extirpated from the Southwest in the early 1900's (Bailey 1931, Murie 1951, Gates 1967). Merriam elk became extinct, but Rocky Mountain elk from Yellowstone National Park were reintroduced into both states beginning in 1910 (Gates 1967).

Distribution and Habitat Use

Presettlement Conditions

Historical accounts of wildlife in the Southwest were collected and summarized for Arizona by Davis (1973) and, in less detail, for New Mexico by Bailey (1931). The status of wildlife prior to settlement in New Mexico is not as well documented as it is for Arizona. This can be attributed, in part, to a relative lack of diaries or other records left by early Spanish explorers (Humphrey 1958).

Southeastern Arizona and parts of New Mexico, particularly the Rio Grande Valley and portions of the Pecos, were settled very early, shortly after Coronado's expedition on the upper Rio Grande in 1540. Santa Fe was established in 1605. With the single exception of the Pueblo Revolt in 1680 and for approximately a dozen years thereafter, Spanish settlement of this region has been constant. When the first trappers from the United States reached the upper Rio

Grande (around 1825) there were about 4,000 people in Santa Fe, 6,000 in Albuquerque, and 9,000 in Taos. These urban centers made up approximately one-half the population of the upper Rio Grande (Pattie 1966).

Spanish settlement in Sonora, Mexico, and extreme southeastern Arizona was more tenuous than in central New Mexico. Conflicts among missionary sects, colonists, local administrations, and Indians prevented long-term settlement (Hastings and Turner 1965, Davis 1973). Thus, when the first settlers for the United States arrived in the Southwest, the upper Rio Grande had been settled for over 200 years. Pristine wildlife populations, particularly large ungulates, had already been severely altered. In Arizona, however, the pristine conditions were essentially unchanged. One exception was the presence of numerous herds of wild cattle left in parts of southeastern Arizona by the retreating Spaniards (Hastings and Turner 1965, Davis 1973).

Although continuous settlement of the middle Rio Grande effectively supplanted wild ungulates in the Valley, the areas on the Rio Grande above Taos and below the vicinity of Socorro were still "wilderness" in the early 1800's. Zebulon Pike, for example, "saw great sign of elk." (Coues 1895) on what is now the New Mexico-Colorado border while coming down the Rio Grande in 1807. Moving down the valley, from Santa Fe to Albuquerque, he described it as "a country better cultivated and inhabited than any I had yet seen." Pike was also impressed with the large flocks of goats. Just below Socorro, however, he "entered wilderness" and "saw many deer." Pattie (1966) also noted large herds of domestic livestock between Santa Fe and Socorro in 1825 but made no mention of wildlife until he was 2-4 days south of Socorro, where he noted a great number of bear, deer, and turkey.

Several sources of information are available, mostly from trappers and military expeditions, that give some indication of the abundance and distribution of deer and elk in Arizona. The following accounts are excerpts from Davis (1973), who collected and summarized these records. They are presented here, in part, to provide some idea of deer and elk distribution before Anglo-American settlement.

Elk

The mountain men must have covered the entire range of Merriam elk in Arizona, but these trappers left only one nebulous reference to the species. In 1826, Pattie's party shot a number of elk on the Colorado River east of the Grand Canyon (near the junction of the Colorado and Little Colorado Rivers). These may have been Rocky Mountain elk rather than Merriam. This reference is also questionable because Pattie did not assemble his notes until later in his life. The narrative also becomes obscure with respect to specific locations at this point.

Members of Sitgreaves' expedition in 1851 and Whipple's expedition in 1853 did not report elk in central Arizona. Because these parties crossed Arizona in fall and winter, it is conceivable that elk populations had moved to wintering grounds to the north, along the Little Colorado River. Thus, no elk may have been left as far south as the wagon route used by Sitgreaves and Whipple.

The next expedition, led by Beale, followed the same route, but in the summer. Its members reported "innumerable" elk sign along the middle stretches of the Little Colorado and on the rolling grasslands on either side of the River. No elk were actually seen, however.

Soldiers and prospectors found elk abundant in the area of the San Francisco Peaks in the Civil War period, but Coues made no reference to them in the vicinity of Fort Whipple (near Prescott) or anywhere else in central Arizona in 1864-65. Prospectors, however, reported them in the Weaver Mountains southwest of Prescott in 1863 and considered them initially abundant in that part of the state. Variability of evidence indicates that elk were present in the White Mountains and westward on the Rim but were probably relatively scarce.

Mule Deer

A vital element in the diet of explorers and settlers, the mule deer, like the white-tailed deer, was simply referred to as "deer" in most early records. Where the two species overlapped, it was often impossible to determine which one was being considered.

Mule deer were not uniformly abundant over the Southwest, but no other large ungulate occupied such a variety of habitats. In the South, they were observed from the Guadalupe Mountains in New Mexico all the way to and including the Colorado River bottoms. Although they were common along the Gila River, they were not numerous along the border except near isolated mountain ranges.

Mule deer were consistently noted in all areas of the Mountains and Associated Region, from the Great Basin Conifer Woodland to the ponderosa pine forests, including associated riparian habitats. Even during the period of low populations, about the turn of the century, mule deer were common in all New Mexico mountains except the San Matco, San Juan, and Zuni mountains where Bailey (1931) considered them absent or scarce.

Davis (1973) did not summarize information from the grasslands and northern deserts per se, but reports he quoted indicated that mule deer in this region depended heavily on riparian habitats.

Sitgreaves, in 1851, sighted both mule deer and pronghorn where the Little Colorado entered the Painted Desert, the deer mainly limited to bottoms. Whipple came down to the Zuni River in 1853 where he reported many mule deer. On the Little Colorado, between the Puerco River and Chevelon Creek, he again reported an abundance of mule deer in riparian growth.

Other expeditions did not report similar results from other areas in the North. Simpson crossed the Chuska Mountains from New Mexico in late summer 1849 and, on Black Creek, between the modern sites of Window Rock and Lupton, noted after a deer had been killed by his party, that, "This is the first deer which has been killed by any of the party. The scarcity of this kind of game may therefore readily be inferred." (Davis 1973).

Later, Merriam (1890) explored portions of the Little Colorado River northwest of Flagstaff. Although he noted that black-tailed deer (mule deer) were abundant on "San Francisco Mountain" and the Grand Canyon, he did not observe them on the "Little Colorado Desert." He had spent 16 days in the region and had crossed the Little Colorado River four times.

White-tailed Deer

Davis (1973) again noted the problem of species identification of mule and white-tailed deer, in that most frontiersmen simply referred to deer. Observers in two boundary

(United States and Mexico) survey parties reported both species of deer in New Mexico and Arizona. Both were abundant in 1851, especially in the mountain ranges along the border and along the Gila River.

Dr. C.B.R. Kennerly, with the Emory expedition in 1855, did not find many mule deer along the border except near the mountain ranges, including the Guadalupe in New Mexico. White-tailed, however, were common in all the mountains and stream valleys from El Paso, Texas, to Nogales, Ariz. They were especially abundant in the Guadalupe Range and other nearby mountains.

In the late 1850's, white-tailed deer were an important staple in the miners' diet, but populations eventually were reduced by hunting, at least locally.

Present Distributions and Habitats

Maps indicating present distributions in the Southwest are presented for mule deer (Fig. 2), elk (Fig. 3), and white-tailed deer (Fig. 4)³

Desert and Rocky Mountain Mule Deer

Desert mule deer (Fig. 2) are not distributed evenly throughout the desertscrub regions. The broad, relatively hot and dry, sparsely vegetated portions of the Mohave, Chihuahuan, and Lower Colorado River Subdivision of the Sonoran Desertscrub do not support deer because of lack of suitable cover and dependable forage and water. There is no historical evidence to suggest that deer ever inhabited these areas and, for the most part, there are no methods available to permit manipulation of these habits so they can support deer.

The Arizona Upland Subdivision of the Sonoran Desertscrub, however, provides some desert mule deer habitat. The additional cover provided in the form of higher, denser vegetation and rougher topography make this an important mule deer habitat throughout Arizona. Cover is generally adequate throughout the Arizona Uplands, but distribution and dependability of free water and food is still irregular—thus, both may be considered limiting factors.

Desert mule deer are more evenly distributed in the Semidesert Grasslands than in desertscrub communities. There are local exceptions, however, because of topography, degree of shrub invasion, and distribution of water. Deer are more prevalent in areas that have been invaded by mesquite and other woody shrubs. The grassland expanses that have not been subject to this invasion do not harbor desert mule deer unless other factors, such as rugged topography or riparian growth, are evident.

The isolated mountain ranges in southeastern Arizona are often considered white-tail habitat. Desert mule deer do, however, occur throughout the life zones present, although in relatively low numbers.

Riparian habitats in the South make up only a small proportion of the total area, but their importance is much greater than indicated by their size. Desert mule deer are restricted almost entirely to riparian habitats in the Mohave, Sonoran, and Chihuahuan deserts, particularly those associated with unaltered portions of the Colorado, Gila, Pecos,

³All distribution maps were prepared with the assistance of the Arizona and New Mexico Game and Fish Departments. Information for Indian lands was provided by the following individuals: Hualapai, William H. Beck; Papago, Charles Whitfield; White Mountain Apache, Kenneth Harper; and Navajo, John E. Antonio, Sr.

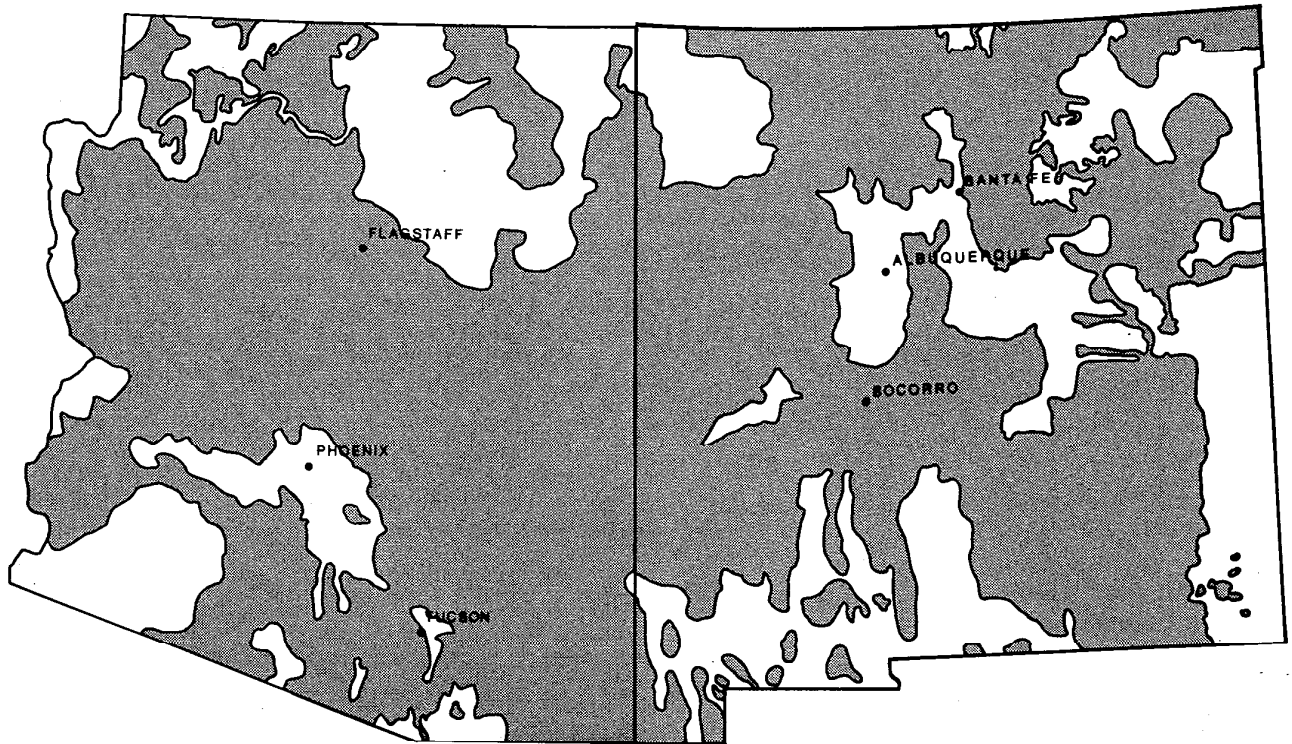


Fig. 2. Distribution of mule deer in Arizona and New Mexico.

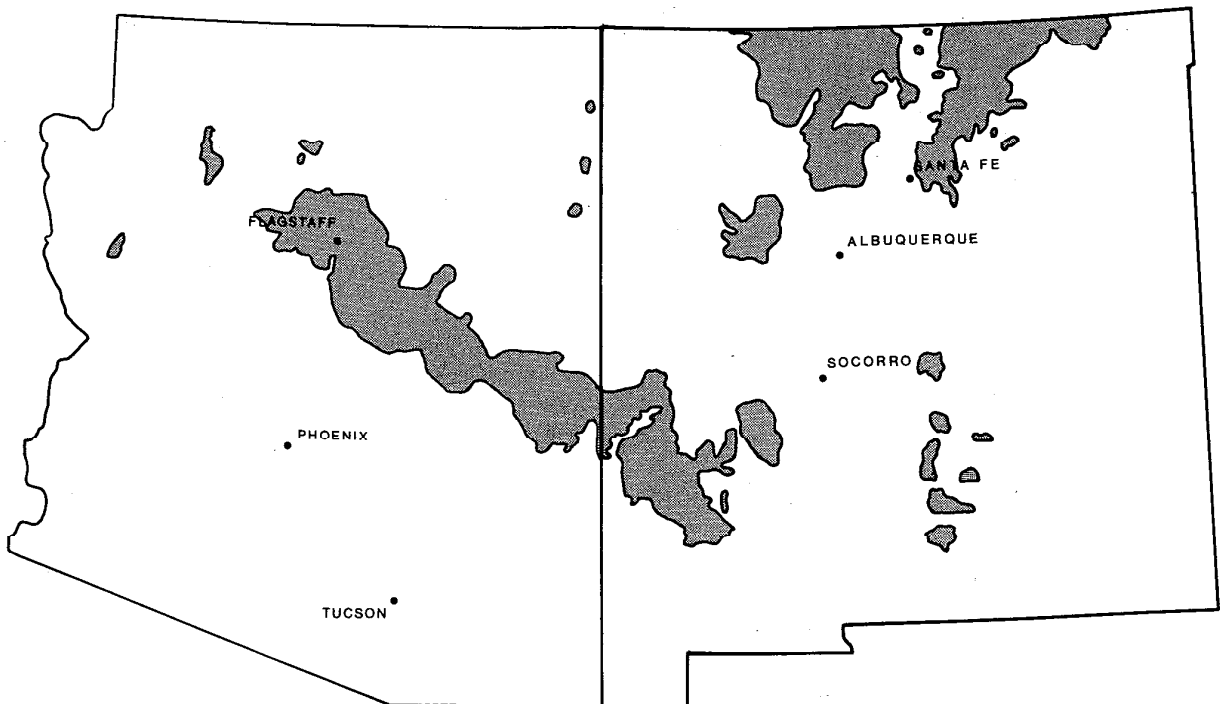


Fig. 3. Distribution of Rocky Mountain elk in Arizona and New Mexico.

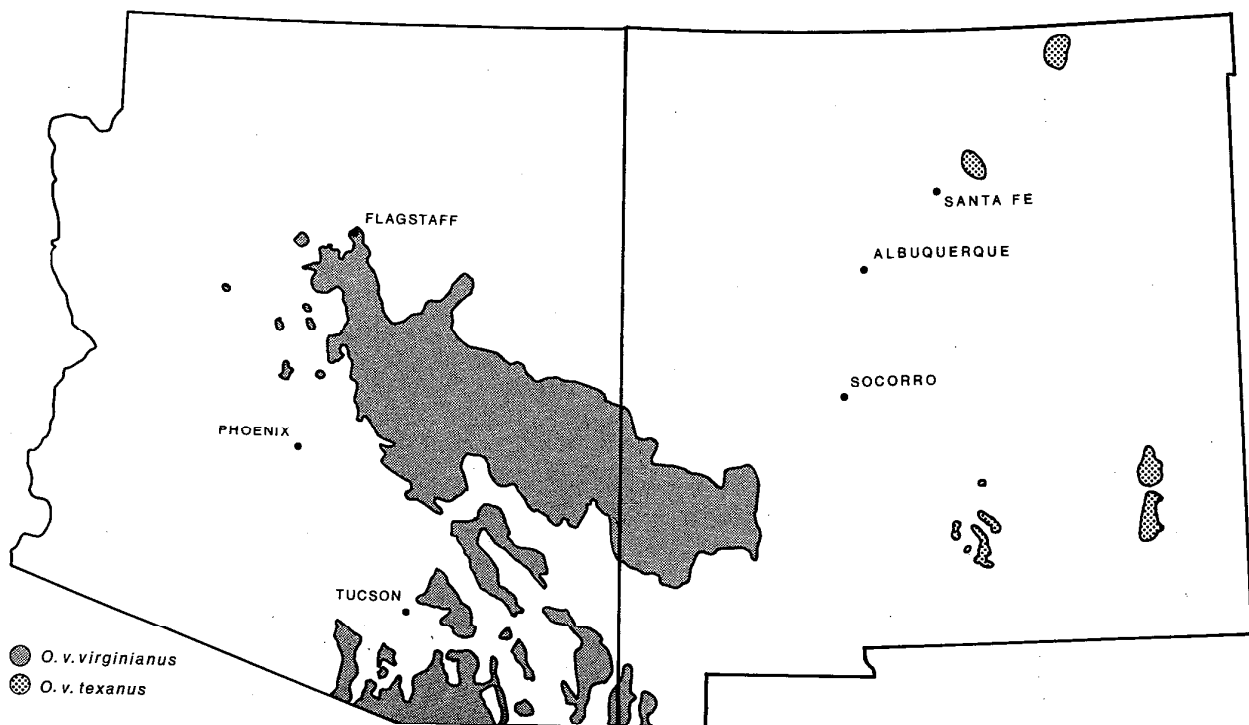


Fig. 4. Distribution of Coues and Texas white-tailed deer in Arizona and New Mexico.

and Rio Grande rivers. Deer use adjacent uplands in all cases but would be scarce in the desert were it not for these and other riparian habitats. Changes in riparian habitats, primarily those associated with dam construction, channelization, and land clearing for irrigation agriculture, have altered most habitat on larger streams. Mule deer, except for an occasional transient, are not found along the Gila River or portions of the Colorado and Rio Grande rivers where they were reported common in the 1800's.

Desert mule deer are more common in the southeastern portions of the Interior Chaparral, with Rocky Mountain mule deer dominating the northwest portions (U.S. Department of Agriculture Forest Service 1975). Mule deer utilize chaparral all year; there are no indications that these herds are migratory, but they may, like deer in the South, shift home ranges during different seasons (Swank 1958).

The Rocky Mountain subspecies is found in all biotic communities within the Mountains and Associated Region. The Great Basin Conifer Woodland provides excellent habitat for Rocky Mountain mule deer (Short and McCulloch 1977). Pinyon-juniper is used as a year-long range in many portions of the two-state area, but its greatest value is as winter range for those populations summering at higher elevations.

Montane and Subalpine Conifer Forests serve almost exclusively as summer range for mule deer, although some year-long residents may be found at extreme lower elevations in the montane zone. Use of subalpine forests is relatively greater near natural or created mountain meadows (Reynolds 1966a) or seral stages such as quaking aspen (Reynolds 1969a). Mule deer may be found occasionally in Alpine Tundra, although they rarely venture above the alpine-subalpine ecotone. Adult males make up most sight-

ings at higher elevations (Thilenius 1975).

Rocky Mountain mule deer are abundant only in scattered locations throughout the North. They are common along portions of the Little Colorado River north of Springerville, Arizona. There have been significant increases of deer since the early 1900's in riparian habitats and river breaks associated with major streams, such as the Canadian and upper Pecos rivers and their tributaries in northeastern New Mexico. Reasons for population expansion in such localized areas appear to parallel similar situations in other portions of the Great Plains, as discussed by Severson (1981).

Mule deer are not numerous in northeastern Arizona and northwestern New Mexico. Reasons for the scarcity in this area of any wild ungulates, including pronghorn antelope, are varied. Most of the territory is composed of Indian lands which are peopled by relatively sedentary tribes, the Navajo and Hopi. Both tribes run sheep, rather than cattle as is the case in eastern New Mexico. Hunting regulations were also far more liberal on reservation lands than in other areas of the Southwest. These circumstances have led some investigators to attribute a lack of large wild animals in this region to a combination of habitat changes through more intensive land use and unrestricted hunting (Ligon 1927, Bailey 1931, Hoffmeister 1962).

In forested areas within the reservations, as the Chuska Mountains, mule deer are common, possibly due to the additional cover available to the animals.

Coues White-tailed Deer

Coues white-tailed deer distribution is centered in the Sierra Madre, Mexico, and extends northward into the United States only in Arizona from San Francisco Peaks

southeastward along the Mogollon Rim and on the isolated peaks of the southeastern part of the state and extreme southwestern New Mexico (Fig. 4). In both states, population densities are highest in the Madrean Evergreen Woodland (Knipe 1977), which has been described as "the ecological metropolis" of the Coues deer (Lowe and Brown 1973). Riparian deciduous forests, particularly those above 3,500 feet, are also important to white-tails.

Although a few deer have been observed in the Arizona Uplands Subdivision of the Sonoran Desert, as Mt. Ajo and the Batamote Mountains, this is not an important habitat. The importance of Semidesert Grasslands is greatest adjacent to the Madrean Evergreen Woodlands, particularly where thickets of ocotillo are present for escape cover (Knipe 1977).

Coues deer north of the Gila River are generally found in lower densities than those in the south. Although they are found in the Interior Chaparral from the Mazatzal Mountains eastward, they are still apparently keyed to islands of oak woodlands containing Arizona and Emory oak. Distribution of Coues deer is continuous in the pinyon-juniper and ponderosa pine forests along the Mogollon Rim, but populations are at relatively low densities (Knipe 1977).

Texas White-tailed Deer

This subspecies is found in several scattered locations throughout eastern New Mexico (Fig. 4) but is common only in the Sacramento Mountains. Remnant populations occur in the Sangre de Cristo Range, one northeast of Santa Fe, the other southwest of Raton. In all of the above areas, these white-tails are most commonly associated with heterogeneous habitat types in pinyon-juniper and ponderosa pine.

A race of Texas white-tails, the sandhill deer, is found in southeastern New Mexico. These deer are associated with Plains Grasslands with a scattered, clumped overstory of Havard oak. Although this race has been given total protection for many years, the population is decreasing (Raught 1967).

Rocky Mountain Elk

Rocky Mountain elk reintroductions into both states have resulted in populations in virtually all areas which were historically occupied by Merriam or Rocky Mountain elk (Fig. 3). Two introductions—one into the Hualapai Mountains south of Kingman, Ariz., and another on the San Carlos Reservation near Cutter, Ariz.—have resulted in stable populations where elk probably never existed.

There is no concrete evidence that elk ever have been present on the Kaibab Plateau, Chuska Mountains, or on the isolated mountain ranges south of the Gila River. Although Davis (1973) indicated the presence of elk on the latter areas, his deductions were based on historical accounts of elk hides possessed by local Indians, not on actual sightings. These skins could have been obtained by trading with hunting tribes from the Rim country and, by themselves, do not constitute adequate evidence for the presence of elk.

Great Basin Conifer Woodlands are primarily used as winter range and the montane and subalpine coniferous forests as summer range. Elk, however, tend to winter and summer at somewhat higher elevations than do mule deer. Summer concentrations are more evident in mixed conifer and subalpine forests; winter distributions are closer to the

pinyon-juniper ponderosa pine ecotone. Bulls will often summer in the Alpine Tundra, at and above timberline, in sheltered areas such as cirque basins (Thilenius 1975). Use of the subalpine spruce-fir zone is dependent on availability of open meadows for feeding, although dense spruce-fir thickets are essential for cover.

Factors Affecting Habitat Quality, Quantity, and Use

Water

Whether or not deer or elk need free (surface) water is debatable. Bailey (1931), for example, attributed relatively higher populations of mule deer in southwestern New Mexico to their ability to live long distances from water, which gave them protection from hunters and concentrations of livestock. He felt they obtain necessary moisture from such plants as agave, yucca, and sotol. Coues white-tailed deer along the northwest coast of Sonora, Mexico, apparently survive on moisture available in succulent vegetation, because surface water is seldom available. In eastern Sonora and southern Arizona, however, these white-tails are less tolerant to aridity and more dependent on free water (Knipe 1977).

Other researchers have stated that an adequate water supply is an essential ingredient of all deer ranges (Dasmann 1971) or that deer require an available source of water to survive (Wood et al. 1970).

The presence of free water appears to influence mobility and distribution of desert deer (Welch 1960, Truett 1972). Hanson and McCulloch (1955) further note that watering sites seem to be the center of the home range when the site is surrounded by suitable habitat. Both mule deer and Coues white-tailed deer tend to restrict their movements to the vicinity of free water. Several investigators have indicated that deer remain within 1-1/2 miles of water during the dry seasons (Hanson and McCulloch 1955, Swank and 1958, Wood et al. 1970, Knipe 1977) although Rodgers (1977) indicated they maintain the same range during wet seasons. Clark (1953) noted that pregnant does tend to remain within 1/4 mile of water.

Elder (1956) may have developed the correct hypothesis when he noted that although deer in desert habitats can survive on moisture from succulents, this may be a marginal condition that does not permit anything beyond survival. Although specific information on deer is not available to test this theory, Hutchings (1958) determined that domestic sheep which receive fresh water every day graze a greater variety of forage, especially drier materials, and utilize the range more uniformly than animals which are allowed to become thirsty. Further, he noted that sheep watered every day gain weight whereas those watered every third day lose weight. If we assume that the requirements for deer are similar to those of sheep and that weight gain or loss is indicative of animal health and productivity, the need for free water in hot, dry seasons becomes evident. Hutchings' (1958) findings imply that animals under moisture stress are forced to spend their grazing time seeking only those plants that are capable of alleviating this stress. Conceivably these may not always be the plants containing other nutrients needed by the animals. Thus, a wildlife manager interested in maintaining healthy and productive deer herds in lieu of those that are only able to survive, should consider develop-

ing free water sources in areas where such sources are limited. Water developments will be further discussed in the section on Range Improvement Practices.

Cover

Along with food and water, cover has long been recognized as one of the three main components of wildlife habitat. Leopold (1933) defined cover rather simply as anything that provides shelter for game. The primary functions of cover are to provide shelter from weather and escape routes and hiding places as security from predators and hunters. Cover has been more recently divided into two types based on these functions (Black et al. 1976; Thomas et al. 1976, 1979a). Hiding cover provides the security that makes an animal's use of an area possible, and thermal cover aids the animal in maintaining body temperatures within tolerable limits. Such cover is used for protection from heat or cold by intercepting incoming radiation or by preventing radiational loss from the animal to the open sky.

Another concept closely allied with cover is edge, which has been defined by Thomas et al. (1979b) as the place where plant communities meet or where successional stages or vegetative conditions within the plant community come together. The importance or function of edge can be best described by considering two concepts summarized by Thomas et al. (1979b). These concepts or laws recognize the effects of dispersion and interspersions. Dispersion describes patterns of distribution in a population; interspersions describes the intermixing of plant species and communities that provide habitat for animals. The law of dispersion, first postulated by Leopold (1933), states: "The potential density of game (wildlife) of low radius requiring two or more types is, within ordinary limits, proportional to the sum of the type peripheries." Although the phrase of low radius implies animals that have small home ranges, later research has demonstrated that the larger mammals with larger home ranges also use edges and ecotones disproportionately more than other habitats (Thomas et al. 1979b). Reynolds (1966b) documented this where there is edge between open (feeding) areas and forested (cover) areas.

The law of interspersions states that the number of species requiring two or more types of habitat depends on the degree of interspersions of numerous blocks of such types.

Thus, when considering both laws, more edge of particular types means more individuals of the species associated with that edge, and the edge effect can be increased by added interspersions of types (Thomas et al. 1979b).

Edge can be quantified in a form related to area by using a diversity index (*DI*) developed by Patton (1975). This index compares perimeter and area by using the ratio of the circumference to area of a circle which is given an index value of 1. Any index larger than 1 is a measure of irregularity and can be used as the *DI*. The formula for habitat diversity is

$$DI = \frac{TP}{2 \sqrt{A \cdot \pi}}$$

where *TP* is the total perimeter around the area plus any linear edge within the area *A*. The *DI* allows a manager to compare different shapes of openings that could be created to improve wildlife habitat. Two modifications of Patton's formula to differentiate between inherent and induced edge have been developed. Inherent edges are site-related and occur where plant communities meet, whereas induced

edges occur where successional stages within a plant community come together (Thomas et al. 1979b).

The amount of cover that large ungulates require for protection from predators and hunters can vary considerably in space and time. Black et al. (1976) and Thomas et al. (1979a) recommended an optimum ratio of 40% cover to 60% foraging area. This suggestion was based on how elk and deer used cover and openings in relation to edge. The 60% forage area includes openings and all forest areas that do not qualify as cover. However, in predicting animal response to alterations of cover/forage ratios, Thomas et al. (1979b) made several assumptions, one of which was that all untreated natural forest sites were potential cover, regardless of their present function.

Hiding or Security Cover

The 40% cover is composed of hiding and thermal cover. Thomas et al. (1979a) and Black et al. (1976) define hiding or security cover as vegetation capable of hiding 90% of an elk from the view of a person at 200 feet or less. This actual distance, called sight distance, varies from stand to stand. Thomas et al. (1976) and Black et al. (1976) stated that for optimum effect hiding cover should be from four to eight distances wide. They estimate a width of four sight distances to be the minimum at which elk would be out of sight from all directions. The maximum width of 8 chains provides necessary cover without leaving a little-used central area. This is based on Reynolds' (1966b) findings that use of cover by elk declines significantly beyond 900 feet from an opening. Hiding cover that meets the requirements for elk will be more than adequate for deer, because deer are much smaller than elk (Black et al. 1976). On ranges where elk are not a management concern, cover criteria for deer would be developed as described above. Shorter sight distances derived from smaller animals would result in smaller minimum areas.

Although topography certainly contributes to security and escape cover, the complexity of combinations of vegetation and cover make it difficult to quantify. Cover type planning must consider topographic features on a site-specific basis.

Thermal Cover

Thermal cover for elk has been defined by Thomas et al. (1976) as a stand of coniferous trees 40 feet or more tall with a canopy cover exceeding 75%. Optimum size on summer range varies from 30 to 60 acres. Areas less than 30 acres do not provide adequate protection from wind, and areas greater than 60 acres do not receive maximum use in the core. Stands with multi-storied cover are better than those with single-stories because of great stability.

There is some latitude in providing thermal cover for deer as compared to elk because deer are able to use lower vegetation. Black et al. (1976) defined thermal cover for deer on summer and spring-fall range as trees or shrubs, coniferous or deciduous, at least 5 feet tall with a 75% crown closure on a forest stand of at least sapling size with 60% crown cover. Optimum size of deer thermal cover areas is 2-5 acres, with a minimum width of 300 feet.

Ward (1980) suggested that needs for thermal cover apparently vary, not only with season (as would be expected), but also with location, number of animals, and animal behavior. He noted that thermal cover areas consistently used by elk on southcentral Wyoming summer ranges are smaller than those suggested by Thomas et al. (1979a).

They are only 50 yards from openings and cover less than 5 acres.

Presently, designers of thermal cover use mostly theoretical criteria. Relationships between weather conditions and the energy needs/demands of wild species are only hypothetical, based largely on generalized responses of animals to certain weather conditions or from comparison of weather regimes under different habitat conditions. Such interactions are extremely complex. Comparison of biological data with weather data does not answer the question. What is needed is knowledge of thermal interaction which, to be ecologically meaningful, involves understanding the metabolic, nutritive, and behavioral characteristics of the organism (Moen 1973). Recommendations on thermal cover requirements are subject to change and must be carefully considered.

Deer populations in many areas of the Southwest face environmental extremes that are entirely different from those considered in the pioneering research work on animal-weather relationships. Although cold weather is a problem in parts of the Southwest, a greater area is characterized by extremely hot temperatures and low precipitation.

Most deer studies on desert scrub or Semidesert Grassland areas mention high temperatures and low precipitation as important climatic factors affecting deer behavior (Truett 1972, Anthony 1976, Rodgers 1977). However, no studies have attempted to quantitatively document thermal cover requirements for deer during hot weather (Fig. 5). Truett (1972) stated that desert mule deer tend to avoid sunlight on warm mornings by feeding on north- or west-facing slopes in dense vegetation and then bedding in deep shade. Rodgers (1977) noted, on an area of more uniform topography, that desert mule deer seek shelter in washes from the heat and sun during midday. He maintained that these dry washes support a more mature and dense plant community dominated primarily by mesquite.

Special Cover Requirements

Special cover for fawning or calving may also be necessary, although its exact role is presently unknown. Black et al. (1976) described elk calving areas as gently sloping, with a mixture of tree cover, shrubs, and succulent forage within 1,000 feet of water. Fawning cover was described as areas 1 to 5 acres in size, with an approximate 50% tree crown cover,



Fig. 5. *Coues* white-tailed deer habitat in the ecotone between Semidesert Grasslands and Madrean Evergreen Woodlands. Thermal cover needs for deer have not been determined for these hot, dry environments.

containing low shrubs or small trees from 2 to 6 feet tall. Vegetation must be plentiful and succulent and within 600 feet of water.

Thomas et al. (1979a) stressed that the optimum cover requirement was 40% of the total land surface, not just 40% of the forested area. Optimum mixtures of cover types have also been suggested for elk on summer and spring-fall ranges as being 20% in hiding cover, 10% in thermal cover, and an additional 10% in either hiding or thermal cover. Recommendations for deer were similar except in the last case where 5% was fawning cover and the remaining 5% was hiding, thermal, or fawning cover.

Precise definitions and quantification of cover types required by deer inhabiting very hot climates should be given some research priority. On many such ranges, particularly in the Interior Chaparral and mesquite-invaded Semidesert Grasslands, brush control efforts for livestock range improvement are popular programs. Wildlife habitat needs are generally overlooked because information is lacking on optimum cover/forage ratios or edge requirements for most wild species, including deer. Summer cover requirements for deer are more critical in hot climates than in cooler climates such as those found at higher elevations. Information is needed on effectiveness of size, shape, and patterning of brush control programs to allow managers to incorporate design features that meet the needs of wildlife species.

Requirements for thermal and security cover considered thus far have concerned only summer ranges. Deer and elk generally are distributed over large areas on these ranges, and concentrations of animals do not approach the magnitude evident on many winter ranges. Increasing snow depths at higher elevations force many elk and deer in the Mountains and Associated Areas into restricted regions at lower elevations in mid to late winter. Although Rocky Mountain mule deer herds in the Plains Grassland of New Mexico are not migratory, requirements for winter cover, particularly thermal cover, become critical in winter because of relatively severe weather conditions.

Because of these circumstances, general recommendations intended to apply to all winter ranges cannot, with existing knowledge, be developed. As Thomas et al. (1979a) pointed out, the consequences of any errors would be greatly magnified. Furthermore, they also stated that each winter range is different in its vegetative mosaic and how it is used by wintering animals. Each winter range must be considered separately before any decision is made on cover alterations—particularly alteration of thermal cover.

Wildlife habitat managers, then, have four facets of vegetation manipulation to consider when designing habitat modifications:

1. The amount of hiding and thermal cover necessary to fulfill the animals' needs,
2. The amount of area needed for food production,
3. The optimum arrangement (interspersed) of various cover and food producing areas to realize,
4. The optimum amount of edge.

Food

Knowledge of foods consumed by wild cervids, relative to availability of forages, is basic to most management decisions concerning habitat. Accordingly, food habit studies have been conducted throughout North America. Results

are applicable only to localized situations, however. Preference or selectivity by the animal is based largely on relative availability, which varies significantly in space and time (Petrides 1975). Hence, habitat management situations often develop where such information is lacking or outdated. Elk food habits, for example, have received considerable attention in certain localities (Kufeld 1973), but, as discussed later, very little work has been done in the Southwest. Studies on mule deer habits have also received considerable emphasis throughout North America (Kufeld et al. 1973). Although there is no information available for the northern desert and grasslands, some information is available for other parts of the Southwest, especially the pinyon-juniper type.

Methods for Determining Food Habits

Primary methods of obtaining food habits are examination of material found in digestive tracts (rumen contents and fecal material), observation of wild animals, observation of tame or confined animals, and measurement of the use of forage plants. The most widely used technique is analysis of rumen contents. This method has been reviewed by Medin (1970) and Ward (1970). Although this method has been accepted by many investigators, it does have significant limitations. Collection of material is made by killing the animal, thus permitting only one sample per animal. Variations in rumen contents is related to individual preferences, time since eating and drinking, social order, and such activities as rut (Short 1966). Such variability indicates a necessity for a large number of samples for quantitative analyses. Anderson et al. (1965), stated that they did not collect sufficient samples for any single season or year but that their estimates of major food items consumed (based on the contents of 93 rumens) had 95% confidence limits of $\pm 15\%$. Another problem with rumen content analysis is that forage items have differential rates of digestion because of fiber content and degree of lignification. Thus, succulent leaves of shrubs, forbs, and grasses are digested faster than woody twigs. Succulent materials are also ground up more quickly during rumination (Bergerud and Russell 1964, Rice 1970). Microscopic point techniques, where the entire sample is ground up and plant species identified by epidermal cell characteristics, apparently alleviate this problem to the satisfaction of many investigators (Ward 1970).

Analysis of fecal material involves drying and grinding samples and identifying forage species microscopically by epidermal cell characteristics. Attempts to verify its accuracy have been varied (Stewart 1967, Zyznar and Urness 1969, Todd and Hansen 1973, Anthony and Smith 1974). Others have documented differences in proportions of forage items when comparing fecal analyses with known diets but have corrected data using regression techniques (Dearden et al. 1975) or have noted similarities when individual species within the diet were ranked from most common to least common (Vavra et al. 1978). These variable findings suggest that determination of diet by analysis of fecal material is subject to limitations similar to those encountered with rumen content analysis.

Some knowledge of food habits of wild animals has been obtained by observations which involve feeding-minutes or bite-count techniques. Feeding-minutes is the length of time spent grazing or browsing each forage species in a mixed stand. The bite-count method is similar; however, instead of recording time spent grazing each species, the number of

bites is recorded (Wallmo and Neff 1970, Bjugstad et al. 1970). Although conceptually simple, this method is difficult to use under normal field conditions. Proper identification of plants being utilized is essential. The difficulty of getting close enough to feeding animals to make proper identification often results in bias towards taller, more easily identified plant species. Wallmo et al. (1973) determined that the feeding-minutes method attributed more use to shrubs and less use to grasses and forbs than actually occurred, and identifiability of individual species varied with plant size, distance, and observer. They noted that an observer would have to be within 75 feet of the animal to correctly identify more than 80% of the species they were grazing.

Many of these problems can be circumvented by using direct observations of tamed animals (Wallmo and Neff 1970, Wallmo et al. 1973). The observer can be positioned close enough to the animal to record use not only on all species but even plant parts being used and to collect specific unknowns for later identification. Another advantage of this method is that observations can be directed towards specific interests such as vegetation types, vegetation treatments, and seasons. Disadvantages include: (1) Quantifying intake is difficult. The method works well for determining what kinds of forage are being utilized but not amounts. (2) Variability between animals makes it difficult to make predictions relevant to populations. (3) It's an expensive technique involving time and effort to obtain, raise, and train the animals. Special physical facilities are also needed. (4) The degree of similarity of forage selection between tame and wild animals is not known (Wallmo and Neff (1970). Neff (1974) later concluded that use of tame deer is a valid research technique because the artificial environment of their rearing and training does not appear to significantly alter their instinctive forage preferences. This was partially reinforced by Regelin et al. (1976), who noted that tame mule deer fed unlimited amounts of concentrated feed select the same forage species in a similar proportion as tame deer kept on native range with no supplemental feed.

Esophageal and rumen fistula techniques have been used on livestock for both food habits and nutritional studies for many years. Although limited use of fistulation has been made on some tamed, native ruminants, particularly white-tailed deer, advantages of obtaining food habits from direct observations has negated their use in this respect. Advantages of esophageal fistulation may not offset the difficulty of installation and maintenance. However, forage collections from esophageal fistulated animals may provide a method of quantifying intake provided the size of the opening is adequate and the sample is not contaminated by regurgitated material. Reviews of these methods are available (Van Dyne and Torrell 1964, Rice 1970).

A few studies have been conducted where varying arrays of native plants species have been offered to penned animals in a "cafeteria style" arrangement. Nichol (1938) used this approach to study foods of Coues white-tailed deer and Rocky Mountain mule deer in Arizona. Although this is potentially the best way to quantify intake as related to availability, arrays of forages are generally artificial; therefore, applicability to natural situations is questionable. Although it would be possible to make offerings in the same proportions that they occur in nature, methods of presenting them to study animals would remain artificial. Other disadvantages include the necessity for specialized holding

facilities and substantial time requirements for maintaining the animal and collecting the forage.

Methods of quantifying food habits of wild animals by direct measurement or estimation of utilization on plants are available but require considerable time. Most of these methods, as reviewed by Martin (1970), were developed for use with livestock and are based on intensive pasture studies. Feeding habits of wild ruminants are generally such that it is difficult to detect use by examining the plants. Wild ruminants tend to move more while feeding and to take less of each plant than do domestic animals. It is also difficult to detect use of certain food items, such as individual leaves on shrubs, certain fruits, mosses, and lichens. Cases where animals consume the entire plant, such as a mushroom, are also difficult to measure. A case has been documented where field utilization studies revealed little evidence of a shrub being browsed, but rumen analysis showed that it contributed significantly to deer diets (Harlow 1979). Another serious limitation imposed by measuring use directly from plants is that it is often impossible to assign use to any one species because several herbivores may be present. Although such data are useful for determining common use impacts on an area, they provide only supplementary data concerning food habits of individual species.

None of the previously described methods of determining food habits of ruminants is without disadvantages. Most are limited by time and special facilities which may not interfere with research programs but would be important to resource managers. Probably the best method presently available to managers would be microscopic analysis of fecal material. Although subject to some of the same limitations as rumen content analysis, collection of sample material would be much easier and the legal ramifications of obtaining samples, other than during hunting seasons, would be avoided. Adequate sample sizes could be obtained and multiple collections from single animals would be possible under certain circumstances, such as by following radio-collared animals or following individuals after a recent snowfall. The time required to train personnel to use this method and to prepare reference slide material can be avoided by using facilities provided by service laboratories equipped to handle such analyses. Charges made by these laboratories are generally keyed to the total number of samples and the degree of accuracy requested by the investigator. Whatever method is chosen, it must be supplemented with a thorough vegetation survey to facilitate proper interpretation of results.

Mule Deer Food Habits

Kufeld et al. (1973) listed 788 plant species consumed by mule deer in 99 food habit studies conducted throughout the animals' range. In the Southwest, 12 food habit studies from Arizona and New Mexico and 2 from southwestern Texas revealed 327 plant species were consumed. Although mule deer utilize a large number of species, relatively few make up a large part of their diet. Hunt (1978), for example, noted that mule deer sampled 113 of the 194 plant species found on his study area, but only 10 were found in quantities equal to or greater than 1% of the deer's diet. He further determined that deer used 81, 63, 16, and 32 species during summer, fall, winter, and spring, respectively; but only 19, 16, 4, and 7 species made up more than 1% of the diet each. Similar patterns of plant species use were noted by Neff (1974).

Appendix Tables C1 and C2 list the most important forage

species for mule deer in the South and the Mountains and Associated Areas, respectively. This partial listing also provides a relative index to their importance. Appendix Tables C3 and C4 summarize most of the food habit studies done on mule deer in the South and the Mountains and Associated Areas, respectively.

Availability of forage species is a major key to their utilization, and considerable year-to-year changes in deer diets result from changes in availability. Forage availability depends on the number of herbivores, domestic and wild, which consume these seasonal crops, weather which alters growing conditions, and other long-term influences such as logging, fire, brush manipulation, and changes in range condition (McCulloch 1978).

Catclaw acacia,⁴ falsemesquite calliandra, California jojoba, kidneywood, and littleleaf krameria are the most preferred shrubs in the deserts and grasslands of the South, whereas hairy cercocarpus, cliffrose, and several oaks, especially Gambel, gray, and wavyleaf, are most abundant in diets of mule deer in the mountainous areas. Wright eriogonum is one of a few species that ranks high in both areas. Heavy use of evergreen browse, such as shrub live oak and junipers, has been presumed by many investigators to be due to necessity rather than preference; that is, it reflected a scarcity of other foods rather than a preference for items consumed (Anderson et al. 1965, McCulloch 1978). The same principle may account for heavy use of mesquite, excepting fruits, and ponderosa pine. McCulloch (1973) considered such species important, however, because of their stable supply in dry years and during dry periods of normal years. Juniper intake has probably been excessive in some areas, although it may be a good food for deer at lower concentrations in the diet (McCulloch 1978). In this respect, juniper may be comparable to big sagebrush. Both have high crude fat contents which contain disproportionately higher amounts of essential or aromatic oils than most other plants. Big sagebrush was the most important winter browse species (of those tested) in northern Colorado, based on nutritive content and digestibility, but mule deer fed rations with a high proportion of big sagebrush lost weight (Dietz et al. 1962). They concluded that mixtures of browse species are necessary to supply mule deer with adequate amounts of nutrients.

Several browse species contain compounds that may influence their palatability. Some may also affect the ruminants' physiological processes. These will be discussed in the section on nutritive contents of forages.

Utilization of fruits, particularly mast and cactus fruits, was emphasized by McCulloch (1973), Short (1977), and Hunt (1978). McCulloch (1973) also noted that many of the least preferred browse plants were important because of their fruit crop.

The relatively small number of forb species listed as being important to mule deer in Appendix Tables C1 and C2 is not indicative of the value of forbs as a group. The large number available to deer coupled with infinite combinations of aggregations precluded notice of individual species. The importance of forbs in the diet is, however, documented in Appendix Tables C3 and C4. The high use of forbs during spring and summer is not unusual or unexpected; however,

⁴Common and scientific names of plants are listed in Appendix B, and follow Nicker-son et al. (1976).

the proportion present in fall and winter diets, particularly on pinyon-juniper winter ranges, is greater than that found in deer diet studies from other areas. Neff (1974) stated that forbs were the major source of green forage (at Beaver Creek) during all seasons. Declines in forb use can be attributed to changes in availability, such as changes related to snow depth, drought conditions, or changes in physical structure.

Grasses become important in spring and early summer, generally only for short periods (Hungerford 1970, Neff 1974). Cool-season grasses, the first to initiate green growth, are often avidly sought by deer. Cool-season, annual bromes are important in the deserts and grasslands of the South, whereas perennial cool-season species such as bluegrass, orchardgrass, and wheatgrass are important in high-elevation areas. Hungerford (1970) noted that introduced, seeded grasses appeared to be more palatable than native species.

Although considered only a miscellaneous food item, fungi, particularly mushrooms, were noted as being used wherever available in most food habit investigations. Hungerford (1970) further noted that mushroom utilization increased during late summer. Hunt (1978) also noted mistletoes, considered miscellaneous because of their low availability to deer, to be used whenever available.

In general, mule deer preferences rank as follows:

1. Fruits, flowers, mushrooms,
2. New green herbage, particularly forbs and new leaves of deciduous shrubs,
3. New twigs and mature green herbaceous material,
4. New leaves and twigs of evergreen species, and
5. Mature leaves and twigs of evergreen species.

Coues White-tailed Deer Food Habits

A generalized breakdown of studies on food habits of Coues deer, including the area and habitat in which each was conducted, is presented in Appendix Table C5. Coues deer are largely browsers. The only significant departure from this trend appears from late summer to autumn when diets from three areas (Santa Rita, Chiricahua, and Mazatzal mountains) show a significant trend toward forbs. This trend was not evident in the other studies (Appendix Table C6).

Appendix Table C6 lists the more common species used by Coues white-tailed deer and gives a general indication of deer habitat, and over 600 of the most common are known to be palatable. Another 441 species are suspected to have forage value (Knipe 1977). This summarization indicates the most prevalent plants in the diet of Coues white-tailed deer to be hairy cercocarpus, Wright eriogonum, falsemesquite calliandra, littleleaf krameria, and junipers, primarily one-seed and alligator.

Patterns of use vary, as expected, depending on geographic area, elevation, and population pressures. McCulloch (1973), working in the Mazatzals in the Arizona Uplands-Interior Chaparral ecotone, found shrub fruits, primarily acorns from shrub live oak, to be the primary food item. Forbs and falsemesquite calliandra dominated early fall diets, and evergreen browse, chiefly from holly-leaf buckthorn, deerbrush ceanothus, desert ceanothus, and birchleaf cercocarpus, was important during other seasons.

The remaining studies were conducted further south, in and around the isolated mountain ranges of southeastern Arizona, and included portions of the Arizona Uplands, Semidesert Grasslands, Madrean Evergreen Woodland, and Montane Conifer Forest biotic communities. Nichol (1938) conducted penned deer studies with 168 plants and concluded that Wright eriogonum was the most important species. This was reinforced in part by Anthony and Smith (1977), who determined it was the most important species during cool, dry periods from February to April, but ranked second to kidneywood and second to hairy cercocarpus in the San Cayetano and Dos Cabezas mountains, respectively, in overall importance.

Leguminous browse plants—velvetpod mimosa, falsemesquite calliandra, and littleleaf krameria—were important forage plants at lower elevations (White 1961). Knipe (1977) was in general agreement although he included Wright eriogonum as equally important.

Variability was greater on high-elevation ranges. White (1961) stated forbs were primary diet items, whereas Day (1964) and Knipe (1977) considered hairy cercocarpus as most important. White (1961), however, thought high forb use was caused by the absence of preferred browse species. Day (1964) also tempered his findings by stating that forb fragments in rumen contents could not be identified because herbaceous species disintegrated rapidly in the deer's stomach.

Day (1964) also noted significant use of oak browse, primarily from Arizona white, Gambel, and netleaf oak, but attributed this, as well as use of juniper species, to severe overuse by deer of more desirable species.

Weather and growing conditions also affect preference. Ocotillo did not rank high as a forage plant; however, its rapid response to available moisture from summer rains produced a green forage (leaves) which was avidly sought by deer whenever available (White 1961). Mistletoes, especially of the genus *Phoradendron*, were very palatable but generally out of reach of feeding deer. The material they did consume came from branches blown down by high winds or stripped from trees by hailstorms (Knipe 1977).

The palatability of fruits, particularly acorns, to Coues white-tailed deer has been noted in several studies (Nichol 1938, Day 1964, McCulloch 1973, Knipe 1977). The importance of acorns to the diet of white-tailed deer has been noted in other parts of the country by Duvendeck (1962) and Severson and Kranz (1978). Although acorn crops are inconsistent, their value to many forms of wildlife indicates that forest management practices should be directed towards maximizing this crop (McCulloch et al. 1965).

Elk Food Habits

Little information is available on plant materials consumed by elk in the Southwest. Of the four sets of data collected, three were from rumen samples of hunter-killed elk taken from unspecified locations; we can only speculate whether they were from summer, transitional, or winter ranges. In the other study, Short et al. (1977), reported food habits during all seasons on pinyon-juniper range in southwestern New Mexico.

In the first study, Lang (1958a) listed species found in rumens of seven elk killed in the Gila National Forest in December 1952 as:

| Common names of food plants | Percent by frequency of occurrence | Percent by volume |
|-----------------------------|------------------------------------|-------------------|
| Grasses | | |
| Grass | 71.4 | 4.6 |
| Needlegrass | 28.5 | T |
| Total grasses | 85.7 | 4.6 |
| Weeds | | |
| Bahia | 28.5 | T |
| Amaranth | 14.3 | T |
| Globemallow | 14.3 | T |
| Mallow | 14.3 | T |
| Groundcherry | 14.3 | T |
| Glorybind | 14.3 | T |
| Total weeds | 57.1 | T |
| Browse | | |
| Cercocarpus | 100.0 | 32.8 |
| Wright siltassel | 85.7 | 36.0 |
| Oak | 100.0 | 16.1 |
| Juniper | 57.1 | 9.6 |
| Saltbush | 71.4 | T |
| Pricklypear | 71.4 | T |
| Skunkbush sumac | 14.3 | T |
| Pine | 28.5 | T |
| Sagebrush | 14.3 | T |
| Total browse | 100.0 | 95.3 |

Some of the plant species present (aspen and spruce) indicate these animals were taken at higher elevations, presumably summer range.

Arizona Game and Fish Department personnel analyzed rumen contents from 82 elk taken during the hunting season in the Apache-Sitgreaves National Forest in November 1967.

| Common name | Percent of rumens in which item occurred |
|-------------------|--|
| Grasses | |
| Sedges | 1 |
| Weeds | |
| Orange sneezeweed | 16 |
| Mullein | 6 |
| Thistle | 1 |
| Agoseris | 1 |
| Grassleaf peavine | 1 |
| Browse | |
| Quaking aspen | 72 |
| Ponderosa pine | 66 |
| Mistletoe | 41 |
| Eriogonum | 10 |
| Douglas-fir | 9 |
| Willow | 7 |
| Gambel oak | 7 |
| Creeping mahonia | 5 |
| Blue spruce | 4 |
| New Mexico locust | 2 |
| White fir | 1 |

Based on species utilized, these elk were apparently killed at lower elevations, probably on winter range. Another study by Lang (1958a) was made in the Pecos Wilderness in the southern Sangre de Cristo Range in 1957 and was based on rumen contents from four bull elk collected in October:

| Common names of food plants | Percent of frequency of occurrence | Percent by volume |
|-----------------------------|------------------------------------|-------------------|
| Grasses | | |
| Green grass | 100.0 | 70.3 |
| Dry grass | 50.0 | 6.3 |
| Wheatgrass | 50.0 | T |
| Total grass | 100.0 | 76.6 |
| Weeds | | |
| Forbs | 100.0 | T |
| Western yarrow | 50.0 | 1.5 |
| Mariposa | 50.0 | T |
| Mushroom | 50.0 | T |
| Bedstraw | 25.0 | T |
| Thistle | 25.0 | T |
| Total weeds | 100.0 | 2.3 |
| Browse | | |
| Quaking aspen | 75.0 | 20.0 |
| Grouse whortleberry | 75.0 | 1.3 |
| Myrtle pachistima | 25.0 | T |
| Western thimbleberry | 25.0 | T |
| Spruce | 25.0 | T |
| Total browse | 100.0 | 21.3 |

These results indicate foods consumed at high and intermediate elevations. Although collected in late fall, the apparent mild winter of that year resulted in hunters being distributed over higher portions of the unit. Lower elevation juniper ranges were not hunted because elk were still in high country (Arizona Game and Fish Department 1968).

The percent of times an item appears in a sample of rumens is not necessarily indicative of the amount of that item consumed. Note that in Lang's (1958a) first study there were grasses in 85.7% of the rumens examined, but the average percent by volume was only 4.6. The disproportion is similar for forbs in his second report.

Table 2 is a summary of rumen contents of 43 elk collected throughout the year from lower elevation pinyon-juniper ranges in southwestern New Mexico (Short et al. 1977). This is the only comprehensive elk food habits study that has been done in the Southwest.

Although the Great Basin Conifer Woodland is considered primarily as a winter range, in the Southwest it also provides valuable summer range for elk (Reynolds 1964). Elk that summer on these pinyon-juniper ranges apparently consume a much higher proportion of shrubs (Table 2) than do elk summering at higher elevations in Colorado. Kufeld (1973) reviewed only two other studies that documented comparatively high summer shrub use, one in Idaho and the other in Montana.

Winter shrub use varied from 83% to 95%. These results are comparable to those found in other parts of North America (Kufeld 1973).

Table 2. Contents of 43 elk rumens collected throughout the year at Fort Bayard, New Mexico. Composition recorded is percent of intercept points determined from the application of the point frame technique. (Short et al. 1977)

| Group Plant | Winter | Spring | Summer | Autumn |
|----------------------------------|--------|--------|--------|--------|
| Forbs | | | | |
| Dalea ¹ | 0 | 0 | 4 | 4 |
| Birdbill dayflower | 0 | 0 | 8 | 0 |
| Common hoarhound ¹ | 5 | 1 | 0 | 0 |
| Tumbleweed amaranth ¹ | 0 | 0 | 4 | 0 |
| Others | 0 | 2 | 1 | 2 |
| Total forbs | 5 | 3 | 17 | 6 |
| Browse | | | | |
| Hairy cercocarpus | 17 | 39 | 27 | 25 |
| Oaks ² | 28 | 25 | 13 | 18 |
| Junipers ³ | 34 | 0 | 1 | 13 |
| Wright siltkassel ¹ | 3 | 9 | 12 | 14 |
| Soap tree yucca ⁴ | 0 | 7 | 1 | 0 |
| Mistletoe ⁵ | 1 | 0 | 3 | 0 |
| Others | 1 | 3 | 1 | 4 |
| Total browse | 84 | 83 | 58 | 64 |
| Grass | 4 | 7 | 19 | 19 |
| Unidentified materials | 7 | 7 | 6 | 11 |

¹Leaves and stems.

²Leaves, stems, and fruits.

³Leaves and fruits.

⁴Flower stalks

⁵Stems.

Nutritional Relationships

Food habits and nutrition are both directly and indirectly associated with cervid productivity. The direct effect of malnutrition is starvation, which is relatively easy to predict or to observe. Indirect effects are many and varied. They tend to work through such vectors as parasites, diseases, weakness, and susceptibility to predation, failure to conceive, reabsorption of fetuses, or reduced ability to digest foods (Dietz and Nagy 1976). Indirect effects of malnutrition are seldom obvious, and it is difficult to separate their influences from other environmental factors. The relationships between fawn survival and nutritional status of habitat serve to illustrate such a case. Fawn condition depends on the diet of the female both during gestation and after parturition. Verme (1963, 1965) and Murphy and Coates (1966) determined for white-tailed deer that when the nutritional level of the diet of pregnant females was lowered, fetal growth was retarded and mortality of fawns increased. Thorne et al. (1976) and Robinette et al. (1973) noted similar patterns for elk and mule deer, respectively. Weights of fetuses from malnourished white-tailed does were not only lower, but fetuses high lighter eye lenses, lighter thyroid and thymus glands, and heavier adrenal glands (Verme 1979), any or all of which could be contributing in some unknown way to indirect mortality.

A slight or gradual reduction in nutritional levels over a period of years could be caused by successional changes, overgrazing, drier weather patterns, increased tree growth, or many other factors. The resultant effects would be equally gradual: a slight, but steady reduction in the number of fawns reaching productive age classes which, over a period of years, would become very crucial.

Resource managers, therefore, have a greater responsibility than just to provide deer and elk with full stomachs. They must also insure that the nutritive quality of the forage ingested is sufficient to meet the needs of the animals. Nutritional relationships may be more important than generally recognized (Schneegas and Bumstead 1977). Knowledge of food habits must be supplemented with information on nutritive contents of forage species to gain insight into the nutritive quality of the animal's diet.

Appendix Table 7 lists nutritive contents of important deer food plants by time of year. This list is not complete. All authorities given in the table have analyses of one or more species that are not listed. Additional information is provided for Interior Chaparral species (Swank 1956, Reynolds 1967, Pond 1976), for several species in the Lower Colorado River Valley (Hanley and Brady 1977), and for forage species in the Guadalupe Mountains, New Mexico (Snyder 1961). Several species of mistletoes common to the Southwest also have been analyzed (Urness 1969).

Selective feeding behavior by cervids necessitates nutritional analysis of each plant part—leaves, stems, fruits, or flowers—whenever feasible or applicable. It is important not only to know which species deer are feeding on, but which plant parts they are eating.

Only a limited number of nutritive attributes are included in Appendix Table C7. Digestible dry matter is a general index to the overall value of a forage item, indicative of energy content. This was obtained through in vitro procedures developed by Tilley and Terry (1963). Although values obtained in this manner often underestimate true digestibility or digestibility obtained in vivo (Urness et al. 1977), comparisons of plants and plant parts listed are still valid because they are relative.

Protein, energy, and phosphorus, or undersirable calcium/phosphorus ratios are the most common limiting nutritional elements on critical deer winter ranges (Dietz et al. 1962, Dietz 1972, Wallmo et al. 1977). Protein, phosphorus, and calcium are included in Table 7 because low phosphorus and marginal protein levels have been noted for many areas of the Southwest (Hanson and McCulloch 1955, Swank 1956, Urness et al. 1971, Short 1977). Also, McNamara (1979) linked protein content to relative palatability, at least among individual plants of the same species.

Acid detergent fiber (ADF) was obtained following methods developed by Van Soest (1966). ADF has been included in many analyses because it has a close inverse relationship to, and is, perhaps, the best predictor of digestibility (Van Soest 1966). Urness (1969) noted that digestibility is inversely related to ADF for mistletoes, but later noted (Urness and McCulloch 1973) that ADF, while showing a similar relationship to digestibility in mule deer, does not demonstrate a consistent relationship to digestibility in Coues white-tailed deer.

Most forage species listed in Appendix Table C7 are shrubs because shrubs commonly are considered most important to deer. However, methods of food habit analyses are generally biased in favor of shrub (woody) material because shrubs are less digestible or more easily observed than herbaceous species. Another reason for past emphasis on shrubs has been that researchers have shown a general preoccupation with winter ranges. Shrubs are often the only food materials available to ungulates because of snow depths or suboptimal range conditions. Such biases have led Gill (1976) to

declare that, "the 'myth of the browser' is one of the most ancient, persistent, and pervasive of all mule deer myths and continues even today to dominate most mule deer management philosophy and activity." The importance of nutritious, herb-dominated summer diets to mule deer productivity was first suggested by Julander et al. (1961) and has only recently been emphasized by Salwasser (1976) and Pederson and Harper (1978).

Studies in the Southwest have indicated high use of browse, but differences exist among investigators regarding its value to deer. Some of these differences can be explained by range condition. Anderson et al. (1965) found that shrub-dominated diets do not provide nutritional levels adequate for lactating females. However, the Great Basin Conifer Woodland ranges studied by Anderson et al. (1965) were in poor condition, and deer were restricted to feeding only on those species that were available rather than on preferred, and presumably more nutritious, species. Boeker et al. (1972), working on good condition pinyon-juniper range, noted that mule deer do well on year-round diets made up almost exclusively of browse, and that even if the forb crop fails to materialize, shrubs are abundant and nutritious enough to provide an adequate yearlong diet. The browse species being utilized on these two areas studied by Anderson et al. (1965) and Boeker et al. (1972) were different, hence the different conclusions.

The importance of nonbrowse material in deer diets has been emphasized in several southwestern studies (Urness et al. 1971, Short 1977, Smith and LeCount 1979). The importance of herbaceous forages in supplying phosphorus and protein cannot be overstated. Although deer appear to be dependent upon shrubs for a major portion of their diet, the superior quality of forbs, taken in smaller amounts, raises nutrient intake to satisfactory levels (Urness et al. 1971). Short (1977) similarly emphasized the importance of cactus fruits during critical periods of the year. Smith and LeCount (1979) noted a high correlation between fawn survival and winter forb yield, although they indicated that forage conditions are not limiting except in years of extreme drought.

One annual grass (foxtail brome) are two forbs, the annual alfalfa and the perennial Wright deer vetch, are included in Appendix Table C7. Note the high protein content of these plants relative to that of most browse species. Similarly, the high digestibility of barrel cactus and jumping cholla fruits approaches optimum.

The nutritive value of individual forage items is not an adequate measure of deer range because of the animal's choice of food items. Plants poor in protein may be good sources of phosphorus, energy, or other nutrients. It is the collective intake of nutrients in the total diet that is critical (Urness 1973). Information from these two sources, food items consumed and nutritive composition of those items, are combined by weighting the contribution of each forage species as a multiple of its nutrient content and its percent in the diet. Although only an estimate, it does provide a useful index that can define a deficiency problem should it exist (Urness and McCulloch 1973). Once nutrient intake is known, it can be compared with the seasonal nutrient requirements of the animals to determinate the nutritional adequacy of the diet. Halls (1970) defined nutritive requirement as the minimum amount of a nutrient that will promote optimum body functions in a balanced ration. Precision, although implied in the definition, does not really

exist because the minimum can vary with animal and environment.

Initial work in nutrition of wild species was based on techniques developed for domestic livestock. Needs of livestock have been expressed on a daily basis, a functional approach both in feedlots and those range operations where supplemental feeding is feasible. Wild ungulates are, however, seldom found in such nutritionally uniform environments. Throughout most of the West these animals spend summers on high-elevation ranges that generally provide adequate foods. Low-elevation winter ranges, however, often have questionable nutritional adequacy. Nonmigratory deer in southern Arizona and New Mexico also face a summer period of stress marked by very hot and dry conditions.

Dcan (1976) has suggested that, for wild ruminants, we should think of seasonal, rather than daily, requirements. Nutrient intake should allow young to survive the winter and females to reproduce successfully. He emphasized that whether or not an animal receives maintenance nutrition each day is of secondary importance.

Appendix Table C8 lists digestible energy, protein, calcium, and phosphorus requirements for several ruminants. Although requirements for domestic cattle and sheep are included, caution should be used when extending these findings to wild animals. There are similarities between deer and domestic ruminants, but there are also important differences because deer have characteristics of their own (Nagy et al. 1969). Information derived from domestic animals can be useful, but intensive cervid management must be based on a thorough knowledge of specific physiology.

Examination of Appendix Table C8 reveals an almost total lack of information specifically for mule deer and elk. Many researchers have assumed that requirements for mule deer are the same as for white-tailed deer, a logical approach because of similarities in size, foods, digestive tract morphology, and general behavior. This approach has been continued in this paper. An attempt should be made, however, to verify this relationship. Specific data on elk also are needed. Extrapolations from cattle, sheep, or white-tailed deer requirements for elk should not be made without verification because of differences in size and food habits.

Nutrient levels of seasonal diets of Coues white-tailed deer are presented in Appendix Table C9 for one habitat type and of mule deer in Appendix Table C10 for four habitat types. Protein levels can be considered adequate on ponderosa pine summer range. Energy intake, indicated by digestible dry matter, is also good. Calcium/phosphorus ratios become marginal during later months, but both the ratios and total phosphorus content are acceptable in early summer. The same is indicated for mule deer on Interior Chaparral ranges, although depressed digestibility during autumn and early winter could indicate potential energy deficiencies. All other diets, Coues deer in Interior Chaparral and mule deer in Semidesert Grasslands and pinyon-juniper, have deficiencies at critical seasons. Levels of protein intake during midsummer (from 9% to 10%, Appendix Tables C9 and C10), the fawning season over much of the Southwest, are considerably below the 12.7% to 20.2% levels recommended for white-tail fawn growth (Appendix Table C8). Phosphorus also is below recommended levels. Short et al. (1977) indicated that phosphorus intake may be borderline for maintenance in pinyon-juniper habitats except dur-

ing spring. Calcium/phosphorus ratios, considered good to fair (Urness 1973) during the summer, become less acceptable at other seasons.

Although there is no direct evidence that links phosphorus deficiencies and undesirable calcium/phosphorus ratios to limited deer reproductive success, sufficient doubt remains concerning such effects on vigor and milk production of females with fawns to make research into these factors desirable and necessary (Urness and McCulloch 1973).

There are several other nutrient-related problems that can be important on southwestern ranges. Little mention has been made of crude fat contents of forages because no information is available for this region. Although the ability to synthesize fats releases ruminants from dependence on external sources of fat, the crude fat content can indicate the energy values of a food item.

The crude fat component can contain fat-related monoterpenoid compounds variously referred to as aromatic, essential, or volatile oils. These compounds are not particularly digestible and may inhibit rumen function. Important southwestern forage plants that contain significant amounts of these oils are big sagebrush and the junipers. Volatile oils of big sagebrush adversely affect rumen function by inhibiting growth of bacteria, reducing rate of cellulose digestion, and decreasing the rate of gas production and volatile fatty acid concentrations (Nagy et al. 1964).

Junipers also contain significant amounts of volatile oils (Jobman 1972, Tatro et al. 1973, Schwartz et al. 1980a), although amounts and kinds vary with species, time of day, and season. Dietz and Nagy (1976) found 20-25 different volatile oils in each of three different species of juniper (alligator, Rocky Mountain, and Utah juniper). They stated that the amount of each individual compound seems a characteristic of the species. Jobman (1972) and Schwartz et al. (1980b) offered the same three species to mule deer in feeding trials and, combining this information with data on volatile oil concentrations and antimicrobial properties, concluded that alligator juniper is the best deer forage because of its relative palatability and because it can be a major diet component without inhibiting rumen microbial action.

Not all researchers agree that volatile oils are significant inhibitors of intake or digestibility. Welch et al. (1981) stated that the proposed role of monoterpenoids in determining preference among species or accessions of sagebrush was questionable. Welch and Jordan (1981) found no relationship between total monoterpenoid content and *in vitro* digestibility of accessions of big sagebrush from mule deer winter range. One study noted an 80% reduction in monoterpenoid levels of mule deer rumen contents compared to the level expected from ingested forages (Cluff et al. 1982) and suggested that these volatile oils were lost during mastication, eructation, absorption, or a combination of factors. A later study provided some evidence that most volatile oils were lost during mastication (White et al. 1982). These recent studies indicate that relationships between monoterpenoid compounds and mule deer selectivity and/or digestibility are more complex than indicated by earlier research.

Other plant species, particularly oaks, contain phenolic compounds called tannins, which are known to depress digestion in the rumen by antimicrobial action and to inhibit enzyme activity (Arnold and Hill 1972, Swain 1965).

Another facet of nutrition seldom considered is the high water content of new growth, especially young grass and forb leaves. Mule deer coming out of hard winters tend to gorge themselves on this lush spring growth. The small rumino-reticula of deer holds only limited amounts, and if 90% of the volume ingested is water they simply cannot consume enough material to meet total dry material and energy demands. This sudden dietary switch can result in scours, impairment of rumen function, and frequent secondary infections in the digestive tract. Combined impacts may often be fatal (Dietz and Nagy 1976).

Habitat Utilization and Management

Nonmanipulative Factors

Nonmanipulative factors are those that influence habitat development and use but that cannot be managed or controlled, at least with present technologies. Such factors include climatic conditions, weather, and geologic features such as parent materials and topography.

Weather conditions that have the potential to influence wild ungulates and their habitats are temperature, relative humidity, wind, atmospheric pressure and precipitation—including amount, seasonal distribution, and form (rain or snow). Discussion of each of these factors is beyond the scope of this paper. Summaries are available concerning their effects on mule deer (Loveless 1967, Miller 1970), elk (Sweeney 1975), and, on a theoretical basis, for wildlife in general (Moen 1973). Precipitation is one of the most important unmanageable components of wildlife habitat in the Southwest and does merit consideration. Snow depths influence utilization of winter ranges throughout the two state area, although impacts are more continuous in the Mountains and Associated Areas and the North. Drought, seasonal and annual, is probably one of the most important factors in desert, chaparral, and grassland habitats although its impacts are occasionally felt throughout the Southwest.

Snow Depths

Snow depths influence migration and within-season movements. Initiation of elk migrations to winter range appears to be governed by increasing snow depths at higher elevations (Anderson 1958, Sweeney 1975). Speed and distance of descent also is influenced by snow (Anderson 1958, Lang 1958a). The spring migration, however, is more closely associated with initiation of spring forage growth rather than the retreating snow line (Knight 1970, Sweeney 1975). Similar patterns have been observed for migratory populations of white-tailed deer (Richardson and Petersen 1974).

Movements and feeding activity are also regulated by snow depths. Sweeney (1975) determined that depths approaching 16 inches cause elk to move to more exposed areas on winter range and that depths in excess of 27 inches prohibit use by elk. Mule deer movements are impeded by snows ranging from 10 to 12 inches deep, and depths of 20-24 inches preclude the animals' use of an area (Loveless 1967). Similar results were noted by Gilbert et al. (1970). Hosley (1956) stated that loose snow depths in excess of 20 inches makes travel difficult for white-tailed deer.

Snow conditions, such as crusting and density, will obviously influence these critical limits. Although crusting may occasionally be heavy enough to support ungulates, oftentimes animals break through suffering injuries or excessive tiring. Although crusted snow prevents any form of foraging for low-growing

plant species, animals can forage through limited amounts of soft snow.

Winter distributions are governed by available forage as modified by snow depth. Snow depths and conditions are not uniform on most western ranges. Crusting, for example, is generally less evident under a partial tree canopy than in open meadows (Knight 1970). Winds cause snow accumulations in certain areas—particularly leeward slopes, depressions, and narrow arroyos—but also remove snow from other sites, such as ridgetops and upper portions of windward slopes. One of the more important factors influencing snow depth is topography, particularly aspect (exposure). Many investigators have noted that cervid distribution is keyed to forage availability on exposed south-facing slopes, because of more direct insolation, as compared to sheltered north-facing slopes where snow does not melt and tends to accumulate (Robinette et al. 1952, Anderson 1958, Lang 1958a, Dalke et al. 1965a, Loveless 1967). Wallmo et al. (1977) concluded that deer mortality rates are controlled by snow conditions and duration of winter rather than by the total forage resource, because snow and cold determine the energy cost for energy gained from grazing.

Differences in snowfall between years can result in a natural, erratic form of rotation grazing on winter ranges (Gilbert et al. 1970). Deer and elk will migrate only as far as forced by snow depth. Varying snowfalls between years can result in use of different winter ranges, thereby avoiding repeated overuse of the same area. This natural rotation is normal on winter ranges in the central and southern portions of Arizona and New Mexico. Snow conditions are relatively more consistent and extend over all winter ranges in northern areas, however, and cervids concentrate on the limited sites that have little or no snow because of exposure or wind. These critical areas may make up only a small portion of the winter range in years of moderate to heavy snowfall and, therefore, require special attention in land management planning (Gilbert et al. 1970).

Drought

Precipitation varies throughout the Southwest. There is a general decline in yearly amounts from northeastern New Mexico to southwestern Arizona as well as significant seasonal differences (Table 3). Precipitation is highest in spring and summer in eastern New Mexico (Tucumcari); high in the summer and low in spring with only a moderate amount falling in winter in southwestern New Mexico and southeastern Arizona (Lordsburg and Tucson); and highest in winter and lowest in spring in southwestern Arizona (Yuma). Intensity of precipitation also varies between seasons. Winter rains are generally gentle and tend to cover extensive areas, whereas summer rains

Table 3. Relative precipitation (percent) during three selected seasons at four locations in the Southwest.¹ Seasonal figures are percentages, totals are inches (U.S. Department of Commerce 1975, 1977).

| Location | Season ¹ | | | Total annual precipitation (inches) |
|--------------------|---------------------|--------|--------|-------------------------------------|
| | Spring | Summer | Winter | |
| Tucumcari, N. Mex. | 35 | 54 | 11 | 14.9 |
| Lordsburg, N. Mex. | 13 | 60 | 27 | 10.4 |
| Tucson, Ariz. | 12 | 61 | 27 | 11.1 |
| Yuma, Ariz. | 14 | 41 | 45 | 2.7 |

¹Spring = March through June, Summer = July through October, Winter = November through February.

are localized and very severe. Most precipitation that falls during torrential summer rains is lost as runoff or is quickly evaporated because of intense solar radiation and high temperatures. Relatively lesser amounts percolate into the soil and become available for plant growth as compared to winter rains (Swank 1958).

The routinely severe spring drought from southwestern New Mexico westward influences quality of habitat. Its effects become more severe in central and western Arizona, especially at lower elevations. Swank (1958) noted two distinct periods of growth in Interior Chaparral. If winter precipitation is normal, a substantial amount of growth occurs in early spring. Another period of growth occurs later in the summer when rains are sufficient to recharge soil moisture, although growth during this period is less dependable. Desert mule deer and Coues white-tailed deer have adapted to the summer growth with later parturition periods. Both species have peak fawning periods in August (Swank 1958, Knipe 1977), whereas Rocky Mountain mule deer on the Kaibab Plateau (Russo 1964) and in northern New Mexico (Lang 1958b) peak in late June.

If the winter and/or summer rains fail to materialize, the resulting reduction in plant growth during either period can affect deer populations. Hanson and McCulloch (1955) have noted that some form of drought, which they define as any year with 75% or less of the long-term precipitation, occurred on the average of every 4.25 years, some extending over a period of several years. Gestating females depend on adequate plant growth in early spring to carry them through the hot, dry spring—early summer period. They further rely on green herbaceous growth, a response to midsummer rains, to provide needed protein for lactation. Drought during either period can force deer to alter their diets significantly (Anthony 1976), and this change can reduce deer populations by decreasing fecundity and fawn survival.

Short (1979) has suggested that an apparent recent decline in southwestern deer herds was caused by limited fawn recruitment combined with antlerless deer hunting. He attributed the low fawn recruitment to habitat quality, expressed as an interrelated function of soils, precipitation, temperature, and vegetation. Short's review indicates the long, hot, dry summers result in light-colored soils that are low in organic matter, have aggregates coated with iron oxides, and have a shallow horizon of calcium carbonate. High levels of calcium, aluminum, or iron reduce the availability of phosphorus to plants, even if phosphorus is present in the soil. Total soil nitrogen is inversely related to mean temperature, and warm desert and semidesert soils have low levels of nitrogen. These low soil phosphorus and nitrogen contents produce forages that contain suboptimum levels of phosphorus and protein. Short's (1979) contention that low productivity of some deer herds is partially caused by this association between aridity, soils, and forage quality, although not conclusive, certainly merits consideration. A marginal situation such as this would be worsened significantly if there were winter or late summer droughts.

Manipulative Factors

This section is a consideration of those facets of the environment that can be managed or controlled to enhance wildlife habitat. Primary purposes of habitat manipulation are to provide adequate food, water, and cover, all of which ameliorate conditions imposed by uncontrolled environmental factors. The two most important functions to be considered are timber management and grazing by livestock. Others include recreational activities, roads and highways, agricultural activities other than

livestock grazing, and urban-suburban development. These may have little overall impact on wildlife habitat but can be locally important. Prescribed or controlled burning can also be a useful tool to manage habitats on forestlands and rangelands. Effects of fire are not limited to controlling overstory but also include changing the composition of understory, influencing nutrient cycling, reducing undecomposed organic materials, and modifying cervid utilization patterns. Hence, use of fire will be discussed in a separate section.

Timber Management

Manipulation of cover/forage area ratios can be accomplished through timber management on forestlands and through brush control techniques developed by livestock interests on rangelands. Although such programs have been continually employed, management procedures have been primarily concerned with maximizing sustained crops of wood fiber or herbaceous forage; seldom have shrub forages or wildlife cover requirements been considered. The 40% cover to 60% forage area concept discussed in an earlier section will provide biologists with a base to provide more understandable prescriptions.

Understory-Overstory Relationships

Many studies have demonstrated increases in understory production following reduction or removal of tree overstory (Ffolliott and Clary 1972). Most have shown significant inverse relationships through use of various regression models. This inverse relationship is generally curvilinear or linear (Clary 1975). Data describing these relationships have been reported for southwestern forest types by Reynolds (1962b), Clary and Ffolliott (1966), Jameson (1967), Clary (1969), Reynolds (1969b), and Clary et al. (1975). Conversely, Beets (1971) did not detect a significant change in total understory between six thinning levels in ponderosa pine stands. The only noticeable increase was in forb production.

The response of understory vegetation to silvicultural treatments is a key factor. A significant increase in understory vegetation must occur before elk or deer will respond. They will not favor logged areas, whether selectively cut or clearcut, until the understory vegetation has developed (Reynolds 1962b, Clary and Larson 1971, Lyon 1976). An increase of herbaceous forage does not guarantee increased use by the animals, however. Reasons for lack of response to what would normally be considered a favorable habitat change are not known (Neff 1972).

It is difficult to determine what an understory response will be when the overstory is changed. Regression models developed to predict understory production based on certain attributes of the overstory, generally basal area or canopy cover after logging (cf. Jameson 1967), should be considered site specific; they should be applied elsewhere with caution.

The timber site index apparently cannot be used as an indicator of potential for understory production, at least for lodgepole pine sites. The failure of site index to predict understory yields may be due to difference between major rooting zones of trees and understory, periods of active growth, and efficiency of moisture and nutrient utilization. Also, the site index represents tree growth over a period of years, whereas understory production is the result of one year's growing conditions (Basile 1971).

Thus, managers selecting sites for wildlife habitat improve-

ment must consider carefully the site factors that could influence understory production. Reynolds (1962b) noted that forage production was much greater on north- and south-facing slopes than on ridgetops, meaning slopes would offer greater opportunities for improvement of forage production. Other areas that might be avoided would be convex slopes, sites with impermeable soils or large amounts of surface rock, and areas with extremely deep undecomposed organic layers (unless, of course, this material could be removed or reduced with prescribed fire).

An initial decline in understory production on any site may be expected immediately after logging operations. Reynolds (1962b), working in North Kaibab ponderosa pine, noted that aspen sprouts and forbs increased, but grasses and sedges decreased, the first year after logging, presumably because of the mechanical disturbances caused by the operation. From the second to the sixth year production of all classes increased. Thereafter all classes declined except aspen sprouts, which were still increasing 11-15 years after logging. Regelin and Wallmo (1978) noted comparable results in subalpine forests in Colorado.

Patton (1969) did not note the initial decline after logging in ponderosa pine stands in the White Mountains of Arizona. Understory production increased 100% from precut conditions 1 year after logging. Similarly, Patton (1976) found that overstory removal and selection-cutting in a mixed conifer forest resulted in an increase of 50 pounds per acre of forbs and grasses 2 years after tree harvest. Browse plants increased from 525 to 820 stems per acre over the same period.

The preceding discussion has emphasized the inverse relationship between amount of overstory and resulting productivity of understory as well as the positive relationships between understory (forage) production and cervid response. This implies an inverse relationship between cervid use and amount of overstory, as demonstrated for elk by Clary and Larson (1971) with some exceptions as noted by Neff (1972). However, when managers make recommendations for timber harvest, they cannot be concerned with only how much timber to remove to stimulate forage production but must give equal attention to how much should be left for cover (Fig. 6). The balance between the two factors is of utmost importance (Patton 1976).



Fig. 6. Thinning dense, immature ponderosa pine. Such timber stand improvement practice will result in increased forage production but will also reduce necessary cover. A balance between these factors must be maintained.

Forest Openings

Although natural and created openings on forest lands have been considered generally beneficial to mule deer and elk, provided they are not too large, there is no consensus of opinion in the existing literature regarding their real value.

Mule deer apparently use openings on forested summer range less than do elk. Reynolds (1962a) determined that deer used natural openings, ecotones between forest and openings, and forested areas about equally in a ponderosa pine area that had been "logged over." Similarly, Pearson (1968) stated that no apparent advantage was gained for mule deer habitat by clearcutting ponderosa pine. Pearson's study was also conducted in an area where the forest around the clearcuts had been thinned to basal areas ranging from 20 to 110 square feet per acre.

Increased mule deer use of a watershed that had been cutover has been documented. Deer use increased 1.68 deer days per acre on a cutover watershed and decreased 0.03 deer day per acre on an adjacent unharvested watershed studied by Patton (1969). Two other studies, one in ponderosa pine (Reynolds 1962b) and one in mixed conifer (Patton 1976), illustrated similar increases in mule deer use of areas in which the overstory had been reduced enough to permit understory response. Clearcuts or natural openings were not mentioned as being present in either area, although Patton (1976) stated that basal area was reduced to 29 square feet per acre on a portion of his study area. The remaining trees were not uniformly distributed but were in patches of varying sizes. This indicates a portion of the area could have been left relatively open.

The results of these studies indicate that clearcutting may not add anything to mule deer habitat if done in stands that are already under some form of overstory management, at least on summer range. Thinning overstory to the point where adequate deer forage is produced apparently can accomplish the same thing, provided sufficient security and thermal cover remain. Deer prefer areas where the forest understory is somewhat open with dense stands of young pine left as bedding cover; however stands can become too open for optimum use (Reynolds 1972). Reynolds recommends selective logging to remove 35-50% of the volume of a stand. In an earlier paper Reynolds (1969b) suggested that the upper limits of a production base of 80 to 100 square feet basal area in pole-sized stands is adequate for achieving major objectives for timber and deer production in ponderosa pine. Clary (1972) found that optimum forest densities for mule deer, elk, and turkey are between 40 and 80 square feet of basal area per acre (60 in smaller size classes and 70 in the 12- to 22-inch d.b.h. class). Densities less than 40 do not provide adequate cover, and those greater than 80 prevent production of an adequate herbaceous understory.

The idea of including both cover and forage types in the same range of stocking levels may have some merit. Suring and Vohs (1979), for example, found that Columbian white-tailed deer utilize habitat types that provide both food and cover more heavily than those that produce food or cover alone. However, it may be equally feasible to separate them. Mule deer may require much greater forest densities on some ranges, particularly during hunting seasons. Heavy stocking levels in excess of 160 square feet basal area per acre in immature stands (Reynolds 1969b) are necessary to provide escape and security cover under these circumstances. Light stocking levels, 20-40 square feet basal area, may also be necessary to insure adequate understory production on some sites, depending on size and density of trees. Another argument against a relatively uniform set of stocking ranges is that it represents a trend towards homogeneity

in forest management rather than diversity, which most wildlife managers feel is better. More diversity exists between stands stocked at 40 and 160 square feet than between 60 and 80, assuming equal tree size classes.

Basal area is a measure frequently used by foresters. Basal area alone, however, is not a good term to describe wildlife habitat because, although a function of the number and size of trees, it does not indicate the value of a stand for cover. Table 4 shows the differences in number of trees per acre across different levels of size and basal area. At a basal area of 80 square feet, for example, tree density can vary from 571 trees per acre when the average d.b.h. is 5 inches to 23 trees per acre when the average d.b.h. is 25 inches. Recommendations on tree stocking levels for wildlife should, therefore, include a statement on size class.

Table 4. Number of trees per acre at given d.b.h. (inches) and basal areas (square feet per acre).

| Average d.b.h. | Basal area | | | | | | |
|----------------|------------|-----|-------|-------|-------|-------|-------|
| | 20 | 40 | 60 | 80 | 100 | 150 | 200 |
| 3 | 400 | 800 | 1,200 | 1,600 | 2,000 | 3,000 | 4,000 |
| 5 | 143 | 286 | 429 | 571 | 714 | 1,071 | 1,428 |
| 10 | 36 | 73 | 109 | 145 | 181 | 272 | 363 |
| 15 | 16 | 33 | 49 | 65 | 81 | 122 | 162 |
| 20 | 9 | 18 | 28 | 37 | 46 | 69 | 92 |
| 25 | 6 | 12 | 18 | 23 | 29 | 44 | 59 |
| 30 | 4 | 8 | 12 | 16 | 20 | 31 | 41 |

Clearcuts may serve as a foraging area for mule deer in forest stands that are extensive, dense, and stagnant. Wallmo (1969) and Regelin and Wallmo (1978) documented a significant increase in mule deer use of an alternate-strip clearcut watershed in lodgepole pine and spruce-fir timber in Colorado. Deer use was also greater on clearcuts than in the adjacent uncut strips on the harvested watershed. Evidence indicates that the clearcuts were the only forage-producing sites available in an otherwise dense forest below 11,200 feet. Thinning the entire watershed at varying intensities rather than clearcutting approximately one-half of it could have resulted in development of an understory that was equally acceptable to deer. The food and cover relationships would be different but not necessarily in a negative way. Either silvicultural option may have resulted in favorable deer response.

Although there is no evidence that creation of openings by clearcutting in timbered stands that have been thinned to reasonable levels will benefit mule deer on southwestern forest summer ranges, complete overstory removal from feeding areas on important winter ranges is a valid and necessary practice (Klebenow 1965). Clearcutting to stimulate development of, or to enhance, existing seral forest stages will be discussed in the following section.

Elk utilize forest openings more consistently than do mule deer, although some contradicting evidence can be found. Edgerton (1972), working in a mixed conifer forest in eastern Oregon, determined elk use to be highest in clearcuts, intermediate in uncut stands, and lowest in partial cut stands, during the 5 years following harvest. He attributed these use patterns to forage production in clearcuts and the excellent hiding cover provided by uncut stands. Partial cut stands lacked adequate amounts of either. Elk use was determined by counting pellet groups.

Reynolds (1962a, 1966b) noted similar results in Arizona ponderosa pine. Elk use, measured by pellet counts, was highest in natural openings, intermediate at the forest edge, and lowest in the surrounding forest. Pearson (1968), also using pellet group counts in ponderosa pine, found that elk use was highest on clearcuts, lowest on the thinned unit with the fewest trees removed, and intermediate in other stocking levels. No pellet groups were found in the unthinned stands over the 4-year study period. Ward (1976) and Lyon (1976) stated elk use clearcuts in Wyoming and Montana, respectively, when palatable vegetation is available.

Other studies have demonstrated less inclination by elk to use openings. Reasons for such differences are not clear but may be explained by local differences, such as human disturbance, degree of livestock use, topography, or elevation. Reynolds (1966a), working in Arizona spruce-fir, noted that elk use was highest closest to the border, on both the forest and opening sides, both for natural openings and clearcuts. However, numbers of pellet groups did not differ appreciably between adjacent forests and openings. In Idaho, elk did not show a preference for clearcuts during daylight hours; however the fact that many radio-determined locations were within 200 yards of clearcuts indicate the elk may have been feeding in such openings at night. Observations of radioed elk and unmarked elk revealed only 31% of the clearcuts available in the study area were used by elk (Hershey and Leege 1976). Western Montana elk used forested areas predominantly during summer for both feeding and bedding. They appeared to prefer mature timber stands characterized by large trees with patches of smaller reproduction. Although elk commonly used natural openings, use of clearcuts was only 0.2% of total use (which included day and night observations), significantly less than the availability of clearcuts in the area (Marcum 1976). Marcum concluded that logging on high-elevation summer ranges produces no measurable benefits for elk.

Some variation is also evident in the recommendations for optimum sizes of forest openings for elk, depending on forest type and age of clearcut. Reynolds (1962a, 1969b) suggested 45 acres as an optimum size in ponderosa pine in Arizona, but Reynolds (1966a) recommended openings of only half that size, 20 acres, in spruce-fir. Lyon (1976) suggested 10- to 40-acre clearcuts with good slash cleanup were most acceptable to elk on summer range in western Montana. In another Idaho study, elk avoided clearcuts of less than 51 acres, made proportional use of those from 51-200 acres, and preferred clearcuts larger than 200 acres. When using the larger clearcuts, however, elk did not use all the acreage, seldom going more than 50 yards from the edge (Hershey and Leege 1976). These investigators suggested that elk used the larger clearcuts not because of size but because these particular clearcuts were recent (less than 10 years old).

A Utah study in lodgepole pine indicated habitat preferences for elk grazing were wet meadows, revegetated roads, clearcuts, and dry meadows, in descending order of importance (Collins 1977). Elk preferred wet meadows, revegetated roads, and mature forest as resting areas. They used dry meadows and clearcuts less as feeding areas. In addition to using wet meadows and revegetated roads as primary feeding sites, elk also used them as resting areas because such areas are close to cover. Wet meadows and revegetated roads seldom exceed 50 feet and 33 feet in width, respectively, and are often broken up by fallen trees or high shrubs and have dense cover on both sides. Elk seldom ventured into large openings more than 200

feet from the forested edge (Collins 1977). Patton and Judd (1970) determined, using pellet counts, that deer and elk in the Southwest utilized forest edge more than meadows, but indicated that time spent in meadows may have been more important because of quantity and quality of forage. The wet meadows investigated by Patton and Judd (1970) were relatively flat areas or potholes with poor drainage, near heads of or along streams, and were apparently larger than those described for Utah by Collins (1977). Hence, elk used these meadows less for resting and other nongrazing activities.

Elk also respond to timber management practices other than clearcutting but, unlike deer, tend to use openings to a greater extent, regardless of treatments imposed on the rest of the stand. Reynolds (1962a) and Pearson (1968) both recognized greater use by elk of natural openings and clearcuts in forests that had been logged or thinned. Patton (1969, 1976) documented increased elk use of watersheds that had various thinning treatments in ponderosa pine and mixed conifer stands. Clearcuts ranging in size from 2 to 32 acres were included in ponderosa pine (Patton 1969) but were not specifically mentioned in mixed conifer (Patton 1976) although the description of harvest implied heterogeneity in remaining tree cover. Clary and Larson (1971) found that relatively consistent elk use patterns showed long-term preferences for areas with higher herbage yields, low timber basal area levels, and some alligator juniper in Arizona ponderosa pine. They did not attempt to measure elk use in openings.

Researchers conducting many of the previously mentioned studies based results on relative accumulations of pellet groups in different habitat units. There is growing concern that this method may result in biased interpretation when used to determine habitat preferences. Julander et al. (1962) suggested that studies are needed to relate the number of fecal droppings on resting and feeding areas to the time animals actually spend on those areas. Neff (1968) stated that many studies have been based on the assumption that pellet group counts in an area are related to the time the animals spend on that area, or that the counts indicate the relative value of the site to the animal's well-being. He further stated that these assumptions do not appear to be trustworthy because statements of the relationship between defecation rate and other animal activities are based on conjecture.

Later studies have indicated that these concerns are valid. Ward (1976) found a poor correlation between time spent by elk on mountain meadows, as recorded by time-lapse photography, and fecal counts. He gave two possible reasons: one was that elk may have been feeding at night, hence not recorded by the cameras; the second was that elk were using meadows mainly for feeding and not resting or ruminating, which could cause variation in fecal deposition. Fecal counts (during 1 year in which elk use recorded by cameras was considered normal) were lower than expected (Ward 1976). Collins and Urness (1979) confirmed these suspicions when they found that tame elk did not defecate at uniform rates but defecated most when they were most active. About 40% of all defecations occurred when the animals were traveling, but this activity represented only 4-6% of the elk day. Collins (1977) noted that elk often graze up to 2-½ hours without defecating, then, while moving to water or another feeding site, defecate one to four times in 3-15 minutes. It was further demonstrated that elk defecate more per unit of time in those habitats where more walking is required in grazing activity. Defecation behavior of mule deer was similar but differences between pellet-group distribution and actual use

were less pronounced (Collins and Urness 1981). These studies (Ward 1976, Collins and Urness 1979, Collins and Urness 1981) indicate that elk and deer use, determined by fecal counts, can be biased against feeding areas in general. Even when comparing different feeding areas, pellet counts would tend to underestimate use made of the most productive or floristically rich areas because the animal would be moving less while feeding.

Cases where comparisons are being made between large treated and untreated areas where the treatment includes all functional habitat types, such as the watersheds described by Patton (1969, 1976), probably are valid uses of pellet count data. However, interpretations on the use of adjacent habitats used for different animal functions (e.g., resting or feeding) must be made cautiously.

Deciduous Seral Stages

The dominant deciduous seral species in the Mountains and Associated Areas of the Southwest are Gambel oak and quaking aspen. Gambel oak does not form the extensive, dense stands in the Southwest as described for parts of Colorado by Brown (1958) but exists as an associate within the Great Basin Conifer Woodland and Montane Conifer Forest. Oak grows in relatively pure stands where the ponderosa pine overstory has been disturbed (Schubert 1974). Although considered a seral type at mid-elevations (5,000-8,000 feet), once established it can become a relatively stable component of the forest. The species is ecologically adapted to reproduce and survive in association with ponderosa pine where it can contribute significantly to the composition of a stand (Reynolds et al. 1970). Gambel oak grows as a shrub or tree, depending on the site, and reproduces primarily from root sprouts. Stem sprouts are also produced from buds at the base of the stem when the top is damaged or killed. Reproduction from seed is rare (Reynolds et al. 1970).

Gambel oak is an important wildlife food; both browse and mast are utilized extensively by deer and elk (Lang 1958a, McCulloch et al. 1965, Reynolds et al. 1970, Neff 1974). Although patch clearing of extensive oak brushlands in Colorado has provided significant benefits to deer and elk (Kufeld 1977), such practices may not be applicable in Arizona and New Mexico. Patch clearcutting will provide an abundance of browse because of the sprouting ability of this species. However, such practices would result in loss of mast.

Selective cutting, in which the best acorn-producing trees are left, would insure the production of both mast and browse. McCulloch et al. (1965) recommended, for mast production, that all oak trees with 80% or more of the crown alive and with a d.b.h. up to 15 inches be retained. Maximum acorn yield is from trees 12-14 inches d.b.h.; those over 15 inches produce fewer acorns regardless of percentage live crown. The smaller diameter classes, with greater than 80% live crown cover, while not producing maximum mast, will provide growing stock for future acorn yields. These findings are similar to those reported by Goodrum et al. (1971), who further suggested that some trees inherently produced more mast than others and that during good mast-producing years such trees should be identified and left. Acorn production by Gambel oak is variable between years (McCulloch et al. 1965), but when oak mast is available, it can allow deer to enter winter in better condition (Duvendek 1962).

Other factors that must be considered are when and how to cut Gambel oak to stimulate sprouting. The palatability and above-average nutritive content of oak leaves has been demonstrated (Urness et al. 1975b). However, the value of Gambel oak browse on winter range (or once the leaves are lost) has

not been investigated. Severson and Kranz (1978), working with bur oak in South Dakota, determined that buds and twigs were not adequate nutritionally for white-tailed deer in winter. Bur oak, although related to Gambel oak, may not have comparable nutritive value because of different sites and associations; however, the bur oak findings indicate a nutritional study of Gambel oak twigs is warranted before large-scale attempts to increase browse production on winter ranges are made.

Quaking aspen stands also are important seral stages in the mixed conifer type of the Montane Conifer Forest and in the Subalpine Conifer Forest. Because of the important role fire has in the perpetuation of this seral stage (Hoff 1957, Morgan 1969, Patton and Avant 1970, Gruell and Loope 1974), management of aspen stands for elk and deer is discussed in the section on Fire as a Habitat Improvement Tool.

Slash

Slash left in areas after timber harvests or precommercial thinning operations can have beneficial or detrimental effects, depending on specific situations. Generally, elk use can be reduced by too much slash. Pearson (1968), Beall (1976), and Lyon (1976) documented adverse elk response to logging debris. Beall (1976) recommended extensive cleanup operations, and Lyon (1976) stated that slash inside clearcut openings and dead and downed material adjacent to openings should not exceed 1.5 feet in height. Thomas et al. (1979a) suggested that reduction of dead and downed material to MM standards (accumulations that would permit a medium rate of fire spread and require a medium effort to control) would minimize the problems. Hershey and Legee (1976) did not notice a difference in elk use of clearcuts treated for slash removal and those not treated but noted that slash buildup was not so tangled that it hindered elk movements or precluded use of forage beneath. They further noted that slash provides elk some escape cover without severely impairing their field of view. The results of this study imply that low levels of slash may have certain advantages for elk. Thomas et al. (1979a) also suggested that long sight distances in critical areas could be broken up by piling or windrowing slash (Fig. 7).

Limited data suggest that the response of deer to slash may be somewhat different than that of elk. Fewer pellet groups were found in areas where slash had been cleaned up as compared to where it was undisturbed (Reynolds 1966c). Slash accumulations covered 3.2% of the ground surface where bulldozers had cleaned up, and 6.8% where the slash was left undisturbed. In a later study, Reynolds (1969a) noted that fallen trees in thinned aspen stands prove effective barriers for deer and elk as well as cattle. Hence, relatively larger accumulations can inhibit use of an area.

Cattle, like elk, prefer areas with less dead and downed material (Reynolds 1966c, 1969a). Slash, properly distributed, could provide initial protection for the developing understory by excluding, or at least partially excluding, larger ungulates. Accumulations, up to a yet undetermined point, do not have measurable effects on the total production or composition of the forage (Reynolds 1966c, 1969a). If mule deer are, indeed, less influenced by such materials, slash in adequate amounts, properly arranged, could provide them with security areas in which to escape adverse behavioral interactions between deer and elk or cattle.

Slash disposal, like clearcutting, is not an all-or-nothing proposition. Large accumulations may have no obvious advantages; but certain amounts, properly placed, can create favorable habitats for wild ungulates. Studies on amounts, kinds, and

arrangements of slash could provide methods to create relatively short-term rest-rotation or animal-selective grazing systems.



Fig. 7. Piling slash can provide a form of security cover by breaking up long site distances.

Harvest Guidelines

Few guidelines have been developed that suggest ways silvicultural treatments, particularly timber harvest, can be modified to improve wildlife habitat. One of the principal reasons for this is that deer and elk react in different ways to any single silvicultural treatment, depending on the environment setting (Thomas et al. 1979a).

Specific guidelines have been developed using the hiding cover, thermal cover, and foraging area requirements for deer and elk previously discussed (Thomas et al. 1979a). The application of these requirements is keyed to land type. Land types were identified by Thomas et al. (1979a), and these account for local environmental variation. Land types were identified on the basis of site potential, soil features, topographic features, and size and general slope of area. Simultaneously achieving wildlife and wood production goals involves complex processes but can be realized by manipulating several variables within the context of a land type.

Hall and Thomas (1979) developed examples of silvicultural options available to forest managers. They give examples for two basic options: management for featured species (using elk and the pileated woodpecker, two species which require completely different habitats) and management for species richness. Management for species richness varies from management for featured species because the objective is to insure that most

wildlife species are maintained in viable numbers; thus, habitat requirements for all species must be met (Hall and Thomas 1979).

Hall and Thomas' recommendations were based on land type and such silvicultural considerations as stand condition, size of treatment, length of rotation, and the scheduling, kind, and number of entries necessary for stand treatment. They emphasized that any prescription must consider local condition; that is, scheduling and selection of silvicultural treatments must be done on a site-specific basis. Regeneration cuts, for example, on 20% of a forested area could give results that range from an anticipated doubling of elk use on a land type that was originally 100% forested, to reducing elk use by one-third on a type that was only 35% forested.

Although Thomas et al. (1979a, 1979b) and Hall and Thomas (1979) advocate principles intended specifically for the mountains of eastern Oregon and Washington, research used in the development of these standards was West-wide, including Arizona. The basic premises, therefore, may have applicability to forestlands of the Southwest. However, caution must be exercised in their use because relevance to conditions in Arizona and New Mexico has not been verified. A joint research-management effort to test the validity of general principles and, if applicable, to develop specific guidelines for southwestern spruce-fir, mixed conifer, and ponderosa pine forests, should be assigned a high priority.

Neff (1979a) suggests general guidelines concerning timber harvest and wildlife habitat for forest types in the Southwest. Although Neff does not directly mention the 40/60 cover/forage ratio, he applies similar basic principles. For example, in mixed conifer stands where cover is limited (i.e., where forest stands are small, isolated upland groves or narrow stringers in drainage bottoms), silvicultural systems should be directed at maintaining the stand in a mixture of species. Harvest, if necessary, should be limited to regeneration or sanitation cuts. Conversely, if forage is limited (as in extensive, mixed conifer stands), harvest of overmature and mature timber should be carried out using several silvicultural systems, including selection, shelterwood, and patch clearcuts depending on and indicated by site and stand conditions. Patch clearcuts should be designed to blend with terrain and to provide maximum deer and elk forage production (Neff 1979a).

Neff developed similar recommendations for ponderosa pine forests, partially accounting for local conditions by basing recommendations on a timber site index. Neff places specific emphasis on perpetuation and replacement of seral deciduous stands—quaking aspen, and Gambel oak—wherever possible.

The previous discussion also emphasizes another important point. Methods to achieve satisfactory partitioning of a stand into security cover, thermal cover, and forage areas depend almost entirely on the silvicultural practices being used in different forest types. Timber management and harvest can result in serious conflicts with wildlife needs, but, properly applied, are potentially the best and least expensive methods of optimizing habitat conditions on forest lands. Wildlife biologists should become familiar with the various silvicultural systems in use within their area of interest. Such a review is beyond the scope of this paper, but summary reports of silvicultural practices are available for southwestern spruce-fir (Alexander 1974), mixed conifer and aspen (Jones 1974), and ponderosa pine (Schubert 1974). Multiple-use implications of harvest systems have also been discussed for spruce-fir (Alexander 1977) and ponderosa pine (Myers 1974) forests of the central Rocky Mountains.

Livestock Management

The conflict or potential for conflict between livestock and wild ungulates has been recognized by and a matter of concern to resource managers and scientists for many years (Mackie 1978). More than 25 years ago, John Hall, then Director of the Arizona Game and Fish Department, observed that, "The problem of Livestock versus Wildlife has plagued us since the turn of the century Instead of decreasing in intensity, this battle has become broader and more bitter as the years go by. It would seem that as we learn more about the proper management of both livestock and wildlife on our ranges, some progress would be made toward solving the battle of conflicting use" (Hall 1955). Apparently, little has been learned over the last 25 years, while the conflict, once limited to livestock and the wild ungulates, has expanded to include all forms of wildlife. The debate also has developed from one that once was restricted to the western regions to one of national importance.

Several recent papers have provided summary reviews of topics directly related to livestock-wildlife relationships in the western United States. Box and others⁶ provide an assessment of history, condition, trends, and problems involved in management of western rangelands. Wagner (1978) developed an analysis of the history and complexity of effects of the livestock industry on wildlife. Urness (1976) and Mackie (1978) have also provided reviews of such interactions with mule deer and wild ungulates in general, respectively. Longhurst et al. (1976) and Salwasser (1976) discussed effects of livestock grazing on mule deer but in the context of other factors which also influence deer habitat (e.g., fire, logging, urbanization).

There are two general ways in which the management of livestock and livestock forage can influence wildlife habitats. One is the influence of livestock per se on structure of habitats and populations expressed through the actions of overgrazing, competition for forage, and behavioral interactions. The other is range improvement practices generally oriented to reducing shrub and tree growth to produce more grass.

Overgrazing and Habitat Alteration

Two basic problems exist regarding overgrazing and wildlife habitats. One is that the concepts, processes, and effects of overgrazing as related to range condition are poorly understood. This problem is partly semantic; many biologists, for example, appear to regard grazing, overgrazing, and overuse as synonymous terms. The second problem is a misconception that any degree of overgrazing can create favorable mule deer habitat.

Range managers generally utilize the range site/range condition concept described by Dyksterhuis (1949) to evaluate rangelands. This concept has not received much attention by wildlife biologists for several reasons. One is that its usefulness is limited to soils and climates where the climax vegetation is suitable for grazing (e.g., grasslands, shrublands, savannahs, or, collectively, rangelands). It is not applicable to, nor intended for, forest sites. Another reason is that it was developed for use by the livestock industry; hence the terminology and method of application often imply exclusion of consideration for wildlife. However, careful review of Dyksterhuis' paper will reveal that the principles have equal applicability to evaluation of wildlife habitat. Indeed, many agencies using the concept are revising guidelines to enhance its usefulness in this respect.

The mere presence of livestock on a unit of rangeland does not necessarily mean that a conflict with wild populations exists. However, a conflict is implied if grazing, overgrazing, and overuse are considered synonymous. Grazing is simply the consumption of standing forage by livestock or wildlife (Kothmann 1974). Overuse occurs if the grazing animals are allowed to utilize an excessive amount of the current year's growth. Overgrazing is caused by continued overuse (over a period of years) and is indicated by a regressive change in plant composition and soil conditions on a site-specific basis.

Proper use or proper grazing, on the other hand, is the degree and time of use of the current year's growth, which, if continued over a period of years, will improve or maintain the range condition consistent with the conservation of other natural resources (Kothmann 1974).

There are varying degrees of overgrazing. A moderately overgrazed range site may exhibit only slightly altered proportions of the plant species which would normally be present in an undisturbed, relatively stable community. A severely overgrazed similar site may support a completely different plant complex.

Changes in composition of vegetation are not the only result of overgrazing. Mulch cover and water infiltration are reduced, soil is trampled and compacted, runoff is increased, and the microclimate becomes drier and more severe. Rate of energy flow is reduced, stratification and periodicity of plants are disrupted, and the orderly operation of many biogeochemical cycles is altered, especially those of water, carbon, and nitrogen. Furthermore, total plant production is reduced (Lewis 1969).

The process of overgrazing is seldom obvious, because it occurs so slowly, a fact which led Gallizioli (1977) to describe it as a particularly insidious form of habitat destruction. Wagner (1978), while not using the term overgrazing, described livestock-induced changes in vegetation as having a profound effect on some wildlife species.

The idea that overgrazing can create favorable mule deer habitats can be difficult to deal with because it does contain elements of truth. The concept, however, must be carefully defined and costs considered before it can be considered valid.

The idea apparently developed from mule deer population responses to a series of conditions in western North America at the time of settlement. There is little doubt that our western rangelands were severely overgrazed at the turn of the century (Box and others,⁶ Wagner 1978). Most damage occurred between 1880 and 1920 in Utah (Julander 1962) and between 1880 and the drought of the 1930's in the Northern Great Plains (Severson 1981). Livestock numbers, however, were drastically reduced on lands administered by the newly created Forest Service between 1905 and 1910. On the Kaibab Plateau for example, livestock numbers were reduced from an estimated 20,000 cattle and 200,000 sheep around 1900 to 8,000 cattle and 10,000 sheep by 1907 (Russo 1964). Mule deer populations began to expand at about the same time these reductions were being made (Rasmussen 1941, Julander 1955). These herds increased on rangelands that had already been overgrazed by livestock. The deer contributed to further declines in range condition. This situation prompted Julander (1962) to ask: How could such large deer populations build up on these already overgrazed ranges?

Several hypotheses have been offered: (1) Utah prohibited all hunting for a period of 5 years beginning in 1908 (Julander 1962), and hunting was also banned in the North Kaibab in 1906

⁶Box, Thadis W., Don D. Dwyer, and Frederic H. Wagner. 1976. The public rangelands and their management. Unpublished report to President's Council on Environmental Quality. [Mimeo copy, Utah State University, Logan]. 57 p.

by President Theodore Roosevelt when he created the Grand Canyon National Game Preserve (Rasmussen 1941). (2) Predator control efforts, while not as efficient as some of the toxicants used after World War II, were widespread and consistent. Mountain lion, grizzly bear, and wolf populations were drastically reduced or exterminated in many areas of the West long before the introduction of 10-80. The North Kaibab received particular attention in this respect. With the creation of the hunting preserve and because of President Roosevelt's personal interest in Kaibab mule deer, government hunters removed 781 mountain lions, 30 wolves, 4,889 coyotes, and 554 bobcats from the Kaibab Plateau between 1906 and 1931 (Rasmussen 1941). (3) The initiation of increases in mule deer coincided with major reductions in livestock numbers, particularly the drastic cuts made in the early part of this century. This could be a response to a decrease in competition for forage (Rasmussen 1941), or perhaps livestock numbers were reduced to a point where social tolerance reached acceptable levels. (4) Another reason for the increase in deer numbers was the change in vegetation composition as a result of overgrazing, particularly in the foothill regions or the winter ranges. As Julander (1962) noted, livestock overgrazing resulted in (a) a serious loss of perennial grasses and palatable forbs, and (b) because of the reduced competition from the herbaceous species, a large increase in several shrub and tree species. Many species which increased were not used by cattle in significant amounts but were valuable to mule deer on winter range.

All of the factors listed above contributed, in varying degrees, to the growth of mule deer herds. The relative importance of each, particularly food supply versus predator control on the Kaibab Plateau, has been vigorously debated by ecologists for years, making this the most well-known deer herd in the world. A prevalent theory is that irruptions in ungulate populations are initiated by a change in food supply or habitat (which occurred, in these cases, with overgrazing by cattle) and terminate with overgrazing or food depletion by the ungulate in question (Caughley 1970, 1979). Peek (1980) has developed a more thorough discussion of the role of habitat and/or predation as related to natural regulation of large ungulate populations.

The same principles appear to fit Coues white-tailed deer populations, although little specific information is available. There was an apparent decline at the turn of the century followed by peak populations in the late 1920's and early 1930's (Knipe 1977).

Thus, the vegetation structure and composition that resulted in part from overgrazing by livestock created conditions that permitted deer populations to increase. These circumstances, however, have been too broadly applied. It is true that overgrazing has been beneficial to deer on winter and fall ranges (Julander 1962). Winter range is a limiting factor for deer populations in much of the West (Klemmedson 1967). Therefore, when these ranges were "improved" as previously described, deer populations responded accordingly until checked by their own numbers and forage demands.

However, mule deer also depend on early growing grasses and forbs in the spring and forbs in summer. Replacement of palatable forms by unpalatable ones means a decrease in nutritional yield on spring and summer ranges. Summer deer losses are generally light, even on the poorest condition range; however, animals going into the winter period in suboptimal condition stand less chance of surviving. A good condition summer range is necessary for maximum herd productivity (Julander et al. 1961, Pederson and Harper 1978).

Julander (1962) also pointed out that an overstory of shrubs does not offer sufficient protection to the soil if the herbaceous understorey is destroyed. The productivity of fragile range sites in many areas of the West may have been impaired for centuries because of soil loss associated with overgrazing, first by domestic livestock, then by excessive numbers of deer. Russo (1964), for example, mentioned that, in the late 1920's, Pleasant Valley and VT Parks on the North Kaibab were completely denuded and that local residents referred to this period as "the years when it was a dust bowl." Additional references were made to extreme gully erosion and evidences of sheet erosion along exposed slopes on both summer and winter ranges. We can only guess how such damage has affected the Kaibab's present deer producing capabilities.

Another argument against overgrazing is that resulting declines in productivity and plant diversity will intensify competitive interactions between livestock and big game, regardless of species. Bryant et al. (1979) demonstrated that on yearlong ranges in the Edwards Plateau of Texas white-tailed deer were favorably affected by excellent range conditions. Increased grass availability under these circumstances apparently reduced the impact domestic sheep and goats had on forbs, a forage class important to deer. On winter ranges, increases in shrub growth would not benefit deer if the herbaceous layer was significantly altered or reduced. This would cause livestock food selection to change to available forage—that which is being consumed by deer. This relationship will be further discussed in the section on Competition for Forage.

Even if a change in vegetational composition is proven advantageous to some wildlife species, it will be detrimental to others. A classic example in the Southwest is the mesquite-invaded Semidesert Grasslands. Increases in shrub/small tree growth have created deer habitat by providing cover. Gambel quail and javelina have also benefited by this change. However, pronghorn antelope and scaled quail, which prefer open grasslands, have largely disappeared (Gallizioli 1977). The masked bobwhite quail may have been extirpated from this region because of the change in vegetative structure (Brown and Ellis 1977).

Little is known about elk response to these conditions prior to the 20th century. Reasons for their original decline are nebulous, but the decline is generally attributed only to the settlement of the West (Murie 1951). Thus, comments on effects of excessive overgrazing by livestock on elk are speculative. Wagner (1978) has suggested that although elk are more cosmopolitan in their selection of forage than are most wild ungulates, their relative dependence on grass would make them more vulnerable if habitats were converted from grass to shrubs, especially on summer range. Wagner (1978) also hypothesized, using peripheral evidence, that the successful reintroduction of elk in many areas may be attributed in part to improved range conditions resulting from reduced livestock numbers on forest lands.

Most of our attention thus far has been directed to overgrazing by livestock, primarily because this was one of the most widespread causes of vegetation change from 1880 to 1930 and because desired improvements have not been realized in many areas even today. However, wild ungulates certainly have the ability to overgraze or overbrowse. Such conditions have been noted for deer in general (Leopold et al. 1947), and for mule deer (Rasmussen 1941, Smith 1949, Julander 1962), Coues deer (Day 1964), and for elk (Croft and Ellison 1960).

It can be concluded that statements implying that overgrazing creates wildlife habitat are applicable only in a few, specific cases. Moderate overgrazing by livestock can improve deer

winter and fall ranges, but may degrade spring and summer ranges. Even on winter range, excessive overgrazing results in a loss of plant diversity and overall productivity of the site. Gallizioli (1977) has suggested that only the early stages of overgrazing be considered beneficial. Ranges used on a year-long basis by deer are apparently enhanced by attaining and maintaining excellent range conditions (Bryant et al. 1979). In other cases, where grasslands have been converted to shrublands (e.g., Semidesert Grasslands), deer habitat has been created or improved by the development of cover. Justification of overgrazing in this respect is questionable, however, particularly when costs are considered. Drastic alterations mean a loss of habitat for other wild species and, with the decline of herbaceous cover, reductions in the amount of forage for livestock.

Competition for Forage

Competition for forage between big game and domestic livestock is difficult to assess, despite the fact that it is a more direct interaction than the response of wild ungulates to habitat alteration induced by overgrazing. Typically, competition has been evaluated both directly (through comparison of relative food and range use habits or relative utilization of various forage plants on areas used individually or in common) and indirectly by inference (through evaluation of trends and changes in composition, vigor, and abundance of plants as a result of game, livestock, and dual usage) (Mackie 1976). Julander (1952, 1955) noted several factors affecting degree of competition in areas grazed by deer and livestock, including topography, water, range type, home range, instincts of deer, management of livestock, intensity of use, and season of use. He further noted other factors such as natural forage preferences, forage composition, intensity of use, and season of use, that affect degree of overlap in species grazed on common-use range. Cole (1958) emphasized that it is not necessary for animals to use the same area or forage plants at the same time for these to be in competition. Moderate to heavy utilization of browse species by livestock in the fall affects the amount of browse forage available to big game during critical winter-early spring periods.

To better understand competition for forage, a discussion of factors affecting extent of overlap of areas and plant species used in common is appropriate. These factors are dependent upon other animal, plant, and environmental influences. Very few big game studies in the Southwest assess the problem of forage competition between big game and livestock; thus most of the work presented is from studies conducted in other western states.

A few researchers (Wright 1950, Kimball and Watkins 1951, Julander 1955, Merrill 1957, Smith 1961, McMahan 1964, Mackie 1970, 1976, McKean and Bartmann 1971, Lauer and Peek 1976) have related forage competition to range condition. In general, these studies indicate an inverse relationship between competition and range condition. Competition is likely to be greatest under poorer range conditions. A good comparative assessment of forage competition studies is difficult because not all studies are related to existing range conditions such as plant composition, plant health, production, and stocking intensities of all classes of big game and livestock.

Results of some studies indicate a direct relationship between stocking intensities and competition for forage. In Texas, Merrill (1957) reported competition between white-tail deer and sheep, goats, or cattle is greatest under heavy stocking rates. Deer use of pastures diminishes under heavy yearlong grazing by sheep and goats. The same relationship was observed between deer, elk, and cattle by Skovlin et al. (1968) and Skovlin

and Harris (1974) in Oregon. In Colorado, dietary overlap increased between deer and sheep or cattle on pinyon-juniper range with increases in stocking (McKean and Bartman 1971).

On the contrary, Leopold et al. (1951), in California, found little evidence of serious competition despite high deer and cattle stocking rates, and McKean (1941, 1971) recommended grazing in combination by deer, sheep, or cattle under moderate rates without adverse effects to the range or animals. Skovlin et al. (1968) recommended conservative cattle stocking at light or moderate levels on ponderosa pine-bunchgrass ranges of northeastern Oregon. Heavy cattle use discouraged big game use and depleted forage resources. Moderate cattle grazing provided full use of forage with little or no effect on deer use, but discouraged elk use. Light use also discouraged elk use, but a better balance among grazing animals resulted.

Forage production influences the extent of dietary overlap. Deer have been reported to suffer from livestock competition during low forage production periods in Texas (Merrill 1957) and Arizona (White 1961). In Montana, Mackie (1970) noted a strong potential for increased elk-cattle interactions during periods of reduced forb growth.

Competition for forage has been reported to be significant for all seasons on different habitat types by various researchers (Wright 1950, Kimball and Watkins 1951, Davis 1952, Smith and Julander 1953, McMahan 1964, Lesperance et al. 1968, Hansen and Reid 1975). Others indicated only a strong potential for competition. Competition between elk and livestock reportedly was significant during summer (Pickford and Reid 1943, Hansen and Reid 1975).

Direct competition may result when both classes of animals occupy the same feeding areas at the same time. Indirect competition may be significant when livestock consume forage that is needed by big game at a later season (Willms et al. 1979). Thus, the management of winter range areas exclusively for big game becomes paramount in some instances.

The distribution of livestock influences degree of overlap and can be affected by habitat type, topography, home range, kinds of livestock, forage conditions, water supply, and weather. Behavioral interactions are also important and will be discussed separately. Some habitat types, such as sagebrush-wheatgrass sites, have been used intensively by deer, elk, and cattle (Mackie 1970, Kombercc 1976). Julander (1952) reported deer use steep, rough terrain in Utah almost exclusively. Mackie (1970) noted extensive elk movements in response to changes in forage availability and distribution of livestock grazing. Julander (1955) reported suitable forage as a major factor affecting deer distribution on summer range and desirable forage, exposure, and cover on winter range. He further noted cattle distribution is influenced mainly by steepness of slope and availability of water and forage.

The degree of competition for common forage plants is influenced by natural forage preference, forage composition, intensity of use, season of use, range condition, and production. These factors were discussed in previous sections. In general, competition for forage between cervids and livestock is largely dependent on extent of dietary overlap and availability of foods. Competition is greatest between elk and cattle (Neff 1979b), and between deer and sheep or goats (Smith and Julander 1953, Merrill 1957) because of similar forage preferences.

Behavioral Interactions

Little is known about interspecific behavioral relationships between cervids and livestock. The information available on these relationships comes from studies of four general types:

ranching operations, stocking intensity, grazing systems, and site preferences. Evidence of social intolerance has not been quantified and often only inferences of potential competition have been made (Pickford and Reid 1943, Smith and Julander 1953, Stevens 1966, and McKean and Bartmann 1971).

The intraspecific behavior of deer can be affected by routine livestock operations. Does and bucks have different behavioral mechanisms for handling disturbances from cattle roundups. Bucks show the greatest response and usually react with flights into adjacent pastures away from the disturbance. Does exhibit greater home range fidelity and usually take a circuitous escape route. Escape frequently takes does out of their home range, but they return within a few hours. The long-term effects often include enlargement and/or complete shifts of home ranges in response to disturbance of repeated roundups (Hood and Inglis 1974). Rodgers et al. (1978) noted similar temporary displacement of deer resulting from livestock roundup activities in Arizona.

Social intolerance and area exclusion of deer by cattle have been indicated by McMahan (1966), Firebaugh (1969), and Dusek (1975). On the other hand, Hungerford (1970), Komberec (1976), and Ward et al. (1973) reported observations of deer and cattle feeding in close proximity with no obvious social antagonism. In Wyoming, Strickland (1975) reported 90% of the deer observed were 1.6 km away from cattle and about 8% were seen near cattle and elk with no obvious social strife.

The social tolerance of elk for cattle appears to be less than that of deer for cattle (Jeffrey 1963, Dalke et al. 1965b, Mackie 1970, Komberec 1976). This intolerance has been related to stocking intensities and grazing systems. Elk use diminishes significantly with moderate and heavy cattle grazing. Elk prefer light cattle stocking rates on season-long ranges or deferred-rotation with heavy cattle stocking. Deer prefer deferred-rotation units above season-long (Skovlin et al. 1968). Mackie (1978) suggested that elk are displaced by cattle and withdraw into habitats important to mule deer. Insufficient evidence makes it impossible to determine whether avoidance responses are caused by effects of cattle on forage or by the presence of cattle themselves. Social tolerance limits are unknown and speculative at this time.

In general, differences in grazing site preferences between deer, elk, and livestock have been reported to overlap, but significantly different habitat types were used (Welch 1966, Komberec 1976). Cattle usually prefer to graze bottoms, flats, and other gentle terrain, whereas deer are frequently observed on slopes and more rugged country. It is uncertain what behavior mechanisms are involved. Do deer actually prefer these sites or are they being displaced by cattle? It is possible that the relationship is a result of an adaptive response to environmental forces by deer.

In summary, knowledge of interspecific behavioral relationships between big game and livestock is limited. It is difficult to assess interactions, because most studies only report observations and make subjective evaluations.

Range Improvement Practices and Cervid Response

Millions of acres of western rangelands have been manipulated in one form or another in attempts to (1) increase forage production for livestock, (2) improve watershed conditions, (3) increase water yield, or (4) improve wildlife habitat.

The perceived need to convert deer habitat is largely based on the premise that deer thrive in the vicinity of immature, unstable, or disturbed forest or shrubland where terrain provides a variety of slope aspects which separate different vegetation

types, as opposed to extensive areas of homogeneous vegetation such as dense and mature woodland, brushlands, or grasslands.

Improvement practices can be categorized as mechanical manipulation, chemical control, fertilization, and water development. The extent to which these practices benefit deer and elk is uncertain. A review of cervid response to these improvement practices follows. In most instances, data on deer and elk use were collected using the pellet group count technique and are subject to scrutiny as previously discussed.

There has been a tendency to apply control techniques based on research from one vegetation type to a different but structurally similar type. A southwestern example is the Great Basin Conifer Woodland (pinyon-juniper) and the Madrean Evergreen Woodland. Although there are similarities in structure, differences in flora and climate could result in different responses of vegetation and animals. Such programs should be approached with caution because there is no evidence that suggests ecological principles applicable to one are equally applicable to the other.

Mechanical Manipulation

Large areas of southwestern pinyon-juniper woodlands have been converted to grasslands by various mechanical methods using heavy equipment to chain, cable, crush, disk, etc. (Fig. 8). In Arizona, more than 1.2 million acres of pinyon-juniper were treated from 1950 to 1961 (Cotner 1963). The following shows the approximate area (thousand acres) and proportion (percent) of pinyon-juniper chaining on lands administered by the BLM in eight western states during the period 1960-1972 (Aro 1975):

| State | Area | Proportion |
|-------------------------------|------|------------|
| Utah | 257 | 50 |
| Colorado | 93 | 18 |
| Arizona | 61 | 12 |
| New Mexico | 46 | 9 |
| Nevada | 43 | 8 |
| Oregon, Idaho, and California | 14 | 3 |
| Total | 514 | 100 |



Fig. 8. A pinyon-juniper woodland area that has been mechanically cleared with undisturbed wooded areas retained for wildlife. These practices have generally not benefited deer because the cleared areas are too large.

Little (1977) makes a distinction between the terms control and conversion. Control applies to treatment of areas where juniper has invaded. Conversion emphasizes the treatment to change areas of "natural" pinyon-juniper woodland to grassland. The term conversion will be used in the following discussions.

Swenson (1977) evaluated responses of wildlife to pinyon-juniper conversions in the Southwest. He concluded that type conversions did not greatly improve wildlife diversity or numbers and often resulted in reduced deer use of the modified environment.

Mechanical treatments on 5,800 acres of pinyon-juniper woodland on the Fort Bayard Watershed in southwestern New Mexico showed that clearing large areas decreased deer and elk use (Short et al. 1977). Deer and elk use increased on areas where islands of living trees were left on north aspects and steep slopes (Reynolds 1972). Mechanical and chemical treatment of 5,200 acres resulted in no significant difference in deer use of converted Utah juniper areas and adjacent standing juniper woodland in Arizona (Neff 1980). McCulloch also reported (in Reynolds 1972) on assessment of deer response to conversion on six different areas. He found a positive response on one and neutral or negative responses on the others.

Most studies in other parts of the West produced similar findings. Terrel (1973) studied three chained pinyon-juniper areas in Utah and found: (1) deer use increased on natural areas, (2) no increases in deer numbers were correlated with treatments during the 12-year post-treatment period, and (3) increases in deer use of chainings were more closely related to increases in shrub densities than any other variable measured. No significant differences in deer use of chained versus unchained areas were noted in Oregon (Winegar 1978) and studies on four conversions in Nevada demonstrated variable findings; deer use increased on one, decreased on another, and did not change on two (Swenson 1977). Minnich (1969), working in Colorado, consistently found more pellet groups within chained clearings, however.

Reasons for variable deer response to mechanical treatments of pinyon-juniper types are likely related to a variety of environmental conditions, especially range site, original stand density, and treatment imposed. Forage production including all classes—forbs, shrubs, and grass—increased after control of trees (Arnold et al. 1964, Clary 1971) with few exceptions,

which have been related to stand density and site conditions (O'Rourke and Ogden 1969). Production of cliffrose increased 3.5 pounds/acre because of stimulated crown growth, but there was no increase in cliffrose regeneration 11 years after control in a sparse (15 trees/acre) pinyon-juniper stand (McCulloch 1966, 1971). However, casual observation indicated a good potential to increase cliffrose populations by bulldozing dense stands (>100 trees/acre) (McCulloch 1971). Although the practice tends to benefit deer by increasing the amount of succulent spring forage (Terrel 1973), its value in improving deer winter range is not known because of variation in shrub response (Terrel and Spillet 1975).

Size or proportion of the area to be treated also influences value to deer. If cleared areas are too large, they can become unacceptable to feeding deer because of distance from security cover. Terrel and Spillet (1975) suggested the proper approach would be to leave strategically spaced openings in pinyon-juniper, rather than to leave islands of trees in chained areas.

Optimum habitat could be created by spot-thinning mature stands to a crown cover of 15% and a density of 50% and seeding the interspaces with native grasses and shrubs (Terrel and Spillet 1975).

Several researchers have recommended guidelines for conversion of pinyon-juniper woodlands. These guidelines are presented in Table 5 to show the variation in recommended size of treated and untreated areas, and proportion of area to be treated. The total proportion varies from about 20% to 75%. The suggested size of clearing also varies from 100 feet to 0.25 mile. Several other criteria are listed in the table.

Dense stands of chaparral can be treated to increase grass production and facilitate penetration and use by grazing animals. Chaparral has been successfully converted to grassland by rootplowing in Arizona. Rootplowing and reseedling with weeping lovegrass and Lehmann lovegrass have increased forage production up to four times (Pond 1961). Later Urness (1974) found that deer did not respond favorably to rootplowing and reseedling. He reported deer spent only 25–50% as much time on three 80-acre treated areas as in adjacent brush fields. The treatment provided usable forage, but the size and shape of clearings may have been undesirable for deer.

One possible shortcoming in most of these studies is that evaluations were based on pellet group counts. These, as previously mentioned, may not indicate overall value of a habitat

Table 5. Comparison of recommended guidelines for conversion of pinyon-juniper woodlands.

| Source | Lamb and Pieper (1971) [N.M.] | Lechenby (1977) [Oregon] | McCulloch (1979) [Arizona] | Terrel (1973) [Utah] | Terrel and Spillet (1975) [Utah] |
|--|----------------------------------|-----------------------------|-------------------------------|-------------------------|-------------------------------------|
| Proportion of total area to be treated | 50–75% | <33% | ≤213 acres per sq. mi. | <20% of winter range | <25% of winter range |
| Size of clearing | 1/8–1/4 mile | <120 meters | 100'–660' | ≤0.2–0.4 mile | <0.2 mile |
| Size of untreated woodland | >1/8 mile (25%) | 40% cover 60% forage | 100'–660' | 0.5–1.0 mile in dia. | 0.5 mile |
| Sites avoided ¹ | a,b,c,d,e,g,k | | a,b | f | c,o |
| Suggested site treatment ¹ | l,m | j | | l,n,h,i | l,m,n,j |
| Slash cover | 25 tall shrubs/acre | | | ≤20% | ≤20% |

¹(a) steep slopes
(b) rocky
(c) ridges
(d) shallow soil
(e) slopes >15%

(f) slopes > 20%
(g) recreation sites, highways
(h) valleys
(i) protected hillsides
(j) numerous small openings

(k) NE exposure
(l) southerly exposure
(m) easterly exposure
(n) westerly exposure
(o) northerly exposure

treatment because of possible bias against feeding areas. As Patton and Judd (1970) suggested, the cover type in which deer and elk spend most of their time may not be indicative of the quality and quantity of forage in their diet. Smaller amounts of time actually spent in foraging areas may be much more important.

Neff (1979b) recommended that 33—50% of a management unit be treated, with numerous cleared areas 20-40 acres in size and with broad undisturbed corridors of chaparral including gullies, rocky areas, natural travel routes, and buffer zones around water. Reynolds (1972) recommended somewhat different criteria. He suggested treating no more than 50% of a watershed, with numerous small areas (5 acres), and leaving cover patches of at least 40 acres no more than 1/10 mile apart.

Mesquite brushlands are an important habitat for desert mule deer and white-tailed deer. Extensive control of mesquite would deplete fruit and browse production as well as eliminate a cover type (McCulloch 1972, Short 1977). This does not preclude establishing smaller openings in mosaic patterns to create forage openings, especially in dense, extensive stands—provided critical habitats, such as the pseudoriparian, mesquite-dominated drainageways identified by Rodgers et al. (1978) are not cleared.

Texas studies have indicated that white-tailed deer prefer native brush stands when compared to rootplowed and reseeded ranges, especially under drought conditions. This preference was related to greater abundance of fruit and leaves of brush species (Davis and Winkler 1968). The cleared area was large (2,770 acres), however, and the authors concluded that small acreages of brush-controlled range may be preferred, particularly in dense, extensive, native brush stands. Results of other Texas studies indicated that, when habitat changes center around removal of preferred browse, white-tailed deer adapt to these changes if adequate cover is retained, whereas populations decline when cover is greatly reduced. These results suggest that managers should selectively remove dense stands of brush while trying to maintain mesquite stands infested with mistletoe, a preferred deer food item (Quinton et al. 1979).

Mesquite encroachment is considered detrimental to grass production on semidesert grasslands (Ames 1966, Martin 1975). Experimental work on control of mesquite, creosote bush, other shrubs, and cacti continues. Efforts must be made to assess and mitigate these impacts on wildlife habitat.

Chemical Control

Experimental work on chemical control of pinyon and juniper has been summarized by Johnsen (1967) and Evans et al. (1975), respectively. Few studies have included assessments of effects of chemical brush control on big game. In Arizona, McCulloch (1962) and Neff (1972) reported on studies conducted on the Beaver Creek watersheds. Neff (1972) found that deer use was significantly higher on a 362-acre, herbicide-treated (2, 4, 5-T, 2,4-D, and picloram) unit than on an adjacent 332-acre, mechanically treated unit. The herbicide-treated watershed had three kinds of habitat available: bulldozed fuel break, standing dead juniper, and standing live juniper woodland outside the spray zone. The positive response by deer was due to the resultant interspersed small areas of different types of vegetation (Neff 1972) and to increased production of preferred forage, coupled with retention of cover (Clary et al. 1974).

Extensive experimental chemical control of chaparral plant species has been done by various researchers in Arizona, but little quantitative information is available relative to wildlife use of altered habitats.

Elk and deer use of two, adjacent, 67-acre Gambel oak sites was evaluated in Colorado. One area was sprayed with 2,4,5-TP, and the other was a control. Two years after treatment, elk density increased 109% and 19% on the sprayed and control areas, respectively, based on pellet group data. Five years later, elk use approximated pretreatment figures. Deer exhibited only a minor response to the treatment, which may have been due to the extensive elk use and/or a general decline in deer numbers throughout the area (Kufeld 1977).

Effects of chemical control of sagebrush or saltbrush on wildlife have not been studied in the Southwest. However, work has been done in other western states. In western Colorado, Anderson (1969) reported mule deer use of sagebrush habitats declined after treatment with 2,4-D. In Wyoming, Wilbert (1963) reported chemical control of sagebrush has significant effects on distribution of elk. Treated areas received 89% heavier use than untreated ones. Ward (1973) reported elk do not change their calving behavior or feeding habits in response to sagebrush spraying.

Guidelines for treating sagebrush types in New Mexico have been established by Lamb and Pieper (1971).

Fertilization

Fertilization studies conducted on the Fort Stanton Cooperative Experimental Range in New Mexico showed mixed results. Anderson et al. (1974) reported no response in leader length of wavyleaf oak and fourwing saltbush to fertilization with urea and ammonium sulfate but did find significant increases in protein content of oak leaves. Hairy cercocarpus showed a significant response in leader length to urea fertilizer. Greater deer use resulted on areas fertilized with urea for 2 years, but no significant use resulted from application of ammonium sulfate.

On the contrary, wedgeleaf ceanothus, deerbrush ceanothus, birchleaf cercocarpus, and fourwing saltbush responded favorably to fertilizers in California. Mule deer exhibited a preference for sulfur-fertilized wedgeleaf ceanothus plants. Fertilizer combinations of nitrogen, phosphorus, and sulfur were effective in increasing deer's preference for mariposa manzanita, a less preferred browse species (Schultz et al. 1958, Gibbens and Pieper 1962). Similar results were obtained for antelope bitterbrush and big sagebrush in Utah (Bayoumi and Smith 1976).

Thus, information is minimal on use of fertilizers to improve the quantity and quality of forage and, subsequently, influence distribution of big game. Other fertilization experiments have been conducted mostly on grassland ranges for the benefit of livestock. Only a few studies have been on shrub-dominated rangelands (Schultz et al. 1958, Gibbens and Pieper 1962, Brown and Mandery 1962). Most have been designed to improve livestock distributions, increase forage production, and improve nutritive values of plants, but they may also benefit big game.

Range fertilization in the Southwest is in the experimental stage. Humplrey (1962), Scotter (1980), and Carpenter and Williams (1972) summarized the extent of fertilization trials in the Southwest and nearby regions. These studies were primarily limited to grassland areas.

It is uncertain what effects fertilization has on range use by cervids. Positive responses of vegetation to fertilization include:

- (1) Extended period of green forage availability (Holt 1959, Humphrey 1962, Bear 1978).
- (2) Changed chemical content of forage (Burzlaff et al. 1968, Carpenter and Williams 1972).

(3) Increased production (Cospers et al. 1967, Carpenter and Williams 1972, Bear 1978).

(4) Changed botanical composition (Goetz 1969, Carpenter and Williams 1972).

(5) Increased plant growth and vigor (Lavin 1967, Honnas et al. 1959).

(6) Changed site ecology (Carpenter and Williams 1972).

(7) Increased forage palatability (Thomas et al. 1964, Carpenter and Williams 1972).

Additionally, fertilizers can be used to increase cover for deer (Curlin 1962, Gibbens and Pieper 1962) and cause selective thinning and browsing of some species (Gibbens and Pieper 1962). The degree to which cervid behavior and distributions are changed by altering habitats with fertilization has not been adequately quantified.

Cervid use of fertilized ranges could be affected positively or negatively, depending on the botanical composition. Some researchers report a reduction in forb composition following fertilizer applications (Cospers et al. 1967, Basile 1970), while others reported that forbs and cool-season grasses benefited at the expense of warm season species (Huffine and Elder 1960, Powell and Box 1967). Thus, in areas where cervids make extensive use of forbs and cool-season grasses, as in springtime, fertilizers could be used to enhance or reduce use, depending on local conditions.

Research on fertilization of native rangelands to improve forage quality and quantity has yielded variable results. Several researchers report improved forage conditions following treatments (Thomas et al. 1964, Carpenter and Williams 1972, Powell et al. 1979). On the other hand, Probasco and Bjugstad (1978) determined acid-detergent fiber and dry matter digestibility values for unfertilized tall fescue were superior to fertilized fescue for white-tailed deer.

Elk have been attracted to fertilized hayfields in Washington (Brown and Mandery 1962) and fertilized clearcuts in Oregon (Geist et al. 1974). In fact, extensive utilization on treated areas made it impossible to determine the effects of fertilization. Other researchers have reported the response of vegetation to fertilization and related results to improved game ranges and better wild ungulate distribution (Bayoumi and Smith 1976, Carpenter and Williams 1972).

Better livestock and cervid distribution is possible through use of fertilizers (Hooper et al. 1969). This has been demonstrated in Washington (Brown and Mandery 1962) and Wyoming (Smith and Lang 1958). However, a comprehensive program of basic research is needed and has been suggested by Martin (1975). General guidelines for fertilization of rangeland in Arizona have been suggested by the Arizona Interagency Range Technical Sub-Committee (1973). Our present knowledge of effects of fertilization on cervids and their habitat is too limited to make general recommendations.

Water Development

Water developments can be an important management tool to improve wildlife habitat and achieve better animal distribution. Use of big game guzzlers has been evaluated in New Mexico, and elk showed a preference for water provided from human-made devices to naturally occurring water. Deer seemed to prefer naturally occurring water, however. Deer numbers decreased in the immediate area of human-made water sources that were turned off (Johnson 1962). Development of permanent water sources affected deer movements and carrying capacity. Mule deer use of New Mexico range decreases as distance to

water increases (Wood et al. 1970). Similar results were reported by Swank (1958) in Arizona. Deer concentrate around water sources. When water becomes limited, deer numbers decrease.

Wood et al. (1970) suggested that maximum distances between water sources should be 2.5-3 miles. This would vary with the topographical situation. Roberts (1977) presented several estimates ranging from spacings of 0.5 mile to spacings of 3 miles apart. Lamb and Pieper (1971) suggested such developments be no more than 2 miles apart.

A loss of watering sites causes deer to abandon areas and concentrate where water remains. In southern Arizona frequent but limited die-offs of white-tailed deer coincide with the normally dry season, May and June, but this problem is compounded during drought years when free water became even less available (Hanson and McCulloch 1955).

Reseeding

Pinyon-juniper conversions are often followed by reseeding of native and introduced forage species. Various native and introduced species of cool season grasses have been artificially established on several areas in Arizona, but efforts to establish browse species and varieties not native to the immediate site on pinyon-juniper conversion projects have failed (McCulloch 1979). Hungerford (1965) found mule deer make extensive use of orchardgrass, smooth brome, timothy, elderberry, and alfalfa reseeding on Kaibab summer range.

The literature indicates that, in general, reseeding projects have little significant effect on big game distribution. In most instances, effects are obscured by the increased use of treated areas by livestock.

Extensive efforts to restore cervid winter range have been made in other western states. Plummer et al. (1968) made detailed recommendations for reseeding shrubs in various vegetation types in Utah. Their recommendations were based on procedures and treatments that proved to be effective in Utah and may be applicable in the northern parts of Arizona and New Mexico.

Fuelwood Harvest

Recent increases in energy costs have resulted in increased demands and accelerating markets for fuelwood. Harvesting of pinyon, juniper, mesquite, oak, and other species both on a commercial and noncommercial basis, has thinned or cleared many woodland thickets. Although this increased demand has resulted in additional work for land managers, it also has created a tool by which managers can manipulate vegetation types where, previously, economic constraints prevented such activity. A significant limitation is that it is difficult to develop precise prescriptions for area treatment because of difficulties in monitoring activities of noncommercial cutters. This problem may be partially overcome on public lands by assigning more precise prescriptions (e.g., thinning, selective cutting, group selection) to commercial fuelwood cutters in the same general way as commercial timber harvests. Less critical treatments, such as patch clearcuts and some thinning operations, could be opened to the public, noncommercial harvest. Either option would be feasible for private land managers, depending on available expertise and management objectives. More precise prescriptions would also involve surveys and/or tree marking activities, which mean additional costs.

Management goals regarding size of area to be treated, size of openings, thinning levels, etc. would be similar to those previously discussed with other conversion methods.

Livestock Grazing as a Wildlife Habitat Improvement Technique

Over 30 years ago Smith (1949), working in Utah, noted significant differences in vegetation composition of pastures grazed either by livestock or wintering mule deer. Essentially, he found that a pasture utilized by deer, but protected from livestock grazing for 11 years, contained an abundant and vigorous cover of perennial grasses and forbs, but that shrubs, particularly big sagebrush, were either dead or weakened because of excessive browsing by deer. In a pasture heavily grazed by livestock, cattle and horses, big sagebrush was abundant and vigorous, but perennial herbaceous forms were scarce. Although not the first study of this type, it did provide one indication of how effects of grazing by one class of animal might benefit another.

Other studies have been conducted in Utah to determine how to use livestock grazing to enhance ranges used by mule deer in winter. Efforts have been directed at balancing utilization of a key winter deer browse, antelope bitterbrush, with utilization of the herbaceous forage. Smith and Doell (1968) found that such ranges can be moderately grazed by cattle from late May to early July without antelope bitterbrush being utilized. Fall grazing also is feasible during years of above normal fall precipitation. Jensen et al. (1972) conducted the same type of study with domestic sheep and derived similar conclusions.

Although they did not mention range condition, the investigators apparently desired to maintain the existing vegetative composition. Their intent was to remove as much herbaceous material as possible to reduce its competitive effects on antelope bitterbrush production, but not to the extent that bitterbrush was used by the domestic animals. This would insure a maximum supply of a preferred browse for wintering mule deer. Smith and Doell (1968) thought it advantageous to increase use of the herbaceous material by cattle, even though it would mean a corresponding slight increase in antelope bitterbrush utilization. They were concerned that a moderate stocking rate would not result in a sufficient impact on herbaceous species to maintain a stable floral composition. Jensen et al. (1972) did not note any short-term vegetation trends but were apprehensive regarding the long-term effects of single-season use. Laycock (1967), for example, had noted rapid deterioration of threetip sagebrush-grass ranges because of heavy spring grazing.

These studies indicate that a grazing plan designed to maintain stable browse forage for wintering mule deer and herbaceous forage for livestock must meet two requirements: (1) utilization of browse by livestock must be kept at a low level, and (2) herbaceous forage must be utilized at a season and intensity that will maintain it, but not permit it to increase at the expense of shrub production.

Smith et al. (1979) studied the effects of this grazing system on food habits and noted that wintering mule deer feeding on pastures grazed by sheep the preceding spring used more herbaceous material and fewer shrubs than did mule deer in a pasture not grazed by sheep. Fulgham (1978) determined that there are no significant differences between the two treatments in overall nutritive quality of forage ingested by deer.

Concurrently, Malechek et al. (1978) studied nutrition and production of the sheep used to improve mule deer winter range. They found no differences between duration of grazing (short-term or season-long) but noted that diet quality, forage intake, and lamb weights were lower under heavy than under moderate stocking rates. Such responses should be known because wildlife biologists considering use of livestock as a habitat

management tool must remember that livestock will probably be owned by a private individual intent on making a living from livestock production. Any special management for wildlife habitat improvement must be compatible with that intent (Longhurst et al. 1976).

Elk and cattle are generally thought to be quite competitive because of similarities in food habits; however, cattle grazing can be manipulated to improve elk winter range. Anderson and Scherzinger (1975), in northeastern Oregon, developed a grazing system keyed to select physiological and morphological features of plants. Basically, cattle grazed the unit in a rotation system during May and June. Specifics of the system were directed to two primary objectives: (1) to remove excessive amounts of old plant material to prevent formation of "wolf plants," and (2) to manipulate the physiology of forage plants to improve the nutritive value at maturity. Despite reported increases in grazing ungulates, both elk and livestock, there has been a concurrent improvement in range condition, from fair to good or excellent depending on range site.

With these few exceptions, directed at big game winter range and using spring or summer livestock grazing, research has not directly addressed livestock grazing as a wildlife habitat improvement tool. Although there is often disagreement on the feasibility of this approach, there is a certain unity of opinion regarding our lack of knowledge on livestock-wildlife-habitat interactions (Mackie 1976, Urness 1976, Connolly and Wallmo 1981). We agree with Connolly and Wallmo's (1981) suggestion that, in reference to mule deer, livestock use can be beneficial to deer ranges under some conditions; however, if mule deer habitats, or other wildlife habitats, have benefited from past livestock grazing, such benefits have been fortuitous rather than planned.

Livestock grazing is one of the most ubiquitous human-induced influences in the West. Its presence has been a dominating factor in management of rangelands (including noncommercial forest lands) and a codominating factor, with timber management, on many commercial forest lands. Longhurst et al. (1976) concluded that livestock grazing is one of the few common denominators throughout the West which could have influenced deer range trends. They further stated that modification of grazing practices appears to be one of the best possibilities for improving deer food supplies. We submit that intentional manipulation of livestock grazing with emphasis on distribution factors, stocking rates, season-of-use, and resultant effects on vegetation has the potential to enhance certain wildlife habitats, particularly for deer and elk. Research has yet to determine not only how, when, or where it can be used but also specific circumstances under which it cannot be used.

Riparian Habitats, Livestock Grazing, and Wildlife Habitat

The value of riparian communities to nongame birds has been well documented (Carothers and Johnson 1975, Johnson et al. 1977, Szaro 1980); however, little information is available concerning its value to cervid species in the Southwest. Patton and Judd (1970) thought that, because of quantity and quality of forage produced, wet meadows were very important to deer and elk, even though they spent more total time in adjacent coniferous forests. "Dry washes" are important to mule deer in Semidesert Grasslands. These drainages, with associated vegetation, provide food as well as resting, escape, and travel cover throughout the year. In one study, the washes, actually pseudoriparian types, made up only 3% of the area but included 33% of all mule deer sightings (Rodgers et al. 1978).

Studies in other areas of the West have indicated similar importance of riparian habitats. Thomas et al. (1979c) reported that elk summering in the Blue Mountains of Oregon spend 40% of their time in riparian zones, which make up only 7% of the summer range. They suggested that these areas are attractive to elk because of the abundance of thermal cover and the microclimate produced by that vegetation. Similarly, Collins (1977), working in Utah summer range, noted that elk spend 44% of their grazing time and 50% of their time for resting and other activities in the wet meadow type, which makes up 3% of the total area. The wet meadow type was described as seldom exceeding 15 m in width, restricted to areas immediately adjacent to stream courses, and often densely covered with live and fallen trees, beaver dams, high shrubs, and tall grasses. Hayden-Wing (1979) studied elk utilization of forage in Idaho and found that, although the wet cover types immediately adjacent to streams or bogs are the most productive, the highest elk utilization was noted in what he called the moist cover type, which is between the wet and dry zones.

Although riparian habitats represent only 1% of the area of the Great Plains (Bjugstad 1977), they are important habitat components for mule deer. Severson and Carter (1978), working in two different study areas, noted that riparian areas account for 42% and 74% of all mule deer observations. The smaller figure was observed in an area that had an alternative cover type dominated by Rocky Mountain juniper, which accounted for 30% of the observations. Juniper was virtually absent in the other study area. The riparian habitats most utilized by plains mule deer are narrow, V-shaped types. The broad, flat, cottonwood-dominated floodplains are only lightly used by mule deer during late summer and fall (Allen 1968, Mackie 1970).

The value of riparian habitats to mule deer inhabiting forested, mountainous ecosystems has not been well documented. There is a general feeling among biologists that mule deer feed and seek shelter in riparian habitats on summer range, but that these are less critical than for elk. Thomas et al. (1979c) stated that riparian zones along streams acted as travel corridors between summer and winter ranges for deer and elk. On winter ranges, evidence indicates the most important types were those which provide available food such as south- and east-facing slopes (Loveless 1967). Although riparian zones can be important sources of food and cover, accumulating snow depths may prevent or reduce their use by mule deer.

In the central and northern parts of the West, white-tailed deer use the broad, flat, generally cottonwood- and willow-dominated floodplains to such an extent that it is considered their most important habitat (Allen 1968, Martinka 1968). Typically riparian, these "bottomlands" or floodplains were of variable width, from several yards to several miles, characterized by being flat enough to permit stream meandering, and often subject to cultivation. Although specific information is lacking for Coues white-tailed deer, mapped ranges of four does presented by Knipe (1977) show a definite orientation to riparian growth in Florida Canyon, Santa Rita Mountains.

Although many factors are responsible for the changes that have occurred in these habitats, much concern in the western states is directed at effects of grazing by livestock (Meehan and Platts 1978, Platts 1978, Severson and Boldt 1978, Cope 1979). The problem is further complicated in that livestock tend to concentrate in these habitats for a variety of reasons, including level or moderate topography, a lush understory, shade, and water. Hence, excessive grazing may be occurring in the ripa-

rian area when associated slopes and uplands are being only lightly utilized. The problem often results from improper distribution of livestock rather than overstocking.

Management of livestock depends on local circumstances. On some areas, particular attention to established techniques controlling distribution of livestock, such as salting, building alternative water sources on uplands, herding, trail construction, and fertilization, may be feasible. Elsewhere it may be necessary to exclude livestock grazing by fencing, but only if the riparian area is deemed particularly critical. Fencing cannot be considered a panacea for riparian habitat management because the long, narrow, and irregular shape of these features would require too many miles of fence. Prohibitive costs of construction and maintenance and detrimental effects of fences on wildlife through direct mortality and interference with movement patterns preclude extensive fencing as means of protecting riparian habitats.

The most promising solutions may involve site-specific grazing systems. There is peripheral evidence that indicates riparian areas at lower elevations, where grazing by livestock is year-long, can be protected or rehabilitated by a system involving rotation of summer and winter use. Reasons for this are that factors which tended to attract and hold livestock in these areas in summer (shade, water, and a green productive understory) are not operative during cold winter months. As a result, riparian types subjected to season-long summer use receive heavy damage, whereas those used only in winter receive little, if any, damage (Severson and Boldt 1978). Martin (1979), in central Arizona, observed that cattle do not graze riparian tree sprouts until after they have leafed out, and that browsing damage can be reduced by removing animals before or soon after the more palatable species (willows and velvet ash) put on leaves. This also indicates that use during the nongrowing season may not be as damaging.

This option is not available on high-elevation ranges, where only summer grazing is possible because of snow depths. Further, riparian vegetation, easily damaged by grazing animals, is essential for protection of coldwater fisheries resources in perennial streams. There is apparently no existing grazing strategy that is compatible with riparian zones bordering trout waters (Meehan and Platts 1978, Platts 1978) although the problem is being addressed in a current study (Skovlin et al. 1977).

Grazing or distribution problems associated with grazing are an important problem in riparian habitats, but we must acknowledge that grazing management alone may not suffice to halt degradation. Crouch (1978), working on the South Platte River in Colorado, noted that cover and height of the understory were about twice as great on an ungrazed area as compared to a grazed area, but there was a significant decrease in the number of cottonwoods on both areas. Crouch did not suggest any reasons for this. However, studies on the Missouri River floodplain in North Dakota led Johnson et al. (1976) to hypothesize that lack of seedling-sapling stands of cottonwood is the result of a reduction in the meandering rate of the river following dam construction and poor seedbed conditions in the absence of flooding. Ohmart et al. (1977) concluded that cottonwood declines along the Lower Colorado River in Arizona are also caused by cessation of annual flooding and meandering rates. However, they further indicated that introduction of tamarisk is of greater influence by noting that cottonwood communities would have lost dominance even without dam construction because of the aggressive and fire-adapted tamarisk.

Another case indicating effects other than grazing is illustrated by treated and untreated watersheds on the Three Bar Wildlife Area in central Arizona. The area was burned in a 1959 wildfire, after which one watershed (D) was allowed to recover naturally as the control and three others (B, C, and F) were subjected to various degrees of chemical and mechanical control of the resprouting shrubs (primarily shrub live oak, birchleaf cercocarpus, sugar sumac, and Emory oak). Water yields increased on all four watersheds immediately after the fire, but the effect had largely disappeared on D within 3 years. All watersheds subjected to chemical or mechanical treatment maintained significant water increases over the expected yield (Hibbert et al. 1974). Improved moisture conditions within the channel resulted in the establishment of a riparian community dominated by willows and cottonwoods in one of the treatment watersheds. The other two treated watersheds could not be evaluated because their channels were too short below the weirs where measurements were made. A corresponding increase of riparian vegetation in the control watershed has not occurred. The entire study area has not been grazed by livestock since 1947.

This illustrates one very important point: Although riparian habitats are physically discrete entities, they are not functionally discrete. Because they are long, narrow areas that represent concentration zones, they are easily affected by natural disturbance or management activities in adjacent habitats. Thus, problems seemingly unique to riparian habitats cannot be resolved by managing only that habitat, but conditions on adjacent, upslope areas must be considered.

Interactions with Other Ungulates

Elk/Deer

Feeding habits of mule deer and elk can be similar, especially on winter and yearlong ranges. Both species, for example, rely heavily on birchleaf cercocarpus and oaks during all seasons on yearlong range in southwestern New Mexico (Short et al. 1977). When deer and elk occupy the same ranges under these conditions and one or both exceed carrying capacity, competition for foods can become intense. Elk, with the physical advantage of size, are more efficient feeders. They can reach higher and appear more adept at scraping snow than are deer. In the Missouri River Breaks of Montana, Mackie (1970) reported the period of greatest conflict between the two species is spring and summer, both in terms of habitats being used and forages consumed. Morris and Schwartz (1957) noted a great amount of overlap in use of grass in both spring and fall. Most authorities agree that deer are less successful competitors than elk (Cliff 1939, Compton 1975, Mackie 1970). However, as Mackie (1976) later indicated, there is little quantitative support that such interactions are important factors in the ecology of either species.

Mule Deer/White tailed Deer

These species coexist in many parts of western North America, a coexistence that many investigators have attributed to differences in habitat preference (Martinka 1968, Krämer 1973, Hudson et al. 1976, Anthony and Smith 1977). Krämer (1973) suggested that, with low population densities, optimum mule deer habitat was free of white-tails, and optimum white-tail habitat was used by few mule deer.

White-tailed deer in Montana prefer deciduous woody-dominated habitat types at lower and intermediate elevations, whereas Rocky Mountain mule deer prefer bunchgrass/shrub habitats at intermediate elevations and pine/bunchgrass/shrub

types at higher elevations (Martinka 1968). Hudson et al. (1976) noted that white-tailed deer are found on gentle slopes at lower elevations in relatively uniform coniferous stands. Mule deer, however, occupy the more open, rugged country at higher elevations.

Coues white-tailed deer/desert mule deer habitat relationships may appear just the opposite of this, at least initially. Coues deer occupy rough terrain at higher elevations, desert mule deer the less rugged bajadas at lower levels. However, Hudson et al. (1976) indicated in their British Columbia studies that the apparent negative response to slope by white-tails is related more to associated vegetation than slope per se. In northern climes, the preferred vegetation, such as the alfalfa-willow types, grows at lower elevations on relatively gentle terrain, whereas in the Southwest, the preferred habitat types, Madrean Evergreen Woodlands and Montane Conifer Forest, are at intermediate and higher elevations.

Food habits of the two species are generally similar, but use of individual plant species overlaps significantly only in limited situations (Martinka 1968, Krausman 1978). Anthony and Smith (1977), working in two study areas in southeastern Arizona, found a high degree of overlap in food habits between the deer species, however. In the San Cayetano Mountains low distributional overlap and high similarity in food habits led them to conclude that the deer species are competitively excluding each other. The high overlap in distribution and food habits in the Dos Cabezas Mountains led to the conclusion that direct competition was operating in that area at that time and was undoubtedly a transient phenomenon. They suggested that factors responsible were vegetation changes, overgrazing by livestock, and/or range fire suppression. Desert mule deer apparently replace Coues white-tailed deer under these conditions. Although Anthony and Smith (1977) did not conduct a detailed population analysis, they noted that the pyramidal age distribution of mule deer suggested increasing numbers while Coues deer populations were declining. They also stated that mule deer were dominant over Coues deer in all behavioral interactions they observed.

Other Ungulates

Interactions between cervids and other ungulates are generally local or limited; hence, little specific information is available. Russo (1956) did not quantify competition between desert bighorn sheep and deer in Arizona but stated that parts of the bighorn range were inaccessible to deer; most of the palatable vegetation was found on overlapping ranges, however. He also noted several cases of apparent social tolerance. Desert bighorns and deer occupied the same habitats on the San Andres National Wildlife Refuge in New Mexico (Halloran and Kennedy 1949). These investigators noted considerable overlap in foods, particularly the dominants, Wright silktassel and hairy cercocarpus.

The range of the javelina overlaps both desert mule deer and Coues white-tailed deer. Although javelina feed primarily on cacti, grasses, and forbs, Sowls (1978) emphasized their preference for berries and mast, when available. Knipe (1977) thought javelina to be serious competitors with Coues deer for mast, particularly in years of poor mast production.

Feral and exotic ungulates may also cause local concern. Zarn et al. (1977a) concluded in a literature review that competition between mule deer and wild horses is minimal; however, deer always yield to horses by leaving an area. Little overlap in forage plants used by mule deer and horses was reported from

Colorado. Diets were so different they were considered complementary rather than competitive (Hubbard and Hansen 1976, Hansen and Clark 1977). A study in Canada revealed that, on a yearlong basis, feral horses used 87% of the sites also utilized by deer, while deer were found on 33% of the sites used by horses (Salter and Hudson 1980).

The few studies available on feral horse-elk relationships indicate a greater overlap of forage and area use than reported for deer. Dietary overlap between horses and elk has been reported at 42% on an annual basis in Colorado (Hansen and Clark 1977), while in Wyoming, seasonal overlaps were 26% in spring, 14% in summer, 70% in fall, and 50% in winter (Olsen and Hansen 1977). Comparable degrees of dietary overlap were noted in Canada, where it was also determined that horses used 93% of the areas occupied by elk and elk were found on only 6% of the sites used by horses (Salter and Hudson 1980).

Interactions between deer and feral burros are apparently minor; the scope of competition is usually small (McKnight 1958, Zarn et al. 1977b). An estimated 200-400 feral hogs are present on the Havasu National Wildlife Refuge along the Colorado River in western Arizona (Decker 1978), but no information is available on possible interactions with mule deer.

Land managers in New Mexico have introduced four ungulates: Barbary sheep, oryx, Persian ibex, and Siberian ibex. The first three now provide limited hunting opportunities (Decker 1978). All introductions have been into mule or white-tailed deer habitats; none have been into areas occupied by elk. Ogren (1962) stated that there is considerable overlap in food items used by Barbary sheep and mule deer, but there are also pronounced differences in kinds of habitats used. Barbarys frequent steepest canyon walls and benches just under caprock. Deer use lower and relatively more accessible slopes. White (1967) and Durham (1969) studied oryx at Red Rock, New Mexico. Both indicated oryx are primarily grazers and, although neither studied competition directly, White (1967) noted that mule deer and oryx exhibit no social intolerance. Durham (1969) also studied Siberian ibex at Red Rock and stated they are primarily grazers, a point refuted by Barraclough (1972) who indicated their primary foods at Red Rock, but in another pasture, were catclaw acacia, broom snakeweed, and mesquite. Barraclough (1972) and Woodroof (1972) studied Persian ibex at Red Rock and in the Florida Mountains, respectively. Both noted a diet high in shrubs. Woodroof (1972) pointed out that the three major species in the area (hairy cercocarpus, Wright siltkassel, and oaks) made up 75% of the ibex diet and 77% of the mule deer diet. He also noted, however, that the two species prefer different habitats. The ibex use steep, rocky, rugged terrain; and mule deer use more open, level terrain.

Fire as a Habitat Improvement Tool

Fire has been present in varying degrees in most, if not all, of the biotic communities of the Southwest for the last 10,000 to 12,000 years (Pase and Granfelt 1977). Until the coming of European man and his livestock, development of plant communities was largely controlled by climate, site, and fire. The importance of fire in shaping vegetation varied. Moist communities such as some riparian habitats, wet meadows, alpine tundra, and upper elevations of spruce-fir were often too wet to burn. Conversely, in very arid environments such as deserts, scarcity of fuels prevented fires. Where sufficient fuels accumulated and where there were seasonal dry periods, fires did influence the development of vegetation (Humphrey 1962, 1963; Pase and Granfelt 1977).

Although wildfires will be mentioned occasionally in this section, we are mainly concerned with prescribed burning. Prescribed burning is the use of fire as a management tool under specified conditions for burning a predetermined area (Kothmann 1974). Controlled, planned, maintenance, and reclamation burning are common synonyms. Wildfires often occur during hot, dry periods and, therefore, tend to be severe. Benefits from such fires are generally incidental and are often offset by detrimental results. Prescribed burning, however, can be implemented under a variety of conditions depending on program objectives. Burning programs have been initiated to increase forage by eliminating undesirable plants, to reduce fuel loads with the long-term goal of reducing the chance for wildfires, to improve water yields, and to improve wildlife habitat (Pase and Granfelt 1977).

Prescribed burning can be used to improve or create wildlife habitat by creating diversity and edge and by improving the quantity and quality of food. Diversity and edge enhancement for cervid habitat is generally accomplished by eliminating overstory vegetation, trees and shrubs, in prearranged patterns that create optimum cover/forage ratios. Benefits to food resources can be realized by eliminating undesirable plants, removing dense, rank, and/or overmature growth to stimulate crown or root sprouting, and to increase the nutritive value.

Effects of Fire on Overstory Trees and Shrubs

Hot Desertscrub. Fuel accumulations in the Mojave, Sonoran, and Chihuahuan deserts are generally too sparse to carry fire except under unusual circumstances. Occasional wet winters and springs can produce an abundant crop of annuals which do provide fuels that permit burning (Pase and Granfelt 1977). Prescribed burning is not commonly used in the desert regions except at the interface of the Arizona Uplands of the Sonoran Desert and the Interior Chaparral or Semidesert Grasslands (Humphrey 1974). Kittams (1972), for example, reported effects of fire on Chihuahuan Desert vegetation, but in fact, studied fire on a range of communities—Chihuahuan Desertscrub, Semidesert Grasslands and a juniper-oak woodland type. Of the 14 plant species he studied, 3 were usually killed by fire while the rest responded by crown, stem, or root sprouting. Eleven were reported improved as deer forage following fire.

Semidesert Grasslands. The historical importance of wildfire in development of the Semidesert Grassland has been discussed by Humphrey (1962, 1963, 1974) and Hastings and Turner (1965). Several studies have evaluated effects of wild and prescribed burns and have been reviewed by Cable (1973). Most have focused on controlling "undesirable" plant species (mesquite, chollas, burroweed, and pricklypears) and determining effects of burning on "desirable" range grasses.

Although no studies were directly applicable to deer, some inferences can be made that could be beneficial to their habitat. Openings in extensive mesquite stands could be maintained, but probably not created, with periodic fires. Mesquite is quite resistant to fire because of its sprouting ability. Mature stands cannot be killed by fire because herbage production is inadequate and because the larger trees are hard to kill. However, fire can possibly be used to maintain a relative degree of control in young, invading stands or in stands where mature trees have been killed by another method. Periodic burning under these conditions would top-kill small trees and keep sprouts close to the ground (Cable 1973). Other shrubs that are important deer foods, such as falsemesquite calliandra and velvetpod mimosa, also sprout following fire (White 1969, Cable 1973).

Interior Chaparral. Most fire work in Interior Chaparral has been directed at livestock forage or water yield improvement. Hence, elimination of chaparral vegetation has been a primary goal. Severe wildfires are common in Interior Chaparral because of heavy fuel accumulations; hot, dry conditions in early summer; and rough, generally inaccessible, terrain. Most shrubs in this type are, however, well adapted to periodic fire and sprout vigorously from root crowns (Cable 1957, Pase and Pond 1964, Pase and Lindenmuth 1971, Pase and Granfelt 1977, Carmichael et al. 1978). The type is so well adapted to fire that secondary succession is not a series of vegetational stages but a gradual ascendance of the long-lived species present in the prefire stand (Carmichael et al. 1978). Pond and Cable (1960), for example, noted that shrub live oak was reduced only after five consecutive annual burns. Skunkbush sumac was equally hard to kill, but Wright silktassel and hollyleaf buckthorn could be reduced by burning two times in 3 years. Pond and Cable (1960) concluded that repeated burning was an impractical shrub control practice. As a result other control methods, primarily herbicides, have been used in conjunction with fire to prevent reestablishment of shrub dominants (Pase and Pond 1964, Hibbert et al. 1974). These methods eliminate desirable deer foods, such as birchleaf cercocarpus, as well as undesirable ones (Pase 1967).

Longevity of fire-only treatments apparently varies with productivity of the site. Cable (1957) reported that shrub growth in a burned study area was the most rapid the first 2 years following fire, and by 5 years shrub density was equal to that on the unburned control. Pase and Pond (1964) noted that chaparral crown canopy was still increasing 6 seasons after a wildfire. It has further been reported that the area may be a barrier to wildfire for 15-20 years (Pase and Granfelt 1977).

The herbaceous understory responds to temporary shrub control. Pond and Cable (1962) found that a variety of herbs become established immediately after a fire but diminish rapidly as shrubs gain prefire stature. Pase and Lindenmuth (1971) also reported a significant increase in forbs but noted that grass abundance is only incidental, both before and after burning.

Prescribed burning does have the potential to be an effective deer habitat improvement tool in the Interior Chaparral. Objectives would not involve conversion to grassland but creation of a mosaic of different aged chaparral stands juxtaposed to provide optimum forage and cover ratios. Much of Arizona's Interior Chaparral vegetation is too low to provide optimum cover (Swank 1958) as it does in California Chaparral (Leopold et al. 1951, Taber and Dasmann 1958). Hence, deer tend to use intermingled habitats, such as riparian communities, and favorable topographic situations for cover. The primary purpose of prescribed fire in chaparral would be to create a relatively more palatable and nutritious food source by producing young shrub shoots and associated forb growth. Particular care must be given to protecting cover types wherever they occur.

The use of controlled fire in the chaparral type is difficult because fire suppression activities over the last few decades have permitted fuel loads to accumulate. Unless carefully applied, even controlled fires can result in enough heat to cause undesirable changes in soil properties. Pase and Lindenmuth (1971) found that narrow contour strips of chaparral could be burned in the fall without special risk. Their prescription, however, involved a desiccation treatment with herbicides (2,4-D and 2,4,5-T) 6 weeks prior to ignition in order to burn during this relatively moist season. Environmental concerns over the use of herbicides and the additional costs involved may preclude

use of this treatment.

Madrean Evergreen Woodland. As with Semidesert Grasslands, considerable debate exists concerning the role fire played in development of the Madrean Evergreen Woodland (Humphrey 1962, Hastings and Turner 1965). Most debate concerns the lower encinal, particularly adjacent to Semidesert Grasslands. Fire is generally acknowledged to have been present and influential in the upper encinal or Mexican pine-oak woodland (Leopold 1924, Marshall 1963). The logic developed concerning effects of fire in this community has been based on subjective analysis. The only quantitative study has been an analysis of sprouting ability of tree species after a wildfire. Johnson et al. (1962) found that one-seed juniper, normally considered a nonsprouter, had a fire mortality of 76% and that 10% of the remaining trees sprouted. Alligator juniper, a sprouting species, had an average fire mortality of 28%, but 42% of the remaining trees sprouted. The two oaks studied, Emory oak and Arizona white oak, had mortality in both control and burned areas, but mortality was slightly less in the control. Sprouting was evident in both areas, but both oaks had a much higher sprouting percentage in the burned area.

A study by Barsch (1977) indicated that Coues white-tailed deer responded favorably to a wildfire in upper encinal, particularly to development of browse-dominated feeding areas, which indicates that prescribed burning has potential as a deer habitat management tool in this complex community.

Great Basin Conifer Woodland. Fire is generally accepted as a major force in development of Great Basin Conifer Woodland or pinyon-juniper. Cessation of fires, through suppression activities and removal of fire-carrying herbaceous growth by livestock, appears to have been a major factor contributing to the spread and increased densities of these stands (Leopold 1924, Humphrey 1962, Wright et al. 1979). The type is not so well adapted to fire as chaparral, however, because successional stages are distinct (Arnold et al. 1964, Barney and Frishknecht 1974). Several researchers have analyzed effects of wildfire while others have worked with prescribed fires (see review by Wright et al. 1979). Guidelines for using prescribed fire have also been developed for use in pinyon-juniper woodlands (Bruner and Klebenow 1979).

Like most fire studies in semidesert Grasslands and Interior Chaparral, most studies in Great Basin Conifer Woodland have dealt with increasing herbaceous forage production for livestock. Little emphasis has been given wildlife habitat.

Limited data indicate manipulation of pinyon-juniper with fire must be carefully planned in order to enhance cervid habitat. Objectives depend on kinds of associated plants, season of use, and whether desired results are to be short- or long-term. Prescribed fire probably could be successfully employed in pinyon-juniper areas adjacent to chaparral stands where many of the understory shrubs would be the same sprouters that occur in Interior Chaparral. Fire could be used here to remove trees and decadent shrubs, to stimulate sprout production of shrubs, and increase herbaceous growth. In northern areas such as the Coconino and Kaibab plateaus in Arizona, most understory shrubs associated with conifer woodlands are easily killed by fire. Two, cliffrose and big sagebrush, are important deer winter foods (McCulloch 1969). Analysis of burned areas on the Hualapai Indian Reservation revealed significant increases in herbage production, however, which caused burns to be heavily used by mule deer in fall and winter. Snow cover was not a problem because it melted within a few days after storms, so shrubs were not critical forage items (McCulloch 1969). On the

Kaibab Plateau, 40 miles north, where snow accumulations were significant, McCulloch (1979) recommended that pinyon-juniper trees be removed by methods such as bulldozing that would not kill palatable shrubs, especially cliffrose. Short-term results possibly could be obtained by treating foraging areas on spring-fall-winter pinyon-juniper ranges for both herb and shrub production. Part of the area could be bulldozed to enhance shrub production and the remainder burned to promote herbaceous growth. Reseeding burned areas is often a necessary practice in pinyon-juniper stands that have been excessively dense.

Great Basin Conifer Woodlands could be managed for mule deer habitat on a long-term basis by partial burning of stands on an extended rotation. Successional patterns following fire in pinyon-juniper stands (Arnold et al. 1964, Barney and Frischknecht 1974) indicate that a pattern of herb-, shrub-, or tree-dominated communities (the first two would be foraging areas, the latter a cover area) could be maintained in a variety of options, depending on cover and forage requirements for the area. As a simplified example, assume an optimum forage/cover ratio of 60/40, then:

- 1) Year 1—burn 30% of the area in patches of an optimum size for mule deer (30% herbaceous forage, 70% cover).
- 2) Year 25—burn another 30% of the area. At this time the portions burned in year 1 will be succeeding to shrubs (30% herbaceous forage, 30% shrub forage, 40% cover).
- 3) Year 50—burn another 30%. The portion burned in year 1 should be close to juniper dominance; the year 25 burn will be succeeding to shrubs (30% herbaceous forage, 30% shrub forage, 30% marginal cover—some forage, 10% excellent juniper cover).
- 4) Year 75—burn the remaining 10%. The portion burned in year 1 should be dominated by junipers; that in year 25 will be changing from shrubs to juniper; and that in year 50 should be dominated by shrubs (10% herbaceous forage, 30% shrub forage, 30% marginal cover—some forage, 30% cover).
- 5) Year 100—back to year 1.

This approach is obviously idealistic. Successional rates and stages would vary depending on site, composition of understory prior to the burn, intensity of fire, postburn treatments (e.g., reseeding), weather conditions, and influence of other land uses (e.g., livestock grazing). Also, this approach implies broadcast burning which, in pinyon-juniper, is very difficult and requires specific conditions (Arnold et al. 1964). The impacts on other land users of burning 30% of an area in 1 year also have not been considered. However, the basic idea of creating a diverse mosaic of different successional stages in otherwise homogeneous pinyon-juniper type would improve mule deer habitat. Although difficulties involved in implementation of a prescribed burning program are impressive, it is still one of the least expensive (Aro 1971) and is an environmentally acceptable means of achieving this goal.

Montane Conifer Forest. More attention has been given fire in the Montane Conifer Forest, particularly pure ponderosa pine types, than in any other biotic community in the Southwest. Effects of historical and recent wildfire and uses of prescribed fire in ponderosa pine have been reviewed by Cooper (1960), Biswell et al. (1973), Biswell (1973), Weaver (1974), and Wright (1978). Prescribed burning programs have been actively pursued in southwestern ponderosa pine types since 1948 (Kallander 1969). Most existing management and past research efforts have been directed at burns designed to remove excessive fuel accumulations (Fig. 9) or to promote natural regenera-

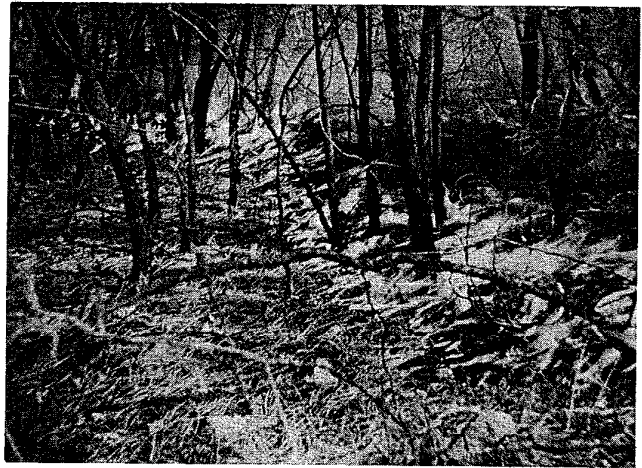


Fig. 9. A prescribed burn in northern Arizona designed to reduce fuel accumulations and induce limited mortality of crowded ponderosa pine.

tion of ponderosa pine by removing duff and reducing competition from other plants, e.g. Gambel oak (Kallander 1969, Ffolliott et al. 1977a, Martin et al. 1979).

Benefits to deer and elk resulting from fire in ponderosa pine have been related to increases in understory vegetation resulting from reductions in number of live, competing trees, amount of litter, and depth of duff. Litter and duff can accumulate to significant proportions on undisturbed forest floor, and these accumulations can result in a decrease in herbaceous understory. Clary et al. (1968) found that herbage decreases from 300 to less than 10 pounds per acre as the total forest floor accumulations increase from essentially zero to 2.5 inches. Ffolliott et al. (1977a) noted similar results. Reductions in litter may also decrease amounts of growth inhibitors reported in ponderosa pine (Jameson 1968).

Fire should be tested as a habitat improvement tool by following logging operations with prescription burns designed to remove excessive litter and promote nutrient cycling using established guidelines (Davis et al. 1968, Chrosciewicz 1978, Martin et al. 1979). Objectives would only be directed to those sites designated as foraging areas. The entire foraging area would not necessarily have to be treated; the treatment should avoid sites with soils particularly susceptible to erosion or those that develop water-repellent layers when burned. Although long-term influences of such programs undertaken on a large scale are unknown, the frequency of historical natural fire indicates that it can be done safely with few detrimental effects. Biswell et al. (1973) and others determined that ponderosa pine forests burned every 6 to 7 years. Dieterich (1980), using all fire-scarred material for a 336-year period, found an average interval of one fire every 4.9 years in northern Arizona. During one 126-year time span, fires burned at 2.5-year intervals, and over a 15-year period at intervals of one fire every 1.25 years. These fires spread in grass and needle material, and were of variable, but generally low, intensity. Cooper (1961) hypothesized that these fires were responsible for maintaining a mosaic pattern of even-aged groups that averaged about 0.2 acres. Assuming considerable variation in ages of groups, such a pattern could provide diversity. From the standpoint of cervid habitat, broadcast burning to simulate these types of fires appears better than a pile-and-burn policy at least in ponderosa pine.

Mixed Conifer, Subalpine Conifer, and Aspen. Information on fire in the mixed conifer type is scarce. It was previously

thought that fire had little influence in its development, but recent information from Thomas Creek in the White Mountains of Arizona indicates that this area was completely burned over at 25-35 year intervals. There were also additional, smaller fires within the period.⁷

Fires occurred naturally in the Subalpine Conifer Forest, but were less frequent than in ponderosa pine because of the mesic nature of the habitat. During extraordinarily dry years, however, fires did occur in spruce-fir. Less frequent burning led to large fuel accumulations, so fires destroyed more trees when they did burn. Kallander (1969) reported, on the Fort Apache Reservation in Arizona, that 80,000 acres burned during the drought of 1903-1904 and that 20,000 acres were spruce-fir on north-facing slopes at lower elevations within the zone. He mentioned that previous, frequent burning on south slopes had reduced fuel loads to the point where mixed conifer trees survived through the 1904 fire. Conditions were moist enough in the upper spruce-fir zone that the fire failed to penetrate this part of the forest even during drought.

The existence of large, overmature, quaking aspen stands in Mixed Conifer and Subalpine Conifer forests also indicates that fires were present, however sporadically. Fire is an important factor in maintaining this seral stage in both communities. Prescribed burning may be a necessary tool for rehabilitation of some aspen stands, depending on seral age. Although relatively stable stands have been identified in Colorado (Morgan 1969), Wyoming (Reed 1971), and South Dakota (Severson and Thilenius 1976), most stands in the Southwest are seral stages that will eventually be replaced by conifers. The rate of conifer invasion depends on site factors and availability of a coniferous seed source (Jones 1974).

Multiple resource values are realized from aspen stands. Three characteristics that make them important components of wildlife habitat are (1) aspen is a palatable and nutritious browse, (2) the herbaceous understory produced in aspen stands is more productive and diverse than in adjacent coniferous stands (Reynolds 1969a, Kranz and Linder 1973), and (3) aspen is the only deciduous tree at higher elevations in the Southwest and can contribute significantly to the enhancement of diversity and creation of additional edge. Other values of aspen stands include the understory which is favored by livestock (Cook and Harris 1968, Reynolds 1969b, Kranz and Linder 1973). Aspen also has potential as an energy source in livestock feed formulations (Kamstra 1977). This forest type provides excellent watershed protection (Betters 1976, Hronek 1976) as well as being esthetically desirable, both as a foreground and background vegetation (Hronek 1976). Aspen stands can be managed as wildfire fuel breaks because of low ignition rates, low burning index, and lack of ability to carry a crown fire (Fechner and Barrows 1976). Aspen wood is extensively utilized in the north-central United States, primarily as pulpwood and chipboard. Southwestern markets for aspen wood fiber are, however, very limited.

Aspen stands can be manipulated to provide both food and cover for deer and elk. Most management considerations would be based on amount of root sprouting, which appears to depend on the age of the aspen stand, measured not in years but seral stage (i.e., the degree of conifer invasion and the general health and vigor of the aspen trees).

Aspen stands have an extensive lateral root system, and the primary method of reproduction is by sprouting from these

roots. Production of viable seeds is erratic, but occasionally excellent crops are realized. The tiny seedlings, however, are very susceptible to both drying and damping off and are established only under exceptionally favorable circumstances (Patton and Jones 1977). When a stand is clearcut, burned, or otherwise disturbed, sucker density may be in the tens of thousands per acre (Jones 1974). Maximum densities are realized the first year, with a gradual decline thereafter (Jones and Trujillo 1975). In Arizona, clearcut aspen stands produced about 14,000 root suckers per acre the first growing season, and after four growing seasons about 10,700 per acre were still alive (Jones 1975). Patton and Avant (1970), however, found that the number of sprouts averaged 11,800 per acre the first year, increased to 14,500 the third year, then decreased to 11,850 the fourth year in New Mexico. Bartos and Mueggler (1979) noted similar results in a Wyoming study.

The reasons for such differences may be the condition or age of the aspen stand before treatment and the type of treatment imposed. Jones (1975) worked in mature, but vigorous and apparently healthy, aspen stands; conifer invasion was present, but aspen dominated. Patton and Avant (1970) studied the effects of wildfire in a more advanced seral stand that had Engelmann spruce, Douglas-fir, and aspen in the overstory. Deep litter helped maintain a hot ground fire that consumed all the understory and completely defoliated the overstory. The Wyoming study used prescribed fire in ten aspen clones, which although dominated by aspen, were described as decadent (Bartos and Mueggler 1979). Clearcutting, as a single treatment, in older deteriorating aspen stands such as these probably would have produced relatively few suckers (Patton and Jones 1977).

These studies indicate that different treatments are necessary to induce favorable responses in aspen stands of different seral ages. Treatment can be varied within stands to realize different objectives. Three examples follow and include: (1) young, vigorous aspen stands with little or no conifer invasion, (2) healthy aspen stands with trees approaching or at maturity but with significant conifer invasion, and (3) overmature or senescent stands in which aspen has yielded dominance to invading conifers.

Young, vigorous stands with little or no conifer invasion (fig. 10) can be managed for two objectives. One would be to establish clearcut openings to induce suckering. In stands of this



Fig. 10. A relatively young aspen stand with little conifer invasion. Root-suckering can be induced by a single clearcutting treatment.

⁷Personal communication, John H. Dieterich, Project Leader, Fuels Management, Rocky Mountain Forest and Range Experiment Station, Tempe, Ariz.

type, this is the only treatment necessary to induce sprouting. This creates a mosaic of even-aged groups and an uneven-aged stand. In addition to diversity, it provides abundant browse for 5-8 years, at which time the leafy crowns grow beyond the reach of deer and elk (Patton and Avant 1970, Patton and Jones 1977). The optimum size of clearcut openings depends on several factors, including the management objectives, size of the original stand, and number of ungulates using the area. Clearcuts that are too small may result in concentration of browsing animals to the point where the aspen regeneration is eliminated (Jones 1967). Smith et al. (1972) found such elimination under severe browsing pressure. Mueggler and Bartos (1977) recommended scattered, 5- to 12-acre, openings for greater habitat diversity and improved forage conditions in extensive areas of aspen in Utah.

The time the clearcut is made may also be important, although evidence is conflicting. Aspen sprouts depend on carbohydrate reserves until they reach the surface and can carry on photosynthesis. Reduction in suckering was once thought to be linked to low levels of carbohydrate reserves in the roots during active leaf development in spring and early summer. Tew (1970), however, determined that carbohydrate reserves vary with season, but low carbohydrate levels do not influence the number of suckers produced. Until this problem is resolved, managers should adhere to Perala's (1972) conclusion that it is probably desirable to harvest aspen during the dormant season to insure maximum sprout response.

The other management option is to manipulate aspen stands to promote herbaceous understory production, particularly forbs (Mueggler and Bartos 1977) rather than suckers. This may be accomplished by thinning rather than clearcutting. Although understory production cannot be related to overstory parameters in aspen stands because of complex root systems (Severson and Kranz 1976), thinning results in significant increases in herbaceous understory (Reynolds 1969a). Some aspen sprouts are produced but fewer than if the stand is clearcut (Shirley 1931). The number of sprouts depends on thinning intensity. Information is lacking concerning optimum thinning intensities to promote herbaceous understory growth while controlling sucker response. A demonstration study is feasible to determine this response by removing 50% or less of the existing basal area immediately after trees become fully foliated.

Healthy aspen stands with trees approaching or at maturity and low to moderate coniferous invasion can be revitalized by clearcutting. Some scarification of the ground surface may be desirable to expose mineral soil to solar radiation. Sufficient scarification can usually be achieved in harvesting if skidding can be routed through the stand (Perala 1972). Understory production in stands with conifer invasion can be increased by removing the coniferous growth (Severson and Kranz 1976). Thinning the aspen may not be necessary because by this age natural thinning should have reduced the stocking rate to an acceptable level. An equally viable treatment, here and in the first example, would be to remove all aspen to promote conifer growth, especially in extensive stands where security and thermal cover may be limited.

Stimulation of suckering in overmature aspen stands, particularly those in which the aspen has lost dominance to invading conifers (fig. 11), is more difficult. Experience has shown that clearcutting, as a single treatment, does not result in an adequate sucker response. Wildfires, however, have been shown to promote sprouting even under these circumstances (Patton and Avant 1970); therefore, clearcutting followed by a controlled



Fig. 11. An older aspen stand containing conifers which are about to achieve dominance. Clearcutting as a single treatment may not result in significant sucker response. A relatively intense fire following clearcutting may stimulate additional suckering.

broadcast burn may yield acceptable results.

Increased root temperature caused by exposure of soil to sunlight is the cardinal factor in stimulation of suckering (Maini and Horton 1966). Apparently, in healthy aspen stands the increased soil temperature resulting from clearcutting is adequate stimulation. Light burning may be used in some instances. Shirley (1931, 1932) stimulated sprout response in Minnesota with an autumn fire at an intensity that just removed the top layer of new-fallen leaves. He attributed success to stimulation caused by increased heat absorption of the blackened surface.

As stands pass maturity, however, higher intensity fires may be necessary to stimulate adequate suckering. Litter and duff materials under older stands are probably deeper. Although aspen leaves decompose relatively fast, the inclusion of the more acidic coniferous materials tends to retard decomposition rates. Litter is an excellent insulator and prevents warming of soil, even if the overstory is removed. Low-intensity fires, whose primary effect is to create a dark surface, may not be sufficient. A moderate- to high-intensity burn creates higher soil temperatures in three ways: directly by heating mineral soil (temperatures up to 233° F have been recorded 8.5 inches below the surface when burning piled slash) and indirectly by removing all insulating effects of litter (Wells et al. 1979) and by creating a heat-absorbing blackened surface.

There is apparently no danger of too much heat, at least as far as aspen regeneration is concerned. Horton and Hopkins (1965) determined that it is impossible to prevent root suckering by intense burning. Temperatures lethal to root tissue are obtained only close to the surface in very dry soil under sustained surface heating. Schier and Campbell (1978) also found where fire was very hot, roots near the surface were killed; but more suckers arose from deep roots in areas where the burn intensity was high than in areas where it was low. There may be differences in rates of sprouting because of burn intensity. Bartos and Mueggler (1979) found that, with high-intensity burning in decadent stands, suckers decrease the first postburn year to about one-third preburn numbers, but by the third year they double preburn numbers. Under moderate-intensity burning, however, the number of suckers triples the first year, increases to seven times the preburn numbers the second year, but by the third year decreases by natural pruning to the first year postburn level.

Thus, after 3 years both burn intensities result in the same number of aspen suckers.

Specific information on effects of clearcutting and burning of overmature or senescent aspen stands which have ceded dominance to conifers is admittedly scarce. However, peripheral information available indicates that clearcutting and prescribed fire may result in significant sucker response. Management of such stands is critical, however difficult. Only 7% of the 479,000 acres of aspen in Arizona and New Mexico is in the seedling and sapling stage. Most is obviously mature or overmature (Fig. 12). If the present aspen acreage is to be maintained or increased, manipulation of older stands is imperative (Patton and Jones 1977). Research on techniques, particularly regarding sucker response to clearcutting and season and intensity of prescribed fire, should be given priority.



Fig. 12. Many southwestern aspen stands are overmature and the sites will soon be dominated by conifers. If present aspen acreage is to be maintained or increased, manipulation of such stands is imperative.

Mountain Meadow Grasslands. Fire frequencies in meadows are somewhat similar to those in surrounding forested areas. Although wet meadows dominated by sedges may burn less frequently than surrounding forest, fire is a natural component of these meadows (DeBenedetti and Parsons 1979). Periodic burning appears necessary to keep conifers from slowly invading. The drier, bunch grass dominated meadows common to the Montane Conifer Forest probably burned as regularly as the ponderosa pine forests. Many are now being invaded, although in some cases quite slowly, by pine. This invasion has been attributed to overgrazing by livestock which removed the competing grasses, suppression of naturally occurring fires, and changes in subsurface moisture regime induced by overgrazing, gullying, road construction, and increased conifer growth on surrounding areas. Prescribed burning has been used to restore mountain meadows (Wagtendonk 1980). Little quantitative information exists regarding specific procedures on effects on other components of the ecosystem, however. One study conducted in the Black Hills, South Dakota, revealed that a late fall prescribed burn of a relatively xeric meadow did not change total forage production, but did alter proportions of major components. Production of warm season grasses was significantly reduced but the shrub component, all important white-tailed deer foods, was increased. All ponderosa pine seedlings less than 6 feet high were "browned," but most from 2 to 6 feet recovered, all less than 2 feet high were killed.⁸

⁸Personal communication, F. Robert Gartner, West River Research and Extension Center, South Dakota State University, Rapid City.

Gartner and Thompson (1973) reported ponderosa pine seedling mortalities of 93% and 85% in grasslands adjacent to pine stands at lower elevations in the Black Hills; mortality was not broken out by height classes.

Great Basin Desertscrub. Humphrey (1974) suggested that the Great Basin Desertscrub is more subject to natural fires than any other desert community in the Southwest. He thought this was due to the prevalence of big sagebrush, often with an understory of associated grasses that provided additional fuel. Most shrubs which are considered important mule deer foods, notably big sagebrush and antelope bitterbrush, were, however, damaged or eliminated by fire. Although some work has been done regarding use of prescribed burning to improve livestock range in the northern areas of the Great Basin Desertscrub (Pechanec and Stewart 1944, Blaisdell 1953), little information is available for the Southwest. A limited amount of sagebrush burning has been done in Arizona on the Hualapai Indian Reservation. Cover is a limiting factor in this desert. Significant amounts may be provided in the ecotone with Great Basin Conifer Woodlands, by riparian habitats, and by local topography. Limited use may be made of fire to create a mosaic of feeding areas near such types but, based on our relative lack of information, the areas should be small and closely monitored.

Plains Grassland. Fire has had a significant influence in development of Plains Grassland communities. Reviews on the role of fire in grassland environments have been prepared by Jackson (1965), Daubenmire (1968), and Vogl (1974). Topography, climate, and vegetation all combined to favor the occurrence of prairie or plains fires. Topography is relatively flat or undulating. Climatic factors include dry periods, high winds, and lightning storms. Finally, dry grass is a highly flammable fuel. The frequency of natural fires in segments of the Great Plains in the Southwest is open to some speculation. Records, gathered from the Texas Panhandle and eastern New Mexico, indicate that fires were common and widespread in the mid- to late 1800's (Jackson 1965). Fire plays a role in maintaining grasslands wherever they border forested or woodland habitats (Humphrey 1962, Gartner and Thompson 1973). Fire is not the primary force responsible for suppressing woody growth on the upland areas of the Plains, however; this is a function of erratic precipitation, punctuated by periodic severe drought (Weaver and Albertson 1956, Humphrey 1962). The woody vegetation that existed in riparian areas probably did not burn as frequently as the grasslands for two general reasons. One was that not all riparian types were susceptible to fires. These were found in the bottoms of deep, steep-sided canyons and arroyos which stopped, or were skipped over by, the rapidly moving grass fires. The more favorable moisture conditions kept the understory greener for longer periods of time. This, coupled with the green foliage of the overstory, made these areas less vulnerable to fire when the grasslands were most apt to burn—late summer and early fall. Another consideration is, even if riparian habitats did burn, the shrubs and trees, like the grasses, are well adapted to fire, either by being fire resistant, such as large, mature cottonwoods, or by sprouting from crowns or roots (Severson 1981).

Most of the research on fire in the Plains has been oriented to response of grasses. The scant amount available on shrubs has been more concerned with their control rather than perpetuation or stimulation. Again because of a lack of information, fires in Plains riparian habitats should be used with caution. Cover is a limiting factor for mule deer in grassland ecosystems, so if destroyed, even temporarily, its effects may be felt. However, fire may be a useful tool to stimulate senescent cover types, to

enlarge clones of some species, or to manipulate nutritive values of forage plants.

Effect of Fire on Plant Nutrients

The influences of fire on chemical properties and nutrient cycling in soils, including studies on nutrient availability from plant seedling trials and pot experiments, has been reviewed by Wells et al. (1979). This discussion will be confined to effects of fire on nutrient contents of naturally growing native plants, because they directly affect health and productivity of cervid populations and because of the possibility that deficient amounts of certain nutritional elements may be limiting that productivity.

Most of the information available indicates that the nutrient content of plants growing on burned areas is higher than that of plants growing on preburn or control areas (DeWitt and Derby 1955, Lay 1957, Leege 1969, Dills 1970, Lawrence and Biswell 1972, Pearson et al. 1972, Stark and Steele 1977) provided that reasonably high-intensity fires are involved. Most studies

have indicated that nutrient contents revert to preburn or control levels in 2 years or less or even before the beginning of the next growing season. Only DeWitt and Derby (1955) and Stark and Steele (1977) reported effects lasting beyond 2 years. DeWitt and Derby reported significantly higher protein contents in four plant species after 2 years following a high-intensity burn. Stark and Steele noted higher concentrations of phosphate, iron, manganese, and sodium in an understory composite 3 years after a hot burn (i.e., soil temperature greater than 300° C and all litter removed). Short duration, light burn intensities (<150° F) did not result in an increase of nutrients in foliage (Stark 1980).

Several studies have documented nutrient changes in vegetation following burning in the Southwest. Two have been in ponderosa pine types in the vicinity of San Francisco Peak in northern Arizona. One documented effects of wildfire (Pearson et al. 1972); the other is a continuing study on effects of prescribed burning being conducted by several individuals at Northern Arizona University in Flagstaff. Because of the southwestern locales, these data are presented in Tables 6 and 7.

Table 6. Nutrient content (%) of native forages on burned and unburned areas on the Wild Bill Range, Ariz. Burn resulted from wildfire on May 9, 1967 (Pearson et al. 1972).

| Date | In vitro digestibility | | Crude protein | | Phosphorus | |
|------------|------------------------|----------|---------------|----------|------------|----------|
| | Burned | Unburned | Burned | Unburned | Burned | Unburned |
| June 1967 | 63 | 63 | 16 | 12 | 0.43 | 0.23 |
| Aug. 1967 | 66 | 57 | 19 | 12 | .39 | .32 |
| Oct. 1967 | 69 | 60 | 10 | 8 | .27 | .23 |
| July 1968 | 65 | 62 | 9 | 10 | .25 | .22 |
| Aug. 1968 | 61 | 54 | 10 | 10 | .22 | .21 |
| Sept. 1968 | 51 | 50 | 10 | 10 | .27 | .22 |
| July 1969 | 57 | 56 | — | — | — | — |

Table 7. Nutrient content of selected species in early and late summer areas, for 3 years following a November, 1976, low-intensity prescribed burn at Fort Valley, Ariz.

| Date ^d | Species | Year of sampling | | | | | |
|--|-------------------|-------------------|----------|-------------------|----------|-------------------|----------|
| | | 1977 ¹ | | 1978 ² | | 1979 ³ | |
| | | burned | unburned | burned | unburned | burned | unburned |
| Percent crude protein⁵ | | | | | | | |
| E | Arizona fescue | 11.0 | 8.4 | 12.6 | 10.9 | 7.0 | 7.2 |
| E | Mountain muhly | 7.6 | 6.4 | 7.8 | 7.6 | — | — |
| E | Fendler ceanothus | — | — | — | — | — | — |
| L | Arizona fescue | 9.1 | 7.6 | 8.3 | 7.8 | — | — |
| L | Mountain muhly | 5.5 | 5.4 | 6.0 | 5.9 | 5.7 | 4.9 |
| L | Fendler ceanothus | 14.7 | 14.8 | — | — | 13.5 | 13.9 |
| Percent phosphorus | | | | | | | |
| E | Arizona fescue | 0.32 | 0.26 | 0.25 | 0.27 | 0.24 | 0.26 |
| E | Mountain muhly | .34 | .26 | .22 | .27 | — | — |
| E | Fendler ceanothus | — | — | — | — | — | — |
| L | Arizona fescue | .28 | .24 | .24 | .25 | — | — |
| L | Mountain muhly | .22 | .18 | .19 | .23 | .18 | .18 |
| L | Fendler ceanothus | .20 | .18 | — | — | .14 | .14 |
| Percent calcium | | | | | | | |
| E | Arizona fescue | .17 | .14 | .29 | .32 | .24 | .24 |
| E | Mountain muhly | .19 | .14 | .26 | .30 | — | — |
| E | Fendler ceanothus | — | — | — | — | — | — |
| L | Arizona fescue | .12 | .12 | .36 | .36 | — | — |
| L | Mountain muhly | .12 | .11 | .23 | .29 | .28 | .31 |
| L | Fendler ceanothus | .36 | .38 | — | — | 1.04 | 1.08 |

¹1977 data from Harris (1978).

²1978 data from W.W. Covington, School of Forestry, Northern Arizona University, Flagstaff (unpublished tables).

³1979 data from Jeffrey L. Kogutt, School of Forestry, Northern Arizona University, Flagstaff (unpublished tables).

⁴Date: E = early summer, L = late summer.

⁵Percent crude protein = 6.25 × percent nitrogen.

Swank (1958) also presented a general discussion of protein, phosphorus, and fire in Interior Chaparral. He reported increased protein contents from one 9-month-old burn at Crown King, Arizona, but not from another 9-month-old burn at Three Bar. He also indicated a higher average protein content in forage plants collected in a 5-year-old burn, but not in those from 3- or 8-year-old burns. He presented tabular information on phosphorus contents of forage plants that had been collected from areas burned 3, 5, and 8 years previously. This information indicated that plants from burned areas are lower in phosphorus than those from unburned sites.

Discussion thus far has been limited to percentage concentration of nutrients from selected forages. Rundel and Parsons (1980) have gone further. Working with two shrub species in California Chaparral and considering nutrient distribution as related to stand structure, biomass, and canopy area distribution from plants growing on several burned sites of varying ages, they concluded that stands at 16 years of age represent the peak of productivity. Their rationale was based on allocation of nitrogen and phosphorus to photosynthetic tissue, which declined sharply in stands after 16 years. They suggested that the limited amounts of available nutrients become tied up in plant biomass and that fire provides a natural means of recycling nutrients that have been locked up by the plants. Zinke (1977) had previously concluded that mineral cycling in fire-type ecosystems is apparently dominated by periodic ashing of the vegetation and organic material on the soil surface. Rundel and Parsons (1980) further noted that natural fires in this vegetation type occurred at 15- to 20-year intervals, reasonably consistent with their data on temporal nutrient distributions.

This raises some rather interesting questions. What are the relationships among increases in shrub and tree growth resulting from intensive fire suppression and overgrazing, suspected and often verified senescence of shrub and tree species, and nutrient availability? How have these relationships affected the health and productivity of southwestern cervids if they have had any influence at all? Nutrient deficiencies, especially phosphorus and nitrogen (protein), previously have been indicated as a potential problem regarding productivity of deer and elk in the Southwest (see previous section on Nutritional Relationships) as elsewhere (Klein 1970). Little work has directly addressed the problem, however. As Wiener (1975) indicated, few studies have considered the dynamics of a vertebrate population from any nutritional standpoint, much less that resulting from fire-induced cycling. As Bendell (1974) stated, the relationships among burning, release of nutrients, and what may be taken up by plants, which, in turn, will be available to animals is very complex.

There appear to be two basic kinds of nutrient cycles that may operate in fire ecosystems. A short-term cycle that makes nutrients from ash immediately available to plants, was termed luxury consumption by Rundel and Parsons (1980). These immediate but short-term increases in plant nutrients are available to herbivores on a fairly consistent basis in areas that burn frequently, such as ponderosa pine. Even though the entire forest floor may not have burned, there are nutritious patches scattered throughout. A longer-term cycle may operate in which most of the available nutrients are tied up in old, woody tissue of senescent shrubs and older trees. These are unavailable to most herbivores until released by fires at 15- to 75-year intervals, depending on the ecosystem. Fires could be important for long-term cycling of nutrients in Semidesert Grasslands, Interior Chaparral, Madrean Evergreen Woodland, Great Basin Conifer

Woodland, and Montane Conifer Forest.

The role of fire in maintaining edge, interspersion, and diversity within an ecosystem is generally understood (Bendell 1974) even if not universally accepted as necessary. The importance of fire in the creation and maintenance of "nutritional diversity" is more subtle and perhaps more complex, but peripheral evidence suggests that it could be a tool worth further investigation.

Cervid Response to Fire

Fire can have two general effects on animals; direct effects are those associated with immediate mortality or displacement, and indirect effects are long-term influences related to habitat alteration. Although some evidence on mortality and displacement has been reported for vertebrates, most investigators generally consider such effects to be relatively minor, especially for the larger forms (Komarek 1969, Bendell 1974, Lyon et al. 1978). Indirect effects are those associated with relatively longer term influences of habitat changes which have been discussed in previous sections. The purpose of this part will be to examine how and why cervids respond to such changes.

Burning has a profound influence on the distribution of ungulates. This response will often start immediately after a fire, when animals gather on the blackened surface. Komarek (1969) noted cases in Africa where native animals were found "nibbling" on recent burned areas, presumably consuming ash for nutrients. The strongest attraction, however, is provided by burns in the "greening" stage. Although little quantitative documentation is available concerning cervid response to this initial green flush of vegetation, most of us have observed its effects and, like Komarek (1969), have probably attributed it to "fresh, nutritious herbage." Cattle are so attracted to burned areas that prescribed fire has been used to develop a grazing system based on rotation burning (Duvall and Whitaker 1964). Increased utilization of burned forage has been reported for white-tailed deer (Lay 1967, Barsch 1977) and cattle (Duvall and Whitaker 1964). Similarly, increased cattle weight gains have been observed for heifers grazing freshly burned pastures compared to those on unburned controls (McGinty and Smeins 1980). Ffolliott et al. (1977b) noted that mule deer, elk, and cattle use was higher in clearcuts where the slash had been piled and burned than in those in which the slash had been piled but not burned.

Immediate effects can probably be attributed to nutrient flush of the forage crop. The response of ungulates to burned areas generally continues past the time that nutrient levels are higher, however (Fig. 13). Kruse (1972) reported that elk and mule deer



Fig. 13. Elk feeding on a 6-year-old burn in northern Arizona. Arizona Game and Fish Department photo.

use increased significantly on a burned area for 2 years. Elk use leveled off the third year, but mule deer use continued to increase. In Wyoming, Davis (1977) found that elk and mule deer both use burned areas more than clearcuts of the same age. He studied burned areas and clearcuts in two different areas that had been treated 5 and 9 years previously. Colorado studies (Roppe and Hein 1978) indicated that in winter, mule deer and elk used a burned area more than an adjacent unburned lodgepole pine stand 8 years after a wildfire. In Arizona, use by mule deer in summer and fall declined the first year after a fire, then increased 2.5 times greater than the control for the rest of the 20-year evaluation period. Use by mule deer in the winter and spring also declined the first year, returned to control level for a few years, then increased to 10 times that of the control. Use by elk in the summer and fall was similar, but winter-spring use was higher than the control throughout the 20-year period, with the highest postfire use at 7 years after burning (Lowe et al. 1978). McCulloch (1969) noted higher mule deer use of burned areas in pinyon-juniper stands in northern Arizona. Ages of burned areas were variable but ranged from 4-12 years on prescribed burns to 15 years on a wildfire burn. Barsch (1977), working with pinyon stands in the Madrean Evergreen Woodlands of southern Arizona, determined that Coues deer deposited 7.2 times more fecal pellets in burned than in unburned stands and that the more abundant browse in burned areas was used 2.5 times more, 6.5 years after a wildfire.

Reasons given to explain the long-term attractiveness of burned areas include increased habitat diversity or edge (Lowe et al. 1978), increased production of preferred forage (McCulloch 1969, Kruse 1972, Barsch 1977, Davis 1977, Lowe et al. 1978), increased forage diversity (Davis 1977), and a combination of the above but including the cover provided by dead, standing trees (McCulloch 1969, Davis 1977, Roppe and Hein 1978).

Not all studies have documented immediate use of burned areas. A prescribed late-summer burn in aspen on elk winter range in Wyoming apparently made the area relatively unattractive to elk during the first winter. Elk use increased to preburn levels in the second and third year following the fire (Basile 1979). There may have been too little time for plant growth to respond to the fire before the first winter, specifically there may not have been enough time to provide available browse (aspen) above the snow.

A considerable amount of work has been done with prescribed burning and its effects on wildlife, particularly elk, in the northern Rocky Mountains. Primary purposes have been to create successional stages, notably shrubfields, in Douglas-fir forests; maintain shrub areas on winter range; reduce the crown height of shrubs, and, by sprouting, make them more available to browsing ungulates; increase total number of shrubs by creating seedbed conditions conducive to seedling establishment; and increase diversity of forage plants (Lyon 1966; Leege 1968, 1969, 1979; Lyon 1971; Leege and Hickey 1971). Martinka (1976) noted that wintering elk populations expand in response to fire and that these population increases are correlated with improved forage conditions and cover provided by young conifer stands. He indicated that extensive shrubfields or a continuous forest cover do not support high densities of wintering elk in Montana.

Recreation, Roads and Hunting

The impacts of human disturbances on big game have not been researched in the Southwest. However, extensive work has been done in other western states.

In Wyoming, Ward et al. (1976) have shown that elk are affected considerably more than mule deer and pronghorn antelope by traffic and human activity on highways and roads. Elk prefer to stay 0.25 mile away from interstate traffic and forest roads. Elk responses to disturbances vary from acknowledgment to flight, as indicated in Table 8. Ward et al. (1973) indicated elk adapt to disturbance activities in due time, provided a buffer zone of at least 0.5 mile is maintained from areas where people are concentrated. Perry and Overly (1976) found roads reduce big game use of adjacent habitat located from road edge to more than 0.5 mile away in Washington. They reported greater impacts result from main roads and open vegetation types as opposed to primitive roads and dense vegetation. A significant reduction in elk use of meadows results from all roads, especially those on west and south slopes. In Idaho, Hershey and Leege (1976) reported elk use of an area seems to decline in proportion to the density of roads in the area and the intensity, type, and use-season of the roads. Their data indicate that an established road open to traffic and crossing through an elk use area is disruptive to elk within 0.25 mile on either side of the road and force elk to disperse to areas further from the road. Several other researchers have documented significant reduction in elk use of habitats adjacent to roads crossing elk use areas (Burbridge and Neff 1976, Marcum 1976, Lyon 1979, Morgantini and Hudson 1979, Rost and Bailey 1979).

The relationship of deer to roads has received less attention. In Colorado, Rost and Bailey (1979) found road avoidance by

Table 8. Heart rate and activity response of cow and spike elk to disturbances. (Heart rate noted if rate increased two standard deviations) (Ward et al. 1976).

| Type of disturbance | Distance to elk (m) | Times occurred | Times heart reacted | Times elk moved away |
|---|---------------------|----------------|---------------------|----------------------|
| Sonic boom | — | 3 | 2 | 1 |
| Gunshots | 30-120 | 3 | 3 | 3 |
| | 140-450 | 3 | 2 | 1 |
| | >500 | 8 | 0 | 0 |
| Human on foot (with and without dog) | 20-100 | 10 | 10 | 9 |
| | 100-300 | 8 | 6 | 4 |
| | >300 | 2 | 1 | 1 |
| Trail bike | 15-50 | 4 | 4 | 1 |
| | 200 | 4 | 1 | 0 |
| | >400 | 3 | 1 | 0 |
| Auto (car horn) | 75-180 | 2 | 2 | 1 |
| | 275-365 | 4 | 2 | 0 |
| | >500 | 1 | 0 | 0 |
| Auto (stopped) | 35-100 | 5 | 2 | 2 |
| | 150-180 | 2 | 2 | 0 |
| | >450 | 3 | 2 | 0 |
| Auto (moving on trail) | 75-120 | 13 | 7 | 2 |
| | 150-180 | 9 | 7 | 2 |
| | >200 | 22 | 1 | 0 |
| Traffic (dirt road) | 365 | 10 | 0 | 0 |
| Traffic (highway) | 365 | 34 | 1 | 0 |
| Airplane | 30 | 1 | 0 | 0 |
| | 150 | 3 | 0 | 0 |
| | >200 | 41 | 6 | 0 |
| Distant noise (auto, train) | 800 | 7 | 2 | 0 |
| Natural noise (coyote, squirrel, thunder) | — | 5 | 0 | 0 |

deer is greater in shrublands than in pine forests and juniper woodlands. However, no significant difference in avoidance was found with respect to intensity of use on roads. In Wyoming, Ward et al. (1976) reported more deer crossings per mile where roads run adjacent to parks (206) and clearcuts (162) and fewer crossings within timbered areas (103), areas within 0.1 mile of streams (77), where the majority of underpasses and bridges are built, and within 0.25 mile of campgrounds (74).

The loss or gain of habitat directly replaced by roads and highway surfaces or rights-of-way is difficult to assess. Davis (1982) estimated a loss of 95,000 acres of wildlife habitat to roads on the national forests of Arizona and New Mexico. Wallmo et al. (1976) reported habitat losses from interstate, rural, and county highways to be 45, 12, and 7 acres of land per mile, respectively. Many roads are located within key big game habitat. Furthermore, they frequently traverse migration routes and/or winter range. Goodwin and Ward (1976) reported that sections of a Wyoming Interstate Highway present a barrier to seasonal migration and bisect critical mule deer winter range. In some instances, a gain in habitat results from revegetated highway rights-of-way. This gain, however, is negated by increased deer mortality from collisions with vehicles. Goodwin and Ward (1976) observed deer in the right-of-way most frequently in spring, during the growing season for crested wheatgrass. Mule deer made considerable use (19%) of this grass in the spring (Goodwin 1975).

Hunting and related activities are perhaps the greatest seasonal disturbance factors for big game. The increased accessibility to forested areas as a result of logging and other forest management practices has resulted in additional impacts on the sanctuary requirements of elk and deer. In western Alberta, Morgantini and Hudson (1979) reported that the hunting season causes a shift in habitat use and distribution pattern in elk and deer. They further noted extensive use and overgrazing of available grasslands by elk because of the shift. In Idaho, Irwin and Peek (1979) found hunters displace elk from preferred areas to areas of similar but more extensive habitat. Traditional use of the same areas each year appears to be induced by rutting, foraging activities, and partly by hunting.

The impacts of roads and hunting activities can be mitigated by road closures. Irwin and Peek (1979) stated road closures allowed elk to remain longer within preferred areas. In Arizona however, Burbridge and Neff (1976) observed no refuge effect from road closures. Burbridge and Neff (1976) indicated weather and livestock grazing as the major factors influencing big game distributions.

Observations by various researchers support the hypothesis that cervids get used to various disturbance factors. Ward et al. (1976) reported that mule deer live within 0.5 mile of Interstate 80 and frequently forage within 100 yards of fast-moving traffic where noise levels of over 70 DBA are common. Ward et al. (1973) concluded that elk are very adaptable. Vehicular traffic on I-80 had little effect on elk behavior within 300 yards. They further indicated that logging and recreation roads with moving traffic have little effect on elk activity once they become used to them. Altmann (1952) reported elk respond little to traffic after being near the road for some time. Carbaugh et al. (1975) observed white-tailed deer frequently foraging, resting, or moving along a Pennsylvania Interstate Highway. Beall (1976) observed elk feeding within 100 yards of a skidding operation. The elk gave no indication of being bothered by the noise. Sight of human movement alerted them, however.

Farming and Urban/Suburban Development

Farming and housing developments affect deer and elk by altering or destroying habitat and by creating a greater disturbance factor caused by related human activity. Most farming in the western states results from availability of irrigation water; hence, the first habitats destroyed were those adjacent to dependable free water sources. Removal of riparian woody cover along the Gila, Colorado, and Rio Grande rivers was one significant factor influencing mule deer declines in these areas.

Although about 40 million acres of land have been brought under irrigation farming in the 17 western states (Wallmo et al. 1976), every conversion has not had detrimental effects on habitat of elk and deer. Development of extensive desert tracts has had relatively neutral effects on deer populations. Small farmed areas scattered within accepted cover may enhance cervid habitat by providing high-quality supplements to diets. The heavier weights and excellent physical condition of mule deer inhabiting the Bosque del Apache Wildlife Refuge in New Mexico (Cotton 1979) were probably caused by high-quality diets derived from food patches farmed for wildlife use as well as from surrounding privately owned lands. These circumstances also create problems. Although mule deer may spend only a small amount of their total time on croplands (Severson and Carter 1978), the damage they cause, real and imagined, can result in considerable controversy.

Suburban housing developments, including summer home, can affect cervid populations, not only by destroying habitat, but also by making adjacent areas unusable because of a general increase in overall human activity. This includes the introduction of free-roaming family dogs which can influence deer directly through mortality and indirectly by displacing them from their home range (Knipe 1977). The overall impact is, therefore, greater than indicated by analysis of the actual land area impacted (Wallmo et al. 1976). The problem is ever-increasing with the current demand for homes in unincorporated subdivisions. Davis (1982) reported 36,000 homes were built in 1981 in such subdivisions in Arizona and projected 40,000 and 43,000 would be constructed in 1982 and 1983.

Another problem is that many developments were located on wintering areas which are critical to both elk and deer. This situation was created by land ownership patterns, general topography, and previously located transportation and utility corridors (Wallmo et al. 1976).

Despite problems associated with habitat destruction by farming and suburbanization in the Southwest, the impacts of these uses are minimal compared to effects of timber management and livestock grazing. The areas involved, except for a few local situations, make up an insignificant proportion of the total cervid range in the two states. Particular care must be taken first to identify and then to protect critical ranges, however.

Management Implications

This paper, by design, emphasizes only habitats of mule deer, white-tailed deer, and Rocky Mountain elk in the Southwest (featured species approach); habitat requirements for coexisting wildlife species are not considered (species richness approach). If, however, habitat needs of featured species vary widely, then featured species management can result in habitat diversity, and the result can be similar to management for species richness (Thomas 1979). The variety of vegetation types, including seral and climax stages, and the possible combinations of pattern and edge

that could be used to develop cover to forage area ratios for optimum cervid habitat suggest an enhancement of diversity which would benefit other wildlife species equally as well.

We have also restricted discussion to functions, such as logging, grazing, and fire, as independent entities. This, of course, is seldom the case. Thinning a forest stand to provide an additional forage base, for example, may not only provide more food for deer and elk, but could also reduce competitive interactions with livestock by eliminating forage as the factor in "short supply." Conversely, using prescribed fire to eliminate invading conifers from a meadow used by elk may also attract cattle and result in undesirable ungulate concentrations if proper precautions are not taken. Clearly, managers must consider the direct and peripheral effects of any habitat manipulation on all resources. Research has seldom considered all effects in single studies; hence, managers must glean related data from other investigations. This paper provides a literature base from which such reviews can be initiated.

Livestock grazing has been, and will continue to be, an enigma to wildlife habitat managers. Although excessive grazing in the late nineteenth century may have contributed to the large mule deer populations reported over much of the West in the early-to-mid-1900's, its overall effects must be considered detrimental because of adverse effects on site productivity. The habitat alterations apparently favored deer but created conditions unsuitable for other wild species. Land managers have a continual obligation to guard against excessive grazing by both livestock and wild ungulates—not just for the enhancement of wildlife habitat, but to maintain maximum productivity and viability of the total resource.

This does not, however, preclude use of livestock to manipulate vegetation for wildlife habitat enhancement. Biologists tend to regard livestock grazing as only a negative influence, but positive results can be obtained with proper manipulation of stocking rates, kinds of livestock, intensity and season of use and livestock distributions. The problem is that research has not provided enough specific information applicable to the Southwest. The studies discussed in the section on livestock grazing as a wildlife habitat improvement technique provide a basic direction, and the ubiquity of livestock grazing as a land use in Arizona and New Mexico suggests a useful tool if functional and feasible methods can be developed.

Range managers must be attentive to the complex interactions between cervids and livestock when proposing range improvements designed to benefit livestock primarily. Pinyon-juniper treatments are an example. Extensive areas are cleared and reseeded in an attempt to increase forage production and subsequently increase livestock use. There is little evidence to indicate that deer and elk have significantly benefited from such practices. The practices may, in fact, be a detriment. In some areas, like above the Mogollon Rim, cover is often more important, especially during severe winters. Furthermore, little is known about the behavioral responses of deer and elk to the modified habitat and increased livestock use.

Wildlife managers tend to view the use of prescribed fire favorably, particularly regarding deer and elk. Some information is available that can be useful in developing prescriptions for cervid habitat. Variability of results, however, espe-

cially on effects of intensity, season, and frequency of burning vegetation type on nutritive values of forage plants, indicates a need for more precise information. Peripheral information from fire studies on fuel reduction and timber stand improvement is available, but managers seeking to improve habitat with prescribed fire should develop objectives which can be realistically obtained.

The effects of human activities are difficult to assess. Direct effects such as the habitat destruction resulting from road construction, channelization, and suburban development are generally obvious. Indirect effects such as increased or continual human disturbance from hikers, campers, and off-road vehicle users, increased number of free-roaming dogs, and increased poaching activities may not necessarily be related to habitat conditions. However, optimum habitats with these influences will produce relatively more animals than poor habitats without them. Often-times, many of these factors are beyond the control of the habitat manager, but there are cases where the profound disturbances could be banned from critical areas such as key winter ranges, some riparian habitats, and calving areas (in season). Careful documentation of disturbance effects may eventually result in an adequate data base with which to assess impacts of future proposals.

Integration of timber harvest and timber stand improvement needs with wildlife habitat requirements has been developed to a greater extent. While this is partly due to a more extensive data base, it is primarily a result of the accumulation, synthesis, and interpretation of this data base into a set of guidelines to predict effects of forest management alternatives on wildlife populations (Thomas 1979). Although data in the guides are applicable to one locale, the concepts and procedures can be used to prepare similar guidelines for other forested areas.

Research Needs

There are a large number of problems that require specific attention. Many deal with techniques or effects and may have regional or local application. Several have been alluded to in preceding sections.

In the broader sense, however, habitat management for all wildlife species, including deer and elk, will continue to center around other dominant wildland uses: timber management and grazing by livestock. Recent legislation, however, has dictated that wildlife be specifically considered in land management plans on all federal- and some state-owned lands, a dramatic change from the historical role of wildlife's being accepted only as a by-product of other uses.

The integration of wildlife habitat needs and timber management/harvest has received considerable attention. Thomas' (1979) development of a framework for planning which considers impacts of changes in forest structure on all forms of wildlife is a significant accomplishment. The procedures outlined may require modification for Arizona and New Mexico forest lands, however. A significant effort should be made by research and management to verify, modify, and implement these procedures for the Southwest.

One of the more important questions concerns cover requirements, particularly thermal cover. The large differences in apparent summer thermal cover requirements found between geographic locations, as previously discussed, indicate a more precise definition is needed. Also, most of the information that

is available pertains only to forested habitats. Criteria for both security and thermal cover in grassland, shrubland, and some woodland habitats have not been defined. This requires some emphasis because these habitats are characteristic of many winter and year-long ranges. Although suggested criteria have been for summer ranges, they are those characterized by relatively mild summers, and we cannot logically accept the same criteria to hold for the hot, dry conditions found on southwestern rangelands.

Integration of wildlife habitat needs and management of livestock has not been refined. Information is available, but most research has been directed at food habits as related to areas occupied by deer, elk, and livestock. Hence, most recommendations have involved "common use" stocking rates. This is an important phase, but such information is not sufficient to answer land manager's questions. Behavior or social interactions, for example have only been subjectively assessed. These relationships could be very important, particularly in restricted areas as riparian habitats and in rest- and deferred-rotation grazing systems.

Most important, however, is that not enough attention has been directed to determining conditions and methods under which livestock grazing could be used to improve wildlife habitat. What levels of plant utilization, forage quality, seasonal animal intake, or range conditions could create optimum conditions for other grazing animals? The ubiquity of livestock grazing as a land use throughout the Southwest suggests a powerful tool if methods could be developed and refined. Implicit in this research would be an assessment of effects on livestock.

Prescribed burning also appears to be one of the more promising tools for wildlife habitat improvement, especially deer and elk. It can enhance diversity, edge, forage production, and may have a significant effect on nutritive value of forage plants. Fire was a natural part of many southwestern ecosystems, which means it could be used with a relative degree of safety because native plants and animals have evolved under its presence. It is also cheaper and more environmentally acceptable than mechanical or chemical techniques. It can be used in conjunction with timber harvest or fuelwood cutting to improve their effects and in areas such as chaparral where other options are not particularly feasible. Most existing research in the Southwest, however, has emphasized its possible role in timber stand improvement and fuel reduction programs. Little information exists regarding specific prescriptions for wildlife habitat improvement.

Nutritional relationships have received considerable emphasis in the last 10 years, but most studies have been directed at nutrient content of forage and animal diets. Little information is available on cervid nutrient requirements, especially for mule deer and elk. Previous research on nutrition of cervids has created more questions than it has answered. Notably, how important are the suspected deficiencies to cervid health and productivity, what caused the deficiencies to develop if indeed they did, and, if important, what can be done about it?

Short (1979) explored reasons to explain recent declines of deer populations in the Southwest by using computer simulation techniques. This method supported the hypothesis that these declines have been caused by increased female mortality (because of antlerless deer hunting) and relatively low fawn recruitment rates (because of habitat quality). Short (1979) further indicated that habitat quality is limited by forage nutritional deficiencies resulting from existing soil conditions and

seasonal precipitation patterns. Although these two causes are basically nonmanipulative, the nutritional status of forage can be mitigated even under these circumstances through manipulation of other factors, notably prescribed fire, grazing by other animals, and controlling overstory vegetation. Research is needed, however, to further define circumstances under which a particular technique can or cannot be used.

A careful analysis of cervid health and productivity should be implicit in any future experimental designs evaluating habitat improvement research. We have put too much emphasis on indirect methods, such as pellet group counts, to evaluate deer and elk response to habitat change. Even more refined techniques, as time-lapse photography and radio telemetry, while more accurately describing habitat use, do not provide any indication of population growth or decline. Justification for the use of any tool or modification of other land uses, whether timber harvest or livestock grazing, for wildlife habitat improvement will require information on the numerical response of deer and elk herds.

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APPENDIX A. Common and Scientific Names of Animals¹



Antelope, pronghorn
 Bear, grizzly
 Burro, feral
 Deer
 Deer, mule
 Deer, black-tailed
 Deer, burro
 Deer, desert mule
 Deer, Rocky Mountain mule
 Deer, white-tailed
 Deer, Columbian white-tailed
 Deer, Coues white-tailed
 Deer, Texas white-tailed
 Elk
 Elk, Rocky Mountain
 Elk, Merriam
 Hog, feral
 Horse, feral
 Ibex, Persian
 (or Persian wild goat)
 Ibex, Siberian
 Javelina
 Lion, mountain
 Oryx
 Quail, Gambel
 Quail, masked bobwhite
 Quail, scaled
 Sheep, Barbary
 Sheep, desert bighorn
 Wolf
 Woodpecker, pileated

Antilocapra americana
Ursus horribilis
Equus asinus
Odocoileus spp.
Odocoileus hemionus
O. h. columbianus
O. h. eremicus
O. h. crooki
O. h. hemionus
Odocoileus virginianus
O. v. leucurus
O. v. couesi
O. v. texanus
Cervus elaphus
C. e. nelsoni
C. e. merriami
Sus scrofa
Equus caballus

Capra aegagrus
Capra siberica
Dicotyles tajacu
Felis concolor
Oryx gazella
Lophortyx gambelii
Colinus virginianus ridgwayi
Callipepla squamata
Ammotragus lervia
Ovis canadensis mexicana
Canis lupus
Dryocopus pileatus

¹After Cockrum (1960), Findley et al. (1975), and Decker (1978).

APPENDIX B. Common and Scientific Names of Plants¹



Grasses, Sedges, Rushes
 Bluegrass
 Brome, foxtail
 Brome, smooth
 Fescue, Arizona
 Fescue, tall
 Lovegrass, Lehmann
 Lovegrass, weeping
 Muhly, mountain
 Needlegrass
 Orchardgrass
 Sedge
 Squirreltail, bottlebrush
 Timothy
 Wheatgrass
 Wheatgrass, fairway
 Forbs
 Agoseris
 Alfalfa
 Alfileria
 Amaranth
 Amaranth, tumbleweed
 Ayenia, dwarf
 Bahia
 Bedstraw
 Bundleflower, James
 Clover
 Dalea
 Dayflower birdbill
 Deervetch
 Deervetch, shrubby
 Deervetch, Wright
 Euphorbia
 Eriogonum
 Eriogonum, Wright

Poa spp.
Bromus rubens
Bromus inermis
Festuca arizonica
Festuca arundinacea
Eragrostis lehmanniana
Eragrostis curvula
Muhlenbergia montana
Stipa spp.
Dactylis spp.
Carex spp.
Sitanion hystrix
Phleum pratense
Agropyron spp.
Agropyron cristatum

Agoseris spp.
Medicago sativa
Erodium cicutarium
Amaranthus spp.
Amaranthus graecizans
Ayenia pilosa
Bahia spp.
Galium spp.
Desmanthus cooleyi
Trifolium spp.
Dalea spp.
Commelina dianthifolia
Lotus spp.
Lotus rigidus
Lotus wrightii
Euphorbia spp.
Eriogonum spp.
Eriogonum wrightii

Fleabane
 Four-o'clock, Bigelow
 Geranium
 Globemallow
 Glorybind
 Groundcherry
 Hoarhound, common
 Larkspur
 Mahonia, creeping
 Mallow
 Mariposa
 Milkvetch
 Morningglory, pinkthroated
 Mullein
 Peavine
 Peavine, grassleaf
 Penstemon
 Sagebrush, Louisiana
 Sneezeweed, orange
 Sweetclover
 Thistle
 Wirelettuce
 Yarrow, western

Shrubs and Trees
 Acacia
 Acacia, catclaw
 Acacia, mesquit
 Apacheplume
 Aspen, quaking
 Ash, velvet
 Beargrass
 Bitterbrush, antelope
 Buckthorn, hollyleaf
 Cactaceae, barrel
 Calliandra, falsemesquite
 Ceanothus, deerbrush
 Ceanothus, desert
 Ceanothus, Fendler
 Ceanothus, wedgeleaf
 Cercocarpus
 Cercocarpus, birchleaf
 Cercocarpus, hairy
 Cherry, southwestern black
 Cholla
 Cholla, jumping
 Cholla, spiny
 Cliffrose
 Cottonwood
 Douglas-fir
 Fiddleberry
 Fendlerbush, cliff
 Fir, white
 Grape, canyon
 Hackberry, netleaf
 Hackberry, spiny
 Janusia, slender
 Jerusalem-thorn
 Jojoba, California
 Juniper
 Juniper, alligator
 Juniper, one-seed
 Juniper, Rocky Mountain
 Juniper, Utah
 Kidneywood
 Krameria, littleleaf
 Lechuguilla
 New Mexico locust
 Manzanita, mariposa
 Mesquite
 Mimosa, velvetpod
 Mistletoe
 Mulberry, Texas
 Oak
 Oak, Arizona white
 Oak, bur
 Oak, Emory

Erigeron spp.
Mirabilis bigelovii
 Geranium spp.
Sphaeralcea spp.
Convolvulus spp.
Physalis spp.
Marrubium vulgare
Delphinium spp.
Berberis repens
Malva spp.
Calochortus spp.
Astragalus spp.
Ipomoea longifolia
Verbascum spp.
Lathyrus spp.
Lathyrus graminifolius
Penstemon spp.
Artemisia ludoviciana
Helenium hoopesii
Melilotus spp.
Cirsium spp.
Stephanomeria exigua
Achillea lanulosa

Acacia spp.
Acacia greggii
Acacia constricta
Fallugia paradoxa
Populus tremuloides
Fraxinus velutina
Nolina spp.
Purshia tridentata
Rhamnus crocea
Echinocactus wislizenii
Calliandra eriophylla
Ceanothus integerrimus
Ceanothus greggii
Ceanothus fendleri
Ceanothus cuneatus
Cercocarpus spp.
Cercocarpus betuloides
Cercocarpus breviflorus
Prunus serotina rufula
Opuntia spp.
Opuntia fulgida
Opuntia spinosior
Cowania mexicana
Populus spp.
Pseudotsuga menziesii
Sambucus spp.
Fendlera rupicola
Abies concolor
Celtis reticulata
Celtis pallida
Janusia gracilis
Parkinsonia aculeata
Simmondsia chinensis
Juniperus spp.
Juniperus deppeana
Juniperus monosperma
Juniperus scopulorum
Juniperus osteosperma
Eysenhardtia polystachya
Krameria parvifolia
Agave spp.
Robinia neomexicana
Arcostaphylos mariposa
Prosopis spp.
Mimosa dysocarpa
Phoradendron spp.
Morus microphylla
Quercus spp.
Quercus arizonica
Quercus macrocarpa
Quercus emoryi

Oak, Gambel
 Oak, gray
 Oak, netleaf
 Oak, shrub live
 Oak, silverleaf
 Oak, wavyleaf
 Ocotillo
 Pachistimia, myrtle
 Paloverde, yellow
 Pine
 Pine, lodgepole
 Pine, pinyon
 Pine, ponderosa
 Porlieria, Texas
 Pricklypear
 Pricklypear, Engelmann
 Sagebrush
 Sagebrush, big
 Saltbush

Quercus gambelii
Quercus grisea
Quercus reticulata
Quercus turbinella
Quercus havardii
Quercus undulata
Fouquieria splendens
Pachistima myrsinites
Cercidium microphyllum
Pinus spp.
Pinus contorta
Pinus edulis
Pinus ponderosa
Porlieria angustifolia
Opuntia spp.
Opuntia engelmannii
Artemisia spp.
Artemisia tridentata
Atriplex spp.

Saltbush, fourwing
 Silktassel, Wright
 Silktassel, yellowleaf
 Smokethorn
 Snakeweed, broom
 Sotol
 Spruce
 Spruce, blue
 Sumac, Mearns
 Sumac, skunkbush
 Sumac, sugar
 Tamarisk
 Thimbleberry, western
 Vauquelinia, Torrey
 Whortleberry, grouse
 Willow
 Yucca
 Yucca, soaptree

Atriplex canescens
Garrya wrightii
Garrya flavescens
Dalea spinosa
Gutierrezia sarothrae
Dasylium spp.
Picea spp.
Picea pungens
Rhus choriophylla
Rhus trilobata
Rhus ovata
Tamarix ramosissima
Rubus parviflorus
Vauquelinia californica
Vaccinium scoparium
Salix spp.
Yucca spp.
Yucca elata

¹After Nickerson et al. (1976).

APPENDIX C. Tables

Table C1. Relative preference of plant species to mule deer in the South at different seasons. Letters L, M, or H indicate that the plant item composed 1-5%, 6-15% and 16% or more, respectively, of the animals diet (after Short 1979).

| Species in diet | Winter | Spring | Summer | Autumn | Reference ¹ |
|-------------------------------------|----------------|----------------|----------------|----------------|------------------------|
| Browse (including shrubs and trees) | | | | | |
| Acacia | L | L | L | L | 2,3 |
| Acacia, catclaw | L | H | H | M | 1,2,3,4,6,7 |
| Acacia, mescat | L | M | L | L | 2,3,6,7 |
| Ayenia, dwarf | M | L | L | H | 7 |
| Buckthorn, hollyleaf | M | M | L | L | 4,5 |
| Calliandra, falsemesquite | H | H | H | H | 1,4,6,7 |
| Ceanothus, desert | L | M | L | L | 4,5 |
| Cercocarpus, birchleaf | M | H | L | L | 4,5 |
| Eriogonum | H | L | L | M | 4,6 |
| Eriogonum, Wright | L | M | M | M | 1,7 |
| Hackberry, netleaf | L | M | L | L | 1,4 |
| Janusia, slender | M | O | L | M | 4,7 |
| Jojoba, California | M | H | H | M | 5,7 |
| Juniper | H ² | L | L | M | 1,5 |
| Kidneywood | H | M | H | H | 1,6 |
| Krameria, littleleaf | M | H | M | H | 1,4,7 |
| Lechuguilla | H | H | L | H | 2,3 |
| Mesquite | H | L | H ² | L | 1,4,6,7 |
| Oak, shrub live | H | O | M ² | L | 4,5 |
| Ocotillo | L | L | L | M | 1,6,7 |
| Paloverde, yellow | O | L | L | L | 4,7 |
| Porlieria, Texas | L | L | M | H | 2,3 |
| Sotol | L | M | M | M | 2,3 |
| Sumac, skunkbush | O | H | L | L | 4,5 |
| Sumac, sugar | L | M | L | L | 4 |
| Yucca | H | M | M | L | 2,3,4 |
| Forbs (including succulents) | | | | | |
| Barrel cactus | H ² | L ² | L ² | H ² | 6,7 |
| Beargrass | L | M | M | L | 1,4 |
| Cholla, jumping | L ² | H ² | L ² | M ² | 6,7 |
| Cholla, spiny | M ² | L ² | L ² | L ² | 6 |
| Deervetch, shrubby | L | L | L | L | 4,7 |
| Euphorbia | M | M | M | M | 1,2,3,4,5,6 |
| Four-o'clock, Bigelow | L | L | L | L | 4,7 |
| Globemallow | L | L | L | L | 1,4 |
| Pricklypear | M | L | H | M ² | 2,3,4 |
| Pricklypear, Engelmann | L ¹ | L ¹ | H ² | H ² | 2,4,6,7 |
| Sagebrush, Louisiana | M | M | L | L | 3,5,7 |
| Wirelettuce | L | M | M | L | 1,7 |
| Grasses | L | M | L | L | 1,2,3,4 |

¹References: 1, Anthony and Smith (1977); 2, Brownlee (1971); 3, Brownlee (1973); 4, McCulloch (1973); 5, McCulloch (1978); 6, Short (1977); 7, Truett (1972).

²Includes fruits.

Table C2. Relative preference of plant species to mule deer in the mountains and associated areas at different seasons. Letters L, M, or H indicate that the plant item composed 1-5%, 6-15%, and 16% or more, respectively, of the animals diet (after Short 1979).

| Species in diet | Winter | Spring | Summer | Autumn | Reference ¹ |
|--|----------------|--------|--------|----------------|------------------------|
| Browse (including shrubs and trees) | | | | | |
| Apacheplume | L | L | L | L | 1,6 |
| Aspen, quaking | O | M | H | H | 3,6 |
| Ceanothus, desert | M | L | M | O | 7 |
| Ceanothus, Fendler | O | L | L | L | 3,6 |
| Cercocarpus, hairy | H | H | H | H | 1,2,4,6,7,8 |
| Cliffrose | H | L | M | M | 5,6,7 |
| Eriogonum | L | L | L | L | 4,5,6 |
| Eriogonum, Wright | M | M | O | H | 7 |
| Fir, white | — ³ | L | M | M | 3,6 |
| Juniper | M | L | M | H ² | 1,6,8 |
| Juniper, alligator | L | L | L | L | 4 |
| Juniper, Utah | M | M | — | H | 5,6 |
| Mahonia, creeping | O | M | M | M | 3,6 |
| Mistletoe, juniper | L | L | L | M | 1,4,5,6,8 |
| Oak | H | H | H | H ² | 2,6,8 |
| Oak, Gambel | H | H | H | H ² | 5,6,7 |
| Oak, gray | H | H | L | H ² | 4 |
| Oak, shrub live | H | M | O | M | 6,7 |
| Oak, wavyleaf | L | H | H | H | 1,7 |
| Pine, ponderosa | H | M | L | M | 3,6,7 |
| Pine, pinyon | M | H | O | L ² | 1,5,6 |
| Sagebrush, big | H | H | O | M | 5,6 |
| Saltbush, fourwing | L | L | L | L | 1,5,6 |
| Silktassel, Wright | L | L | L | L | 1,2,4,6,8 |
| Smokethorn | O | M | H | L | 4,8 |
| Snakeweed, broom | L | L | L | L | 1,5,6 |
| Sumac, skunkbush | O | L | M | M | 1,2,4 |
| Forbs (including succulents) | | | | | |
| Bundleflower, James | O | M | M | L | 4,6,7,8 |
| Cholla | M ² | O | O | L ² | 1,8 |
| Clover | O | M | H | M | 1,6,7 |
| Deervetch | L | L | L | L | 4,5,6 |
| Deervetch, Wright | O | L | M | O | 3,7 |
| Fleabane | O | L | L | L | 5,6,7 |
| Geranium | M | L | L | O | 6,7 |
| Globemallow | O | L | L | L | 4,5,6 |
| Milkvetch | L | M | M | L | 3,5,6 |
| Peavine | O | L | L | L | 4,7 |
| Penstemon | L | L | L | M | 5,6 |
| Sagebrush, Louisiana | L | L | M | L | 1,6 |
| Sweetclover | O | L | L | M | 3,4,6,7 |
| Grass and Grasslike | | | | | |
| Bluegrass | L | L | L | L | 3,5,6,7 |
| Orchardgrass | O | M | M | O | 3,7 |
| Squirreltail, bottlebrush | L | M | L | O | 3,6,7 |
| Wheatgrass, fairway | M | O | L | M | 3,7 |

¹References: 1, Anderson et al. (1965); 2, Boeker et al. (1972); 3, Hungerford (1970); 4, Hunt (1978); 5, McCulloch (1969); 6, McCulloch (1978); 7, Neff (1974); 8, Short et al. (1977).

²Includes fru.'s.

³—Indicates no data.

Table C3. Approximate composition of seasonal diets of mule deer in the South.

| Geographic area (method of analysis) Habitat | Season | Browse | Forbs | Grass | Reference ¹ |
|---|--------|--------|-------|-------|------------------------|
| | | | | | |
| Brewster County, Texas (Rumen analysis) | Winter | 83 | 12 | 1 | 2 |
| | Spring | 76 | 17 | 4 | 2 |
| | Summer | 33 | 65 | 1 | 2 |
| | Autumn | 48 | 48 | 0 | 2 |
| | Winter | 61 | 38 | 1 | 3 |
| | Spring | 33 | 67 | 1 | 3 |
| Chihuahuan Desertscrub | Summer | 37 | 58 | 1 | 3 |
| | Autumn | 70 | 29 | 1 | 3 |
| | Winter | 87 | 6 | 4 | 1 |
| San Cayetano Mountains, Arizona (Fecal analysis) | Winter | 87 | 6 | 4 | 1 |
| | Spring | 70 | 27 | 7 | 1 |

Table C3—Cont.

| Geographic area (method of analysis) Habitat | Season | Browse | Forbs | Grass | Reference ¹ |
|--|-------------------------------------|------------------------------|-------|-------|------------------------|
| | | Percent of diet ² | | | |
| Semidesert Grasslands to Oak Woodlands | Summer | 77 | 18 | 3 | 1 |
| | Autumn | 89 | 5 | 6 | 1 |
| Dos Cabezas Mountains, Arizona (Fecal analysis) | Winter-Spring | 96 | 2 | 1 | 1 |
| | Spring-Summer | 84 | 7 | 7 | 1 |
| Semidesert Grasslands to Oak Woodlands | Summer-Autumn | 80 | 8 | 4 | 1 |
| Santa Rita Mountains, Arizona (Rumen analysis) | Winter | 37 | 60 | 1 | 6 |
| | Spring | 42 | 52 | 3 | 6 |
| Semidesert Grasslands | Summer | 62 | 27 | 0 | 6 |
| | Autumn | 51 | 47 | 0 | 6 |
| Mazatzal Mountains, Arizona (Rumen analysis) | Winter (Jan.) | 75 | 21 | 4 | 4 |
| | Winter (Feb.-Mar.) | 53 | 45 | 2 | 4 |
| Arizona Uplands-Interior Chaparral | Spring (May) | 70 | 30 | 0 | 4 |
| Mazatzal Mountains, Arizona (Rumen analysis) | Summer (July) | 90 | 10 | T | 4 |
| | Autumn (Oct.) | 92 | 8 | T | 4 |
| Arizona Uplands-Interior Chaparral | Autumn (Nov.) | 89 | 6 | 5 | 4 |
| | Autumn (Nov.) (after dry summer) | | | | |
| | Autumn (Nov.) (after wet summer) | 74 | 20 | 6 | 4 |
| Prescott National Forest, Arizona (Rumen analysis) Interior Chaparral | Winter | 100 | T | 0 | 5 |
| | Spring | 90 | 10 | 0 | 5 |
| Bloody Basin, Tonto National Forest, Arizona (Rumen analysis) Interior Chaparral | Winter | 70 | 30 | 0 | 5 |
| | Spring | 38 | 62 | 0 | 5 |

¹References: 1, Anthony and Smith (1977); 2, Brownlee (1971); 3, Brownlee (1973); 4, McCulloch (1973); 5, McCulloch (1978); 6, Short (1977).
²Totals may not equal 100% because of rounding errors and omission of other categories, e.g., "unidentified material."

Table C4. Approximate composition of seasonal diets of mule deer in the Mountains and Associated Areas.

| Geographic area (method of analysis) Habitat | Season | Browse | Forbs | Grass | Reference ¹ |
|---|--------------|------------------------------|-------|----------------|------------------------|
| | | Percent of diet ² | | | |
| Guadalupe Mountains, New Mexico (Rumen analysis) | Winter | 80 | 17 | 3 | 1 |
| | Spring | 68 | 31 | T ³ | 1 |
| Semidesert grasslands-Pinyon- Juniper | Summer | 62 | 34 | T | 1 |
| | Autumn | 71 | 28 | T | 1 |
| Fort Bayard, New Mexico (Rumen analysis) Pinyon-Juniper | Winter | 86 | 10 | 1 | 8 |
| | Spring | 58 | 32 | 1 | 8 |
| (Tame deer-bite count) | Summer | 50 | 42 | 1 | 8 |
| | Autumn | 86 | 5 | 4 | 8 |
| | Winter | 94 | 2 | 1 | 2 |
| | Spring | 58 | 32 | 1 | 2 |
| | Summer | 50 | 42 | 1 | 2 |
| | Autumn | 87 | 4 | 4 | 2 |
| Hualapai Reserve, Arizona (Rumen analysis) Pinyon-Juniper, unburned Pinyon-Juniper, burned | Winter | 99 | T | 0 | 4 |
| | Spring | 91 | 7 | 2 | 4 |
| | Summer | 45 | 55 | T | 4 |
| | Autumn | 88 | 12 | T | 4 |
| Hualapai Reserve, Arizona (Rumen analysis) Pinyon-Juniper, burned | Autumn | 60 ⁴ | 33 | 7 | 5 |
| North Rim, Kaibab National Forest, Arizona (Rumen analysis) Pinyon-Juniper | Autumn | 15 | 52 | 33 | 5 |
| | Early winter | 66 | 13 | 21 | 6 |
| | Early winter | 62 | 20 | 18 | 6 |
| | Midwinter | 98 | T | 2 | 6 |
| Pinyon-Juniper | Midwinter | 87 | 1 | 12 | 6 |
| | Late winter | 75 | 1 | 24 | 6 |

Table C4—Cont.

| Geographic area (method of analysis) Habitat | Season | Browse | Forbs | Grass | Reference ¹ | |
|--|------------------|------------------------------|-------|----------------|------------------------|---|
| | | Percent of diet ² | | | | |
| North Rim, Kaibab National Forest, Arizona (Rumen analysis) | Late winter | 83 | 2 | 15 | 6 | |
| | Winter | 72 | 10 | 18 | 6 | |
| Pinyon-Juniper, trees uncut | Late autumn | 46 | 34 | 20 | 6 | |
| | Late autumn | 30 | 48 | 22 | 6 | |
| Pinyon-Juniper, trees cut | Autumn | 67 | 33 | T | 6 | |
| | Autumn | 41 | 58 | 1 | 6 | |
| Beaver Creek, Coconino National Forest, Arizona | Autumn | 22 ⁴ | 78 | T | 6 | |
| | Autumn | 51 ⁴ | 45 | 4 | 7 | |
| SE of Mormon Lake, AZ (Tame deer-bite count) | Winter | | | | | |
| Pinyon-Juniper, untreated | | | | | | |
| Beaver Creek, Coconino National Forest, Arizona (Tame deer-bite count) | Spring | 69 | 24 | 7 | 7 | |
| | Summer | 75 | 24 | 1 | 7 | |
| | Yearlong | 62 | 31 | 7 | 7 | |
| Pinyon-Juniper, untreated | | | | | | |
| Pinyon-Juniper, cabled, burned, reseeded | Winter | 68 | 22 | 10 | 7 | |
| | Spring | 21 | 33 | 46 | 7 | |
| | Summer | 45 | 53 | 2 | 7 | |
| Pinyon-Juniper, cable, burned, reseeded | Autumn | 33 | 66 | 1 | 7 | |
| | Yearlong | 44 | 48 | 8 | 7 | |
| | Yearlong | 44 | 48 | 8 | 7 | |
| Pinyon-Juniper, herbicide | Spring | 2 | 83 | 15 | 7 | |
| | Summer | 28 | 62 | 10 | 7 | |
| | Autumn | 1 | 88 | 11 | | |
| | Yearlong | 18 | 71 | 11 | | |
| Pinyon-Juniper, untreated | Winter | 94 | 5 | 1 | 7 | |
| | Spring | 66 | 21 | 13 | 7 | |
| | Summer | 51 | 47 | 2 | 7 | |
| | Autumn | 60 | 39 | 1 | 7 | |
| | Yearlong | 65 | 29 | 6 | 7 | |
| Pinyon-Juniper, trees out | Winter | 70 | 11 | 19 | 7 | |
| | Spring | 45 | 15 | 40 | 7 | |
| | Summer | 22 | 69 | 9 | 7 | |
| Beaver Creek, Coconino National Forest, Arizona (Tame deer-bite count) | Autumn | 14 | 53 | 33 | 7 | |
| | Yearlong | 41 | 34 | 25 | 7 | |
| Pinyon-Juniper, trees cut | | | | | | |
| Ponderosa pine, untreated | Winter | 64 | 33 | 3 | 7 | |
| | Spring | 23 | 45 | 32 | 7 | |
| | Summer | 53 | 44 | 3 | 7 | |
| | Yearlong | 46 | 44 | 10 | 7 | |
| | Yearlong | 48 | 18 | 34 | 7 | |
| Ponderosa pine, various treatments | Winter | 48 | 18 | 34 | 7 | |
| | Spring | 33 | 36 | 31 | 7 | |
| | Summer | 52 | 46 | 2 | 7 | |
| | Autumn | 58 | 40 | 2 | 7 | |
| | Yearlong | 46 | 39 | 15 | 7 | |
| North Rim, Kaibab National Forest, Arizona (Rumen analysis) | Spring | 47 | 51 | 2 | 6 | |
| | Autumn | 62 ⁴ | 36 | 1 | 6 | |
| Ponderosa pine (Feeding minutes) | June | 34 | 19 | 47 | 3 | |
| | June | 21 | 34 | 44 | 3 | |
| Ponderosa pine to spruce-fir | July | 47 | 34 | 18 | 3 | |
| | July | 47 | 49 | 4 | 3 | |
| | August | 33 | 1 | 0 ⁵ | 3 | |
| | August | 52 | 16 | 0 ⁵ | 3 | |
| | (Rumen analysis) | Midsummer | 58 | 39 | 2 ⁵ | 6 |
| | Mixed conifer | Early summer | 36 | 62 | 2 | 6 |
| Mountain meadows | Summer | 18 | 71 | 11 | 6 | |
| | Autumn | 49 | 48 | 3 | | |
| Willow Creek, White Mountains, Arizona (Rumen analysis) | Autumn | 65 | 13 | 8 ⁵ | 6 | |
| | Autumn | 52 | 41 | 5 ⁵ | 6 | |
| Mixed conifer | | | | | | |

¹References: 1, Anderson et al. (1965); 2, Boeker et al. (1972); 3, Hungerford (1970); 4, Hunt (1978); 5, McCulloch (1969); 6, McCulloch (1978); 7, Neff (1974); 8, Short et al. (1977).

²Totals may not equal 100% because of rounding errors and omission of other categories, e.g., "unidentified material."

³T—indicates items occurred in trace amounts, less than 0.5%

⁴Includes mast.

⁵Diet includes various mushrooms to total 100%.

Table C5. Approximate composition of seasonal diets of Coues white-tailed deer in different habitats.

| Geographic Area (method of analysis) Habitat | Season | Browse | Forbs | Grass | Reference ¹ |
|--|---------------|------------------------------|-------|----------------|------------------------|
| | | Percent of diet ² | | | |
| Santa Rita Mountains, Arizona (% feeding time) | August | 91 | 4 | 5 | 4 |
| | September | 45 | 51 | 4 | 4 |
| Oak Woodland | October | 97 | 0 | 3 | 4 |
| Chiricahua Mountains, Arizona (% feeding time) | August | 79 | 21 | 0 | 4 |
| | September | 8 | 87 | 5 | 4 |
| Ponderosa Pine | October | 9 | 91 | 0 | 4 |
| Semidesert Grasslands to Ponderosa Pine (Rumen analysis) | Autumn | 98 | 2 | 0 | 2 |
| | Yearlong | 92 | 8 | 0 | 2 |
| Mazatzal Mountains, Arizona (Rumen analysis) | Winter | 67 | 28 | 5 | 3 |
| | Winter | 68 | 27 | 5 | 3 |
| Arizona Uplands-Interior Chaparral | Winter | 50 | 40 | 10 | 3 |
| | Spring | 90 | 10 | T ³ | 3 |
| | Summer | 91 | 9 | T | 3 |
| | Autumn | 65 | 35 | 0 | 3 |
| | Autumn | 49 | 48 | 3 | 3 |
| San Cayetano Mountains, Arizona (Fecal analysis) | Winter | 85 | 2 | 6 | 1 |
| | Spring | 57 | 12 | 18 | 1 |
| Semidesert Grasslands to Oak Woodlands | Summer | 70 | 14 | 7 | 1 |
| | Autumn | 73 | 7 | 9 | 1 |
| Dos Cabezas Mountains, Arizona (Fecal analysis) | Winter-Spring | 93 | 2 | 1 | 1 |
| | Spring-Summer | 74 | 3 | 8 | 1 |
| Semidesert Grasslands to Oak Woodlands | Summer-Autumn | 83 | 10 | 6 | 1 |

¹References: 1, Anthony and Smith (1977); 2, Day (1964); 3, McCulloch (1973); 4, White (1964).

²Totals may not equal 100% because of rounding errors and omission of other categories, e.g., "unidentified materials."

³T—indicates items occurring in trace amounts, less than 0.5%.

Table C6. Relative preference of plant species to Coues white-tailed deer at different seasons. Letters L, M, or H indicate that a plant item composed 1-5%, 6-15%, and 16% or more, respectively, of animals diet (after Short 1979).

| Species in diet | Winter | Spring | Summer | Autumn | Reference ¹ |
|-------------------------------------|----------------|----------------|----------------|----------------|------------------------|
| Browse (including shrubs and trees) | | | | | |
| Acacia, catclaw | M ² | L | M ² | L | 1,4 |
| Black cherry, southwestern | L | L | O ² | L | 3,4 |
| Buckthorn, hollyleaf | M | M | L | M | 4 |
| Calliandra, falsemesquite | M | M | M | H | 1,4,5 |
| Ceanothus, deerbrush | L | L | L | L ² | 4 |
| Ceanothus, desert | L | L ² | L | M | 2,3,4 |
| Ceanothus, Fendler | O ³ | O | M | H | 3,5 |
| Cercocarpus, hairy | H | H | H | H | 1,2,3 |
| Eriogonum | H | L | L | L | 4 |
| Eriogonum, Wright | H | H | M | M | 1,5 |
| Fendlerbush, cliff | L | M | M | L | 1,3 |
| Grape, canyon | L | L | M | H | 2,3 |
| Hackberry, netleaf | L | M | H | H | 1,3,4 |
| Hackberry, spiny | L | L | L | L | 3 |
| Jerusalem-thorn | L | L | H | O | 3 |
| Jojoba, California | O | L | O | L | 3,4 |
| Juniper | M | M | M | H | 1,2,3 |
| Kidneywood | M | L | H | M | 1,3 |
| Krameria, littleleaf | M | M | M | H | 1,5 |
| Mesquite | M | M | M ² | M | 1,3,4 |
| Mimosa, velvetpod | — ⁴ | — | H | H | 5 |
| Mistletoe | L | O | L | M | 2,4 |
| Mulberry, Texas | O | L | L | L | 3 |
| Oak, Arizona white | M | M | M | M ² | 2,3,4 |
| Oak, Emory | M | M | L ² | L | 1,2,3,4 |
| Oak, Gambel | O | O | M | H | 1,2,3,5 |
| Oak, netleaf | O | M | O | O | 3 |
| Oak, shrub live | L | M | H ² | L | 4 |
| Oak, silverleaf | O | O | M | M | 3,5 |
| Ocotillo | O | L | M | H | 1,5 |
| Silktassel, Wright | H | M | L | M | 2,3 |
| Silktassel, yellowleaf | L | M ² | M ² | M ² | 4 |

Table C6—Cont.

| Species in diet | Winter | Spring | Summer | Autumn | Reference ¹ |
|------------------------------|--------|--------|----------------|--------|------------------------|
| Sumac, Mearns | L | L | M | H | 2,3 |
| Sumac, skunkbush | L | L | H ² | L | 3,4 |
| Sumac, sugar | L | H | O | O | 3 |
| Vauquelinia, Torrey | L | L | L | O | 3 |
| Forbs (including succulents) | | | | | |
| Agoseris | — | — | H | H | 5 |
| Beargrass | L | M | M | O | 1 |
| Cactus, barrel | — | — | H | H | 5 |
| Larkspur | — | — | H | H | 5 |
| Morningglory, pinkthroated | — | — | H | H | 5 |
| Sagebrush, Louisiana | M | L | M | M | 4 |
| Wirelettuce | L | M | L | L | 1 |

¹References:

1. Anthony and Smith (1977)
2. Day (1964)
3. Knipe (1977)
4. McCulloch (1973)
5. White (1961)

²Includes fruits.³O indicates that samples were taken during the season but item was not utilized.⁴—Indicates that samples were not taken that season.

Table C7. Nutritive contents of selected important Southwestern forages. All figures percent of dry weight.

| Species, Plant part ¹ Habitat-area | Time of year | Digestible dry matter ² | | Crude protein | ADF ³ | Ca | P | Reference ³ |
|--|-----------------|---------------------------------------|-----------------|------------------|------------------|------|------|------------------------|
| | | MD ⁴ | CD ⁴ | | | | | |
| Acacia, catclaw (LE) Semidesert Grassland | Spring | 43 | — ⁵ | 14 | — | 0.82 | 0.23 | 1 |
| | Summer | 30 | — | 16 | — | 2.00 | .12 | 1 |
| | Autumn | 34 | — | 17 | — | 1.70 | .12 | 1 |
| | Winter | 24 | — | 9 | — | 2.25 | .09 | 1 |
| Acacia, catclaw (FR) Interior Chaparral | June | 32 | 29 | 15 | 35 | .70 | .28 | 3 |
| Alfileria (WP) Interior Chaparral | Nov. | — | 40 | 20 | 31 | 2.51 | .47 | 3 |
| | Jan. | 57 | 66 | 19 | 24 | 2.55 | .59 | 3 |
| | Mar. | 66 | 60 | 22 | 24 | .66 | .16 | 3 |
| Brome, foxtail (WP) Interior Chaparral | Dec. | — | 74 | 27 | 26 | .33 | .65 | 3 |
| | Jan. | 55 | 52 | 18 | 35 | 1.29 | .64 | 3 |
| | Mar. | 72 | 67 | 11 | 35 | 1.96 | .39 | 3 |
| Cactus, barrel (FR) Semidesert Grassland | Spring | 60 | — | 8 | — | .31 | .20 | 1 |
| | Summer | 78 | — | 9 | — | .44 | .21 | 1 |
| | Autumn | 61 | — | 6 | — | .38 | .18 | 1 |
| | Winter | 74 | — | 11 | — | .38 | .23 | 1 |
| Calliandra, falsemesquite (LE) Interior Chaparral | May | 28 | 20 | 15 | 37 | .91 | .20 | 3 |
| | June | 32 | 29 | 15 | 35 | .70 | .28 | 3 |
| | Sep. | 33 | 32 | 13 | 38 | 1.30 | .29 | 3 |
| | Nov. | 27 | 31 | 11 | 25 | 2.04 | .14 | 3 |
| Cercocarpus, hairy (CAG) Pinyon-juniper | Spring | 47 | — | 16 | — | .72 | .32 | 2 |
| | Summer | 55 | — | 17 | — | 1.15 | .23 | 2 |
| | Autumn | 57 | — | 13 | — | 1.91 | .21 | 2 |
| | Winter | 62 | — | 10 | — | 1.11 | .20 | 2 |
| Cholla, jumping (FR) Semidesert Grassland | Spring | 60 | — | 3 | — | 3.33 | .10 | 1 |
| | Summer | 62 | — | 10 | — | 2.60 | .13 | 1 |
| | Autumn | 48 | — | 8 | — | 1.86 | .13 | 1 |
| | Winter | 44 | — | 5 | — | 2.17 | .13 | 1 |
| Deervetch, Wright (WP) Ponderosa Pine | May | 39 | — | 19 | 31 | 1.13 | .21 | 4 |
| | July | 38 | — | 16 | 40 | 1.44 | .16 | 4 |
| | Sept. | 42 | — | 11 | 39 | 1.56 | .44 | 4 |
| Eriogonum, Wright (WP) Interior Chaparral | Sept. | — | 24 | 8 | 51 | .94 | .15 | 3 |
| | Nov. | 16 | 27 | 5 | 50 | 1.00 | .15 | 3 |
| | Jan | 16 | 13 | 6 | 52 | .61 | .15 | 3 |
| | Mar. | 19 | 22 | 7 | 41 | 1.21 | .19 | 3 |

Table C7—Cont.

| Species, Plant part ¹ Habitat-area | Time of year | Digestible dry matter ² | | Crude protein | ADF ⁵ | Ca | P | Reference ³ |
|---|-----------------|---------------------------------------|-----------------|------------------|------------------|------|-----|------------------------|
| | | MD ⁴ | CD ⁴ | | | | | |
| Jojoba, California (LE) Interior Chaparral | May | 47 | — | 10 | 26 | .57 | .20 | 3 |
| | June | 43 | — | 20 | 42 | .85 | .16 | 3 |
| | Sept. | 40 | — | 13 | 34 | .93 | .17 | 3 |
| | Nov. | 36 | — | 13 | 31 | 1.35 | .12 | 3 |
| | Jan. | 44 | — | 11 | 25 | .93 | .10 | 3 |
| | Mar. | 45 | — | 11 | 29 | 1.53 | .28 | 3 |
| Jojoba, California (ST) | May | 47 | — | 10 | 41 | .31 | .25 | 3 |
| | June | 35 | — | 9 | 38 | .72 | .21 | 3 |
| | Sept. | 30 | — | 8 | 49 | .87 | .15 | 3 |
| | Nov. | 31 | — | 8 | 41 | .66 | .11 | 3 |
| | Jan. | 35 | — | 8 | 41 | .42 | .11 | 3 |
| | Mar. | 28 | — | 8 | 40 | .73 | .32 | 3 |
| Jojoba, California (FR) | May | 46 | — | 11 | 40 | .79 | .20 | 3 |
| | June | 44 | — | 12 | 29 | .72 | .32 | 3 |
| Krameria, littleleaf (LE) | May | 40 | 31 | 14 | 43 | .72 | .46 | 3 |
| Mesquite (LE) Semidesert Grassland | Spring | 62 | — | 18 | — | 2.00 | .23 | 1 |
| | Autumn | 45 | — | 16 | — | .82 | .10 | 1 |
| | Winter | 45 | — | 16 | — | 2.00 | .10 | 1 |
| Mesquite (FR) Interior Chaparral | June | 54 | 51 | 17 | 35 | .77 | .25 | 3 |
| Oak, Arizona white (FR) Interior Chaparral | Sept. | — | — | 4 | 43 | .80 | .43 | 3 |
| Oak, Emory (FR) Interior Chaparral | June | — | 30 | 6 | 58 | .35 | .13 | 3 |
| Oak, Gambel (LE) Ponderosa Pine | May | 58 | — | 24 | 22 | .25 | .57 | 4 |
| | July | 46 | — | 12 | 21 | .56 | .14 | 4 |
| | Sept. | 46 | — | 12 | 31 | .88 | .20 | 4 |
| Oak, gray (CAG) Pinyon-juniper | Spring | 41 | — | 11 | — | 1.03 | .30 | 2 |
| | Summer | 47 | — | 7 | — | .59 | .20 | 2 |
| | Autumn | 39 | — | 9 | — | 1.18 | .26 | 2 |
| | Winter | 51 | — | 9 | 9 | .73 | .24 | 2 |
| Oak, shrub live (LE) Interior Chaparral | May | — | 33 | 13 | 38 | .43 | .32 | 3 |
| | Dec. | — | 41 | 8 | 40 | .77 | .19 | 3 |
| | March | 36 | 34 | 10 | 40 | .66 | .17 | 3 |
| Oak, shrub live (ST) | May | — | 31 | 9 | 37 | .49 | .24 | 3 |
| | Dec. | — | 33 | 4 | 53 | 1.01 | .10 | 3 |
| | March | — | 18 | 5 | 54 | .90 | .11 | 3 |
| Oak, shrub live (FR) | June | 35 | 34 | 6 | 29 | .66 | .16 | 3 |
| | Sept. | — | — | — | — | — | — | — |
| Pricklypear, Engelmann (FR) Semidesert Grassland | Spring | 42 | — | 12 | — | 3.33 | .33 | 1 |
| | Summer | 44 | — | 5 | — | 1.41 | .24 | 1 |
| Silktassel, Wright (CAG) Pinyon-juniper | Spring | 58 | — | 6 | — | 2.60 | .13 | 2 |
| | Summer | 56 | — | 7 | — | .87 | .13 | 2 |
| | Autumn | 53 | — | 6 | — | 1.50 | .09 | 2 |
| | Winter | 55 | — | 5 | — | 1.09 | .12 | 2 |

¹Plant parts: LE=leaves, ST=stems, FR=fruits, WP=whole plant, CAG=current annual growth.

²Digestible dry matter obtained in vitro.

³References: 1, Short (1977); 2, Short et al. (1977); 3, Urness (1973), 4, Urness et al. (1975a).

⁴MD=mule deer, CD=Coues white-tailed deer.

⁵ADF=acid detergent fiber.

⁶—Indicates no data.

Table C8. Some nutrient requirements for beef cattle, sheep, white-tailed deer, mule deer, and elk.

| Species, class and conditions | Body wt. (BW) | Dry matter intake | Digestible energy for maintenance (DE) | Protein | Calcium | Phosphorus | Reference ¹ |
|---|---------------|-------------------|--|------------------|----------------------|----------------------|------------------------|
| | pounds | pounds/day | Kcal DE/BW ^{.75} | percent of diet | | | |
| Beef cattle, steers 1.1 pounds per day weight gain | 441 | 12.8 | 169.2 ² | 9.9 | 0.24 | 0.22 | 6 |
| Beef cattle, cows | | | | | | | |
| Last third of pregnancy | 992 | 16.5 | 184.2 ² | 5.9 | .18 | .18 | 6 |
| Lactation (ave. milk prod.) | 992 | 24.9 | 343.9 ² | 9.2 | .28 | .28 | 6 |
| Beef cattle, bulls | 1,102 | 26.9 | — | 8.8 | .18 | .18 | 6 |
| Sheep, ewes, late gestation | 132 | 4.2 | 204.2 ² | 9.3 | .23 | .22 | 5 |
| Sheep, ewes, early lactation, suckling twins | 132 | 5.1 | 269.1 ² | 11.5 | .50 | .36 | 5 |
| Sheep, lambs, finishing | 88 | 3.5 | — | 11.0 | .31 | .19 | 5 |
| White-tail, males, growth | 50-60 | 2 | — | 13-16 | — | — | 3 |
| | 100 | 3-4 | — | — | — | — | 3 |
| | 150 | 5-6 | — | — | — | — | 3 |
| White-tail, fawn maximum growth | — | — | — | 12.7 ♀ 20.2 ♂ | — | — | 9 |
| White-tail, males, maximum antler & skeletal growth | 190.6 (Oct.) | 6.6 | — | 17 | .64 | .56 | 4 |
| winter maintenance | 145.4 (Mar.) | 2.6 ³ | — | 7-9 | .30 | .30 | 4 |
| White-tail, adult, pregnant does, maint. | — | — | 158 | — | — | — | 10 |
| White-tail, fawns maintenance | — | — | 155-168 | — | — | — | 2 |
| White-tail fawns maximum growth | — | — | — | — | .46-.51 ⁴ | .26 | 12 |
| White-tail fawns normal development | — | — | — | — | .40 | .25-.27 ⁵ | 11 |
| Mixed herd, Coues deer and Rocky Mtn. mule deer maintenance | 100 | 2.5 | — | — | — | — | 7 |
| Rocky Mtn. mule deer fawns winter maintenance | — | — | 157 | — | — | — | 1 |
| Rocky Mountain elk maintenance of confined pregnant animals | 576 | 10.1 | — | — | — | — | 8 |
| Rocky Mountain elk, maintenance, free ranging pregnant animals (est.) | 576 | 13.1 | — | — | — | — | 8 |

¹Reference, Baker (1976); 2 Croyle (1969); 3 French et al. (1956); 4 McEwen et al. (1957); 5 National Research Council (1975); 6 National Research Council (1976); 7 Nichol (1938); 8 Thorne et al. (1976); 9 Ullrey et al. (1976); 10 Ullrey et al. (1970); 11 Ullrey et al. (1973); 12 Ullrey et al. (1975).

²Authors calculation from digestible energy data given by Halls (1970).

³Food consumption decreased and weight losses occurred every winter regardless of diet. See also Moen (1978).

⁴Level of Ca fed in ration to determine P requirement.

⁵Level of P fed in ration to determine Ca requirement.

Table C9. Estimated digestibility and nutritive levels of seasonal diets of Coues white-tailed deer in the Interior Chaparral habitat of the Southwest. All figures percent of dry weight (Authority: Urness and McCulloch (1973)).

| Time of year | Percent of diet analyzed | Digestible dry matter ¹ | Crude protein | ADF ² | Ca | P |
|-------------------|--------------------------|------------------------------------|---------------|------------------|------|------|
| May-June | 82 | 36 | 12 | 35 | 0.88 | 0.28 |
| July-September | 94 | 37 | 9 | 38 | 0.66 | 0.22 |
| October | 88 | 56 | 12 | 39 | 1.23 | 0.21 |
| November-December | 85 | 47 | 10 | 34 | 1.74 | 0.22 |
| January | 89 | 42 | 11 | 36 | 1.24 | 0.23 |
| February-April | 80 | 52 | 13 | 32 | 1.31 | 0.31 |

¹Digestible dry matter obtained in vitro.

²ADF=acid detergent fiber.

Table C10. Estimated digestibility and nutritive levels of seasonal diets of mule deer in the Southwest. All figures percent of dry weight.

| Habitat | Time of year | Percent of diet analyzed | Digestible dry matter ¹ | Crude protein | ADF ² | Ca | P | Reference ³ |
|----------------------|--------------|--------------------------|------------------------------------|---------------|------------------|------|------|------------------------|
| Semidesert Grassland | Spring | 87 | 52 | 10 | — | 1.55 | 0.23 | 2 |
| | Summer | 94 | 48 | 10 | — | 1.09 | 0.16 | 2 |
| | Autumn | 93 | 45 | 10 | — | 1.17 | 0.15 | 2 |
| | Winter | 91 | 54 | 10 | — | 1.33 | 0.15 | 2 |
| Interior Chaparral | May-June | 94 | 43 | 12 | 36 | 0.88 | 0.25 | 3 |
| | July-Sept. | 83 | 42 | 14 | 34 | 0.83 | 0.23 | 3 |
| | Oct. | 98 | 35 | 12 | 38 | 1.21 | 0.26 | 3 |
| | Nov.-Dec. | 92 | 34 | 10 | 40 | 1.56 | 0.16 | 3 |
| | Jan. | 92 | 40 | 11 | 38 | 1.00 | 0.21 | 3 |
| | Feb.-Apr. | 96 | 51 | 13 | 29 | 1.27 | 0.29 | 3 |
| Pinyon-juniper | Spring | 87 | 45 | 11 | 52 | 0.78 | 0.26 | 1 |
| | Summer | 73 | 37 | 9 | 24 | 0.59 | 0.17 | 1 |
| | Autumn | 89 | 43 | 9 | 29 | 0.92 | 0.18 | 1 |
| | Winter | 95 | 23 | 12 | 44 | 1.16 | 0.19 | 1 |
| Ponderosa pine | May | 97 | 68 | 25 | 26 | 0.46 | 0.48 | 4 |
| | June | 92 | 54 | 17 | 29 | 0.72 | 0.24 | 4 |
| | July | 97 | 56 | 16 | 31 | 1.02 | 0.19 | 4 |
| | August | 93 | 46 | 13 | 32 | 0.82 | 0.19 | 4 |
| | September | 94 | 49 | 10 | 32 | 1.04 | 0.24 | 4 |

¹Digestible dry matter obtained in vitro.

²ADF=Acid detergent fiber.

³Reference: 1, Boeker et al. (1972); 2, Short (1977); 3, Urness and McCulloch (1973); 4, Urness et al. (1975b).