HABITAT GUIDELINES FOR MULE DEER
COLORADO PLATEAU SHRUBLAND AND FOREST ECOREGION

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INTRODUCTION

Mule and black-tailed deer (collectively called mule deer, *Odocoileus hemionus*) are icons of the American West. Probably no animal represents the West better in the minds of Americans. Because of their popularity and wide distribution, mule deer are one of the most economically and socially important animals in western North America. A survey of outdoor activities by the U.S. Fish and Wildlife Service in 2001 showed that over 4 million people hunted in the 18 western states. In 2001 alone, those hunters were afield for almost 50 million days and spent over $7 billion. Each hunter spent an average of $1,581 in local communities across the West on lodging, gas, and hunting-related equipment. Because mule deer are closely tied to the history, development, and future of the West, this species has become one of the true barometers of environmental conditions in western North America.

Mule deer are distributed throughout western North America from the coastal islands of Alaska, down the west coast to southern Baja Mexico and from the northern border of the Mexican state of Zacatecas, up through the Great Plains to the Canadian provinces of Saskatchewan, Alberta, British Columbia, and the southern Yukon Territory. Within this wide geographic range, a diversity of climatic regimes and vegetation associations has resulted in dynamic relationships between mule deer and their habitats that can vary greatly from one part of their range to another.

To better address this variability, the overall distribution of mule deer can be divided into "ecoregions" with similar management issues and challenges. In these guidelines we have designated 7 separate ecoregions: 1) California Woodland Chaparral, 2) Colorado Plateau Shrubland and Forest, 3) Coastal Rain Forest, 4) Great Plains, 5) Intermountain West, 6) Northern Forest, and 7) Southwest Deserts.

The diversity among the ecoregions presents different challenges to deer managers and guidelines for managing habitat must address these differences (deVos et al. 2003). In many ecoregions, water availability is not a major limiting habitat factor. However, in others, such as the Southwest Deserts ecoregion, water can be important. A significant factor affecting deer population fluctuations in the Northern Forest is severe winterkill. Winterkill is not a problem in the Southwest Deserts, but overgrazing and drought can seriously impact populations.

The shrubs that deer heavily rely on in the Intermountain West are disappearing from the landscape, partially because invasions of exotic plants like cheatgrass (*Bromus tectorum*) have increased frequency of fire and resulted in a more open landscape. In contrast, California Woodland Chaparral and many forested areas are lacking the natural fire regime that once opened the canopy and provided for growth of important deer browse plants. Yet, an intact forest canopy is important in some northern areas of coastal rainforests to intercept the copious snow that falls in that region and impacts black-tailed deer survival.

Across these different ecoregions, the core components of deer habitat are consistent: water, food, and cover. An important aspect of good mule deer habitat is juxtaposition of these components; they must be interspersed in such a way that a population can derive necessary nutrition and cover to survive and reproduce. We have learned much about mule deer habitat requirements, but much remains to be learned. For example, we know cover can be critical for mule deer survival but quantitative cover requirements and the optimal balance between cover and food resources in highly variable environments are mostly unknown.

Mule deer are primarily browsers, with a majority of their diet comprised of forbs (broad-leaved, non-woody plants) and browse (leaves and twigs of shrubs and trees). Deer digestive tracts differ from cattle and elk in that they have a smaller rumen in relation to their body size and so they must be more selective in their feeding. Instead of eating large quantities of low quality feed like grass, deer must select the most nutritious plants and parts of plants. Because of this, deer have more specific forage requirements than larger ruminants.

The presence and condition of the shrub component is an underlying issue found throughout different ecoregions and is important to many factors affecting mule deer populations. Shrubs occur mostly in early successional habitats; that is, those recently disturbed and going through the natural processes of maturing to a climax state. This means disturbance is a key element to maintaining high quality deer habitat. In the past, different fire cycles and human disturbance, such as logging, resulted in higher deer abundance that we see today. Although weather patterns, especially precipitation, drive deer populations in the short-term, only landscape-scale habitat improvement will make long-term gains in mule deer abundance in many areas.

Mule deer are known as "K-selected" species. This means that populations will increase until biological carrying capacity is reached. If deer populations remain at or beyond carrying capacity they begin to impact their habitats in a negative manner. The manager must be aware that factors such as drought and successional changes can substantially lower carrying capacity for deer for long periods.

Because of the vast blocks of public land in the West, habitat management throughout most of the geographic range of mule deer is primarily the responsibility of federal land management agencies. Mule deer habitats are facing unprecedented threats from a wide variety of human-related developments. If mule deer habitats are to be conserved, it is imperative that state and federal agencies and private conservation organizations are aware of key habitat needs and participate fully in habitat management for mule deer. Decades of habitat protection and enhancement under the nomer of "game" management benefited countless other unhunted species. A shift away from single-species management toward an ecosystem approach to management of landscapes has been positive overall; however, some economically and socially important species are now de-emphasized or neglected in land use decisions. Mule deer have been the central pillar of the American conservation paradigm in most western states and are directly responsible for supporting a wide variety of conservation activities that Americans value.

Habitat conservation will mean active habitat manipulation or conscious management of other land uses. Treated areas must be sufficiently large to produce a "treatment" effect. There is no one "cookbook" rule for scale of treatment. However, the manager should realize the effect of the treatment applied properly is larger than the actual number of acres treated because the value of untreated habitat in the vicinity of treatments can also increase. In general, a number of smaller treatments in a mosaic or patchy pattern are more beneficial than 1 large treatment in the center of the habitat. Determining the appropriate scale for a proposed treatment should be a primary concern of the manager. Treatments to improve deer habitat should be planned to work as parts of an overall strategy. For example, treatments should begin in an area where benefit will be greatest and then subsequent habitat improvement activities can be linked to this core area.

The well-being of mule deer now and in the future rests with condition of their habitats. Habitat requirements of mule deer must be incorporated into land management plans so improvements to mule deer habitat can be made on a landscape scale as the rule rather than the exception. The North American Mule Deer Conservation Plan (NAMDCP) provides a broad framework for managing mule deer and their habitat. These habitat management guidelines tier from that plan and provide specific actions for its implementation. The photographs and guidelines herein are intended to communicate important components of mule deer habitats across the range of the species and suggest management strategies. This will enable public and private land managers to execute appropriate and effective decisions to maintain and enhance mule deer habitat.

THE COLORADO PLATEAU SHRUBLANDS & FOREST ECOREGION

INTRODUCTION

The Colorado Plateau Shrubland and Forest Ecoregion (CPE) is known for its open spaces, diverse topography, and sparse human population. Some people would refer to this region as "waste land". However, for those interested in wildlife, the CPE is known for its productive habitats that support some of the largest mule deer populations in North America. The mule deer has factored prominently in the human history of the CPE (Fig. 1).

Since European settlement, domestic livestock grazing has been the primary historic use of the CPE. Herbivory on plants has been a constant factor. It is an area of the country that is typically dry during much of the year and, in some cases, very susceptible to periods of extended droughts. However, it is also an ecoregion that can experience significant snowfalls in winter, which can impose serious limiting factors for mule deer populations.

The CPE is very susceptible to the general phenomenon of "drying of the landscape" which greatly influences vegetative succession. Recent concerns about climate change and its impact on habitats at the landscape scale complicate this climatic picture considerably.

Historically, wildlife mangers often assumed the wide-open spaces, variable topography, and vegetation of the CPE would always provide necessary habitats for mule deer. However, successional changes, invasive weeds, and rapid development have dealt land managers a new deck of cards. How these changes will impact habitats for wild animals in the future must be better understood by land managers. Much of the ecoregion's habitats lie above vast reservoirs of natural gas, oil, oil shale, and coal. A compounding factor accompanying development of energy resources will be an increased need and demand for the already short supply of water. Another major change is that human impacts are increasing rapidly from residents and seasonal visitors who recreate in the area because of the natural beauty.

As demonstrated in these habitat guidelines, traditional assumptions that have guided land and wildlife managers in the past (i.e., little active management or essentially hands-off management) are proving to be unfounded. Anthropogenic landscape changes are quickly becoming pervasive in the CPE and managers must address these in the very near future.

In short, humans have discovered this portion of North America offers many values beyond wide open spaces. Wildlife managers of tomorrow will need to effectively compete with many other resource demands if mule deer and other wild creatures are to have habitat where they can exist in something more than remnant populations. Hopefully, these guidelines will be useful as managers face these challenges.

Figure 1. Petroglyph from Newspaper Rock, Utah.
(Photo by Bruce Watkins/Colorado Division of Wildlife [CDOW]).

Figure 2. The Colorado Plateau Shrubland and Forest Ecoregion.
(Sue Boe/Arizona Game and Fish Department [AGFD]).

Description

The CPE is a rugged area of high mountains, mesas, deep valleys and canyons, shrublands, and deserts in western Colorado, northwestern New Mexico, northeastern Arizona, eastern Utah, and the southwestern corner of Wyoming (Fig. 2). Elevations range from 2,000 feet to > 14,000 feet, resulting in a diverse array of habitat types along altitudinal gradients. Most of the CPE exceeds 5,000 feet in elevation. Annual precipitation varies from < 8 inches in some low elevation deserts to > 40 inches in the higher mountains. Snow is the primary form of precipitation in most mountainous areas, but summer monsoons from July to September can also provide substantial moisture. Important habitats for deer in the CPE include sagebrush (*Artemisia* spp.)-steppe, pinyon pine (*Pinus edulis*) and juniper (*Juniperus* spp.) woodlands, mountain shrub communities, aspen (*Populus tremuloides*) forests, and montane and subalpine coniferous forests. The CPE includes some of the largest and most productive mule deer herds in North America.

Ecoregion-specific deer ecology

Most deer in the CPE make seasonal movements, sometimes exceeding 50 miles, between higher elevation summer ranges and lower elevation winter ranges. This behavior allows deer to avoid deep snow during winter but still take advantage of higher quality montane, subalpine, and alpine forage during summer. Exceptions to this pattern sometimes occur in southern parts of the CPE as well as in riparian corridors, irrigated valleys, and residential areas where some deer remain resident year-round.
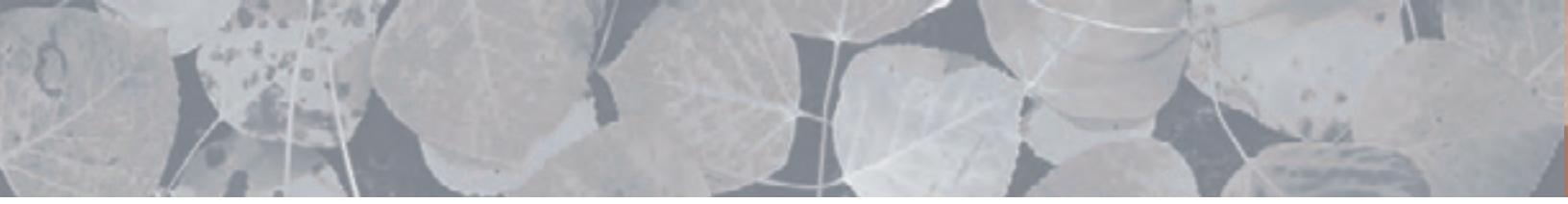
Winter range is often considered the most limiting habitat type for deer in the northern portion of the CPE (Wallmo et al. 1977). Important winter range habitats include sagebrush-steppe, pinyon-juniper woodlands, mountain shrub, and ponderosa pine (*Pinus ponderosa*) forests below 7,500 feet. Many deer also winter in and around irrigated agricultural areas. Winter diets are often a diverse combination of various forbs, browses, and new growth on cool-season grasses (Wallmo and Regelin 1981). Browse becomes an increasing portion of the diet as snow accumulates or forbs and grasses become depleted. Winter diets are typically sub-maintenance, but higher quality diets can substantially reduce rate of weight loss. In Colorado, Utah, Wyoming, and northern New Mexico and Arizona, there is often less suitable winter range than summer range, resulting in deer becoming concentrated on winter ranges with densities of 20-100 deer/mile2 typical in suitable habitat. Major deer mortality can result from severe winters in higher wintering areas and is exacerbated as deer densities increase. Winter range areas are often the focus of

attention for deer managers in the CPE because these areas 1) must support higher densities of deer on less available forage, 2) are often less tolerant of high herbivory rates, 3) are more prone to non-native weed invasion, and 4) are more likely to be developed for energy and mineral extraction or for residential subdivisions. Most summer range areas in the CPE occur on U.S. Forest Service (USFS) lands, whereas a high proportion of winter range is privately owned.

In the southern portion of the CPE, relative importance of focusing management on winter range versus summer range becomes less apparent. This shift in management priorities is particularly true in those areas where summer habitats are primarily mid-elevation, ponderosa pine forests; winters are seldom severe; and deer are less concentrated on winter ranges. Unlike the northern part of the CPE, where very productive summer and fall habitats can often support high fat accretion, deer in the southern part of the CPE are more likely to be influenced by drought on summer and fall ranges and enter winter in suboptimal condition. Highly productive aspen forests are an important summer habitat component for deer in the northern portion of the CPE, but are typically limited in distribution or absent farther south. In addition, mountain shrub communities used as high quality, transitional ranges by deer are much more extensive in the northern portion of the CPE. Summer diets are typically comprised of a mixture of succulent forbs, deciduous browses and leaves, and growing grasses.

During all seasons, deer in the CPE show a strong preference for habitats that provide a mosaic of plant species, age classes, and successional stages in close juxtaposition to cover. Areas with good shrub and forb components, high landscape heterogeneity, and a high ratio of edge between cover and openings typically support the highest deer densities. Areas that have low plant species diversity or few shrubs, are dominated by grass or dense forests, or lack appropriate juxtaposition between cover and foraging areas have few deer.

Deer numbers in the CPE frequently show some fluctuation based on winter severity, harvest management, local habitat changes, and other factors. Occasionally, CPE deer populations have undergone widespread declines that do not appear to be directly associated with harvest management or unusual winter mortality. Such declines occurred in the early 1970s and in the mid-1990s in many areas (Gill et al. 2001). Habitat changes, predation, and disease have each had their proponents as potential causative factors. Although debate continues in some circles, most wildlife managers consider habitat quality and density-dependent effects to be underlying causes for widespread population declines in the CPE.

Studies of deer survival rates and post-hunt fawn:doe ratios from western Colorado have indicated that declining deer numbers in the 1990s were primarily a result of low fawn survival prior to 6 months of age (Pojar and Bowden 2004; Bishop et al. 2005; Colorado Division of Wildlife [CDOW], unpublished data). Malnutrition and disease appeared to be at least as important as predation in causing early fawn mortality. Further studies have shown that improved nutritional status of pregnant does on winter range can increase early survival of their fawns the following summer and improved nutritional status of fawns during winter can significantly increase their over-winter survival rates even when predation is the primary cause of mortality (Bishop et al. 2005). These studies indicate fawn predation rates should be viewed as a direct function of habitat quality and deer condition with relatively small changes in survival rates capable of causing significant population change.

In addition to an abundance of mule deer, the CPE is also home to the largest elk (*Cervus elaphus*) herds in North America. Deer and elk are sympatric throughout much of the CPE with the exception of the northeast corner of Arizona. Elk numbers have increased considerably since the 1980s in many parts of the CPE. With their larger body size and broader foraging capabilities, elk can easily out-compete deer when vying for limited resources. However, competitive relationships between deer and elk are likely very dynamic and population effects of elk on deer in the CPE are mostly speculative (Lindzey et al. 1997, Keegan and Wakeling 2003). Elk appear to be much more adaptable to changing range conditions than deer and will more readily shift their distribution from year to year depending on snow depth, hunting pressure, burns and habitat treatments, and other factors. Management of deer habitat in the CPE should always take into account potential impacts from elk. Whenever possible, attempts should be made to segregate elk from deer on winter ranges by actively managing winter elk habitat at elevations higher than traditional deer winter ranges.

Ranching continues to be a major economic cornerstone in the CPE even though increasing numbers of ranches are being converted into residential areas and ranchettes. Public land grazing, primarily by cattle, occurs in most areas with USFS and BLM land. Private ranch lands often include some of the highest quality deer wintering areas especially in valley bottoms. Livestock and deer commonly share the same ranges much of the time.

Chronic wasting disease (CWD) has been documented in deer in the CPE in all states except Arizona. To date, population effects from CWD have not been apparent. Survival and recruitment rates in deer herds with a low incidence of CWD are indistinguishable from those without

CWD (CDOW unpublished data). Habitat management that encourages wider distribution and lower densities of deer is the only alternative currently available to address CWD from a habitat perspective.

Land ownership in the CPE is a patchwork of federal, state, and private lands. Deer habitat management in the CPE should be based on an ecosystem approach rather than on the basis of administrative and ownership boundaries. The future of effective habitat management in the CPE lies in successful collaborative efforts among federal and state agencies, conservation and sportsmen's organizations, private landowners, and local communities.

1.Vegetative species composition has been modified. Invasive, non-native plants with little or no forage value for deer are increasing across the CPE. The greatest impacts have occurred on deer winter range areas with low precipitation and are associated with the invasion of cheatgrass. Not only can cheatgrass out-compete most native plants when moisture is limited, it can also change site-specific fire ecology and result in the loss of critical shrub communities. In addition to the effect of invasive species, plant species composition in the CPE has changed as a result of successional changes (often facilitated by fire suppression), excessive foraging from livestock and wildlife, disturbance, and intentional conversion.

Vegetation structure has been modified. Active fire suppression has changed the vegetation structure in many areas of the CPE by allowing an increasing proportion of forested areas to mature and accumulate unnaturally high fuel loads that will eventually result in catastrophic fires if not actively managed. In winter range areas, expansion and maturation of pinyon-juniper woodlands in the absence of fire has decreased understory diversity and productivity resulting in less winter forage for deer. In ponderosa pine forests, suppression of regular, natural understory fires has increased ladder fuels, increased crown fire potential, and reduced understory productivity. At higher elevations, absence of fire or other disturbance has resulted in some aspen forests being replaced by spruce (*Picea* spp.)-fir (*Abies* spp.) forests with much lower value for deer.

Nutritional quality has decreased. In addition to changes in plant species composition that favor less palatable and often non-native species, nutritional quality of deer habitat can also decline as preferred plant species mature and older growth accumulates. As plants mature, cell walls thicken, anti-herbivory defenses become more developed, and the relative amount of nutritious, current annual growth decreases. Periodic disturbance is often necessary to stimulate plant productivity. Disturbance can be achieved through controlled grazing, fire, or chemical or mechanical means.

Loss and fragmentation of usable habitat due to human encroachment and associated activities. The human population of the CPE is increasing rapidly as many people move to the area because of the natural beauty, desirable climate, and recreational opportunities. High land prices make subdividing ranches an appealing alternative for many landowners. More people results in more roads, infrastructure, and fragmentation that compounds habitat loss. In addition to residential development on private lands, large reserves of oil, oil shale, and natural gas occur in the CPE, resulting in extensive development for energy extraction on public and private lands. Lower elevation winter range areas are being most impacted by

development, but higher elevation developments, often associated with ski areas, are also increasing and can be particularly detrimental if they occur in aspen forests or migration corridors. In addition to development impacts, an ever increasing number of people are recreating on public lands in the CPE and use of motorized transportation in the backcountry is becoming more popular every year.

EXCESSIVE HERBIVORY

BACKGROUND

History of Livestock Grazing

Livestock were introduced to the CPE by Spanish explorers and missionaries during the 1500s and 1600s and gradually proliferated with increased settlement (Wildeman and Brock 2000, Holechek et al. 2001). Completion of railroads in the Southwest in the late 1800s sparked a boom in the livestock industry which, in combination with drought, led to severe degradation of western rangelands (Wildeman and Brock 2000, Holechek et al. 2001).

Public lands grazing came under government control in the early 1900s as a way to limit range deterioration and resolve range conflicts. The Taylor Grazing Act was passed in 1934, in part due to concerns expressed by ranchers in the CPE. This Act led to the eventual establishment of the Bureau of Land Management (BLM) and continues to be a source of many present-day issues surrounding public lands grazing (Klyza 1996, Holechek et al. 2001).

During the 1940s – 1960s, the range management profession began to flourish, thereby allowing scientific approaches toward livestock management on public lands. Multiple-use legislation and environmental policy during the 1960s and 1970s diversified the missions and management approaches of the USFS and BLM, leading to an overall decline in livestock grazing on public lands (Klyza 1996, Holechek et al. 2001). At present, livestock grazing remains a principal use of public lands across the CPE. Rangeland condition has improved in recent decades, yet many challenges render management difficult (Holechek et al. 2001). Land managers must manage livestock grazing in the context of noxious weed invasions, human-altered disturbance regimes, water quality issues, energy development, and increased recreational demands, to name a few. Management for healthy ecosystems that support native flora and fauna is often neglected or of secondary importance. Thus, livestock grazing on public lands in the CPE remains a controversial issue.

History of Mule Deer Populations

Mule deer numbers in the CPE prior to the 1900s are difficult to assess because reports are purely anecdotal. Diaries and reports of early explorers and settlers suggest mule deer were widespread and commonly encountered in the CPE during the 1800s (Denney 1976, Carmony and Brown 2001, Gill et al. 2001). Deer populations were reduced to very low levels during the late 1800s and early 1900s because of unregulated hunting (Julander 1962, Robinette et al. 1977, Brown and Carmony 1995, Gill et al. 2001). Mule deer populations began recovering by the 1930s following the advent of game laws and perhaps changing habitat conditions and intensive predator control. One common theory suggests that excessive livestock grazing in combination with drought conditions favored a conversion of many grasslands to shrub-dominated habitats, favoring an increase in mule deer (Julander 1962, Julander and Low 1976, Urness 1976, Robinette et al. 1977, Austin and Urness 1998). Liberal predator control practices during the early-mid 1900s and restrictive deer harvest may have facilitated a rapid increase in deer once habitat conditions became amenable. Whatever the mechanism, mule deer numbers increased dramatically during the 1940s, reaching historic highs (Julander 1962, Workman and Low 1976, Gill et al. 2001). Populations remained high overall through the 1950s and early 1960s and biologists became concerned that deer numbers had greatly exceeded carrying capacity of the range. At this time, big game managers attempted to reduce populations through liberal buck and doe harvests (Denney 1976, Robinette et al. 1977, Kufeld 1979, Connolly 1981). Subsequently, widespread declines in deer numbers were observed during the late 1960s and early 1970s (Workman and Low 1976), although sound documentation of the apparent declines was lacking (Gill 1976). Deer populations generally rebounded during the 1980s but experienced another decline during the 1990s (Unsworth et al. 1999, Gill et al. 2001, Heffelfinger and Messmer 2003), which was more adequately documented with population data.

Long-term studies of several mule deer populations in the CPE are useful for comparison to the general description of population dynamics provided above. The Kaibab deer herd in northern Arizona is one of the most infamous examples of a deer population irruption and subsequent decline (Rasmussen 1941). Although historic population estimates for the Kaibab deer herd lack any scientific merit and cannot be substantiated (Russo 1964, Caughley 1970, Burk 1973), the Kaibab provides a case history of how grazing, drought, game management, and predator control interacted to define the general picture of deer populations in the CPE during the first half of the 20th century (Heffelfinger 2006). Trends of several other deer populations in the CPE support the general notion that deer populations peaked sometime during the late 1940s through early 1960s before experiencing declines, including the Uncompahgre Plateau herd in western Colorado, the Oak Creek and LaSal-Henry Mountain herds in Utah, and the Three Bar Wildlife Area herd in Arizona (Julander and Low 1976, Robinette et al. 1977, Kufeld 1979). Recent data support the contention that at least some populations rebounded during the late 1970s and 1980s before declining yet again during the 1990s. For example, the Uncompahgre Plateau population reached an estimated high of 60,000 deer in the early 1980s and then experienced a steady decline during the 1990s, reaching a low of 28,000 deer in 1999 (B. E. Watkins, CDOW, unpublished data). In summary, available management reports and population data associated with specific herds in the CPE support the broad characterization of deer population fluctuations during the 20th century.

History of Potential Native Competitors

Elk were common across much of the CPE prior to Spanish exploration and during early settlement, particularly in Colorado. However, elk were extirpated from Arizona, New Mexico, and Utah during the late 1800s and early 1900s because of uncontrolled hunting, and elk were nearly extirpated from Colorado, where only 500-1,000 elk were thought to remain by 1910 (Brown and Carmony 1995, O'Gara and Dundas 2002). Elk were reestablished throughout their historic range in the CPE through a series of translocations, primarily from the Yellowstone area during the early-mid 1900s. Elk are presently distributed throughout western Colorado and New Mexico, central and east-central Arizona, and eastern, central, and north-central Utah (Brown 1994, O'Gara and Dundas 2002).

Bighorn sheep (*Ovis canadensis*) are thought to have occupied most suitable mountain or canyon habitats in the CPE prior to settlement. Colorado and northern Utah populations were comprised predominantly of Rocky Mountain bighorn sheep (*O. c. canadensis*) whereas southern Utah, Arizona, and New Mexico populations were primarily desert bighorn sheep (*O. c. nelsoni*). Many bighorn sheep populations were extirpated during the late 1800s and early 1900s because of market hunting and disease transmitted from domestic sheep (Ellenberger 1999, Fisher 1999, Karpowitz 1999, Lee 1999, Shields 1999). Reintroductions of bighorn sheep have been fundamental in establishing present-day populations, which encompass much of the historical range. Even so, habitat loss, successional changes, and disease have kept many populations at low levels.

Historic pronghorn (*Antilocapra americana*) occupancy in the CPE was primarily restricted to central and east-central Arizona and western New Mexico, although several other pronghorn populations were scattered across the remainder of the CPE (Yoakum 2004a). Similar to other ungulates, pronghorn had been reduced to low numbers by the early 1900s due to excessive harvest (Brown and Carmony 1995). Current pronghorn occupancy largely mimics historic distribution, in part because translocations to non-native ranges were largely unsuccessful. As such, pronghorn in the CPE are most common in central and east-central Arizona, with a few scattered populations throughout New Mexico, western Colorado and eastern Utah (Yoakum 2004a). White-tailed deer (*Odocoileus virginianus*) were virtually non-existent in the CPE historically, and remain largely absent to the present. Coues white-tailed deer (*O. v. couesi*), which primarily inhabits the Southwest Deserts Ecoregion,

extends into the southern edge of the CPE in Arizona and New Mexico (Knipe 1977). White-tailed deer are also present immediately east and north of the CPE (Baker 1984).

Seasonal Ranges

The CPE encompasses 5 general winter range habitat complexes used by mule deer: 1) high-elevation (7,200-9,000 feet) big sagebrush (*Artemisia tridentata*) parks in central Colorado on the eastern edge of the CPE (Fig. 3); 2) mid-elevation (5,000-7,500 feet) pinyon and juniper woodlands, often with interspersed sagebrush, across the entire CPE (Fig. 4); 3) low to mid-elevation (4,600-6,900 feet) big sagebrush range scattered throughout Utah,

Figure 3. A mule deer group foraging in typical high-elevation (7,200 – 9,000 feet) big sagebrush winter range in Middle Park, Colorado, on the eastern edge of the CPE. (Photo by David J. Freddy/CDOW).

Figure 4. Mule deer foraging on the edge of pinyon-juniper and sagebrush-grass habitat. Pinyon-juniper with interspersed sagebrush is the dominant winter range habitat type across the CPE. (Photo by Chad J. Bishop/CDOW).

western Colorado, northern New Mexico, and northern Arizona; 4) low to mid-elevation salt-desert shrub in Utah and western Colorado and semi-desert shrub-steppe in Arizona and New Mexico; and 5) low to mid-elevation grasslands in New Mexico and Arizona (Russo 1964, Plummer et al. 1968, Gilbert et al. 1970, Robinette et al. 1977, Wallmo et al. 1977, Kufeld 1979, Garrott et al. 1987, Carrel et al. 1999, Bender 2003, SWReGAP 2004). The vast majority of deer occupy pinyon-juniper and big sagebrush winter ranges. During mild winters, deer distribution extends into the transitional range at higher elevations.

Transitional range comprises several habitat types that are typically located directly above the pinyon-juniper zone. Dominant transitional habitat complexes include Gambel oak (*Quercus gambelii*), mountain shrub, and ponderosa pine (Russo 1964, Plummer et al. 1968, Robinette et al. 1977, Wallmo et al. 1977, Kufeld 1979, Garrott et al. 1987, SWReGAP 2004). Common species in the mountain shrub complex include Gambel oak, chokecherry (*Prunus virginiana*), serviceberry (*Amelanchier* spp.), big sagebrush, true mountain mahogany (*Cercocarpus montanus*), antelope bitterbrush (*Purshia tridentata*), cliffrose (*Cowania* spp.), and snowberry (*Symphoricarpos* spp.). The transition zone is relatively narrow across much of the CPE. Thus, most deer are capable of crossing the transition zone in less than a day. The length of time actually spent by deer in transitional range is dependent on weather-habitat interactions that influence forage quality and availability, among other factors.

The CPE encompasses 6 dominant summer range habitat complexes used by mule deer, listed in order of increasing elevation: 1) Gambel oak and mountain shrub (Fig. 5), 2) ponderosa pine, 3) montane sagebrush-steppe, 4) aspen (Fig. 6), 5) spruce-fir, mixed conifer, and 6) montane parks and meadows (Russo 1964, Plummer et al. 1968, Robinette et al. 1977, Kufeld 1979, Garrott et al. 1987, Carrel et al. 1999, Bender 2003, SWReGAP 2004, Bishop et al. 2005). Summer range elevations range from primarily from 6,500-11,500 feet. The spruce-fir, mixed-conifer complex includes Engelmann spruce (*Picea engelmannii*), blue spruce (*Picea pungens*), subalpine fir (*Abies lasiocarpa*), Douglas-fir (*Pseudotsuga menziesii*), and white fir (*Abies concolor*); relative proportions of the different species vary across the CPE. Diverse habitat mosaics are common at

interfaces between habitat complexes. Generally speaking, each of the 6 summer range complexes are commonly found across Colorado and Utah whereas ponderosa pine and spruce-fir are the primary habitats in Arizona and New Mexico. Pinyon-juniper woodlands also serve as summer range for relatively small numbers of deer throughout the CPE.

Most deer in the CPE are migratory, with normal straight-line migration distances ranging from a few miles to >43 miles (Carrell et al. 1999; C. J. Bishop, CDOW, unpublished data). Non-migratory deer are much less common and are

Figure 5. An oakbrush-aspen interface on mule deer summer range on the Uncompahgre Plateau, Colorado. (Photo by Chad J.Bishop/CDOW).

Figure 6. An aspen stand with herbaceous understory in late spring. Aspen is an important, productive summer range habitat for mule deer across the northern half of the CPE. (Photo by Chad J. Bishop/CDOW).

often associated with small, resident deer populations located in irrigated, agricultural valleys.

Forage Quality and Deer Nutrition
Forage quality and availability are critical factors when evaluating habitat quality for mule deer. Cover is seldom considered limiting in most parts of the CPE except in some habitats such as the pine-grass vegetation type in the southern part of the ecoregion. Forage quality is controlled by digestibility, protein content, mineral content, and plant defenses. Digestibility relates, in part, to the ratio of plant cell wall:cell contents. Cell contents comprise readily-digestible materials whereas cell wall contains less-digestible cellulose and hemi-cellulose and non-digestible lignin (Short 1981, Van Soest 1982, Robbins 1983). A diversity of wild and domestic ruminant species have evolved different physiological and morphological adaptations for selecting and processing forage resources (Demment and Van Soest 1985). Mule deer select forage high in plant cell contents, which is often high in lignin as well (e.g., current annual growth of shrubs). Deer optimize nutrient intake through repeated, selective feeding bouts with short rumination periods and high rates of passage (Fig. 7, Hanley 1982, Hobbs et al. 1983, Hofmann 1989). Passage rates decline, however, with increases in lignin consumption (Spalinger et al. 1986). Deer compensate by increasing gut fill as the proportion of browse in the diet increases, until total gut capacity is reached (Baker and Hobbs 1987). This foraging approach allows deer to capitalize on cellular contents while effectively handling lignified cell wall. In comparison, domestic cattle and sheep are classified as "roughage eaters," consuming larger amounts of forage high in cellulose (e.g., grass). Cattle and sheep optimize nutrient intake from cellulose through fewer, longer feeding bouts and longer fermentation periods, which is facilitated by a proportionately large rumen. Elk are intermediate between deer and cattle/sheep (Hanley 1982, Baker and Hobbs 1987, Hofmann 1989), demonstrating selectivity for higher quality forage but in an opportunistic manner. Elk are better adapted to graminoid diets than deer yet are capable of consuming relatively large amounts of browse. Elk should be capable of achieving maintenance nutrient requirements across a greater spectrum of habitat conditions than deer as long as adequate forage quantities are available.

Diets with 50% digestible energy and 5-7% crude protein should allow adult deer to meet maintenance requirements (Einarson 1946, Dietz 1965, Murphy and Coates 1966, Ammann et. al. 1973, Holter et al. 1979). Winter diets typically result in negative energy balance and deer must rely on tissue catabolism to provide energy and protein to meet metabolic demands. Overwinter survival is therefore often determined by the duration and severity of winter, which affects the rate at which fat and lean body stores

are utilized (Wallmo et al. 1977, Torbit et al. 1985). This is especially true for fawns because their nutrient and energy reserves are more limited than adult deer. Diets containing 16–17% crude protein are thought to meet the maximum needs of growing fawns and lactating does (Verme and Ullrey 1972). During spring, summer, and fall, diet quality often exceeds deer maintenance requirements, thereby facilitating production and growth. This basic picture reveals several key concepts: 1) forage quality during spring and summer determines how well adult does meet late-gestation and lactation nutrient requirements; 2) forage quality during summer and fall determines how well deer accumulate fat stores and lean body mass for the ensuing winter; and 3) forage quality during winter in part determines the rate at which body stores are depleted. Winters on lower elevation winter ranges tend to be relatively mild across the CPE, which lowers daily energetic requirements. However, winter range forage quality tends to be marginal in the CPE, primarily because of limited precipitation.

As described above, pinyon-juniper woodlands and big sagebrush are the dominant winter range habitat complexes. Big sagebrush and juniper have high digestibility relative to most other winter range browse, but they contain volatile oils (i.e., secondary compounds, monoterpenoids) that inhibit digestion to varying degrees. During winter, in vitro dry matter digestibility (IVDMD) ranges from 45 to 65% for big sagebrush (Ward 1971, Welch and Pederson 1981, Welch 1989) and 40 to 48% for Utah juniper (*Juniperus osteosperma*, Bunderson et al. 1986, Welch 1989). Little is known about the quality of pinyon pine for deer, other than that it contains volatile oils and is generally avoided when other forage options are

Figure 7. Mule deer select plant parts high in cell solubles and rely on short fermentation periods with high rates of passage to quickly pass the non-digestible materials consumed. During the 1970s and 1980s, researchers conducted foraging trials with tame mule deer to study foraging strategies and diet selection. (Photo by R. Bruce Gill/CDOW, retired).

available (Wood et al. 1995, Sandoval et al. 2005). Volatile oil content varies within and among accessions, subspecies, and species of sagebrush and juniper (Schwartz et al. 1980a, Welch and McArthur 1981, Welch and Pederson 1981, Behan and Welch 1985). Many studies have focused on evaluating effects of volatile oil concentrations on ruminant digestion and forage selection, and understanding how much of the diet may be comprised of forage items containing these compounds (e.g., Nagy et al. 1964, Jobman 1972, Carpenter et al. 1979, Schwartz et al. 1980a,b, Welch and Pederson 1981, Cluff et al. 1982, Welch et al. 1983, Behan and Welch 1985, Personius et al. 1987, Welch 1997). While many questions remain, scientists generally agree that species, such as big sagebrush, containing volatile oils are often valuable winter browse, and can be consumed in relatively large quantities by deer as long as they are supplemented with other forage items (Fig. 8). In some cases, winter deer diets included >50% sagebrush, juniper, and/or pinyon pine in aggregate without obvious detrimental effects (Trout and Thiessen 1973, Bartmann 1983, Wambolt 1996). However, some researchers observed declines in diet quality when sagebrush exceeded 30%, or juniper exceeded roughly 20%, of the diet (Nagy et al. 1964, Carpenter et al. 1979, Schwartz et al. 1980a, b, Wallmo and Regelin 1981). Regardless of optimal thresholds, other browse species, grasses, and forbs are critical components of winter diets (Carpenter et al. 1979, Wallmo and Regelin 1981). The diversity and availability of these other species is often limited in CPE pinyon-juniper and sagebrush winter range, thereby reducing overall habitat quality. Efforts to increase diversity and availability of plant species via habitat manipulations in pinyon-juniper and sagebrush have been justified on these grounds.

Summer and transition range across the northern half of the CPE typically comprises productive habitats with high forage diversity that should allow deer to achieve optimal dietary intake under most circumstances (Wallmo et al. 1977, Wallmo and Regelin 1981). Therefore, winter range habitat management remains the higher priority in this region. Summer range in the southern half of the CPE is dominated by ponderosa pine, which typically has lower forage diversity and availability than other summer range habitats. Reduced forage diversity and availability could lead to composite deer diets of lower quality than diets farther north, particularly in a multi-grazer environment (Bender 2003, Sandoval et al. 2005). Winter and summer range habitat management are of relatively equal priority across much of New Mexico and Arizona.

Other Habitat Attributes: Cover, Space, and Water
Although nutrition is usually the priority management consideration for deer habitat in the CPE, hiding cover and winter thermal cover can also be important. Hiding cover

allows deer to avoid predators and lessens the impact of human disturbance. Reductions in deer hiding cover have resulted from moderate to heavy cattle grazing (Loft et al. 1987). However, grazing intensities that substantially reduce structural cover are likely to have equally or more detrimental effects on forage quality and availability. Heavy grazing may be most detrimental from a cover standpoint in summer habitats where understory vegetation is used as neonatal hiding cover. The need for thermal cover may be important in more extreme environments such as on sagebrush winter ranges at higher elevations. However, Freddy (1985) found that presence or absence of thermal cover influenced deer behavior but had no effect on deer body mass, suggesting effects on survival were probably minimal.

Spatial considerations for deer become increasingly important as the amount and size of quality habitat patches decrease (Geist 1981). However, nutritional carrying capacity is typically of greater importance than social tolerance and territoriality. Deer densities in most CPE winter range habitats are lower than what deer social tolerance would allow. The possible exception is irrigated agricultural habitats where small parcels of ground support large numbers of deer. Regardless of mechanism, reductions in space, particularly in multi-grazer systems, lead to reductions in carrying capacity that can only be partially compensated by improving quality of remaining habitat. Habitat loss represents the single largest threat to mule deer in the CPE.

Although water might be limiting for deer in some parts of the CPE, particularly during drought years, the primary concern is how free water sources affect distribution of deer and other grazing ungulates across the landscape (see Water Availability). Free-water sources are more likely to influence the distribution of livestock, particularly cattle, than native ungulates. Water developments can potentially increase competition between deer and livestock by allowing livestock to use new areas already occupied by deer. Conversely, developing water sources to intentionally redistribute livestock could be beneficial in some cases (e.g., protect riparian habitat through upland water developments; Kie et al. 1994, Cartron et al. 2000).

Potential Benefits of Multi-Species Grazing Systems
Livestock grazing can potentially have positive effects on rangeland habitat quality when stocking levels are appropriate and implemented as part of a detailed management plan. Carefully-managed grazing can be used to improve forage productivity and quality and alter community composition to achieve specific habitat management objectives (Paige 1992, Holechek et al. 2001, Vavra 2005). In this regard, grazing can be treated as a habitat manipulation tool to address specific management

goals nested within a comprehensive wildlife or range management plan (Holechek et al. 2001, Vavra 2005). Success of multi-species grazing plans is dependent on collaborative approaches involving multiple stakeholders who are committed to a common outcome. In practice, this approach can prove challenging because of competing objectives. However, the rapid rate of urban and exurban development occurring throughout much of the CPE and the American West is creating a mutual need for ranchers and wildlife managers to develop collaborative multi-species grazing systems (Jensen 2001). Additional incentive is provided by the economic value of deer and elk hunting for local communities and landowners.

Mule deer winter ranges are particularly vulnerable to habitat loss resulting from development across the CPE. Large ranches have been sold for urban or exurban development, often resulting in considerable wildlife habitat loss. Perhaps less obvious, grazing management on public lands can potentially influence development rates and patterns. Many small ranches located in valleys are dependent on public land grazing leases. These small ranches often provide critical winter range habitat, and when sold, are susceptible to high-density urban development. Low-density, exurban developments (i.e., 1 dwelling per 35 – 80 acres) may not be detrimental, or could even be beneficial. However, available research indicates that rangeland condition, as judged by several habitat characteristics, was better on operating ranches than exurban developments (Maestas et al. 2002). Also, at least some portions of the CPE have seen a trend toward higher-density developments during recent years, even in remote, high-elevation locations. As a general rule, mule deer populations will fare better when managed in conjunction with livestock than with development. Many long-time ranching families prefer to keep an operating ranch rather than sell to developers. These 2 factors have created a reason for natural resource interests to collaborate with ranching interests in managing multi-species grazing systems on the landscape (Fig. 9).

ISSUES AND CONCERNS

Grazing and Mule Deer Habitat: An Overview

Livestock, elk, or other grazers may affect mule deer habitat directly, through removal of forage that otherwise would be available to deer, and indirectly by causing long-term changes in community composition (Holechek et al. 2001). The type and extent of direct impacts depends on grazing pressure; light to moderate pressure might be beneficial while moderate to heavy pressure will generally be detrimental (Fig. 10). The extent of dietary overlap among interacting ungulates is equally important when evaluating potential for direct competition (Mackie 1970, Peek and Krausman 1996, Beck and Peek 2005). Studies of direct competition are often best accomplished under controlled

Figure 8. Big sagebrush is an important forage item for wintering deer. Healthy sagebrush stands with an herbaceous understory provide critical winter range habitat for deer populations in the CPE. (Photo by Chad J. Bishop/CDOW).

Figure 9. Many ranches across the CPE have been converted to urban or exurban developments. Habitat loss from development has become a substantial threat to mule deer and numerous other wildlife species. Ranchers and wildlife managers have mutual interests in developing sustainable multi-species grazing systems. (Photo by Chad J. Bishop/CDOW)

Figure 10. Overgrazed aspen summer range (right) contrasted with ungrazed aspen habitat (left). Heavy grazing pressure will almost always be detrimental regardless of the grazing system employed. (Photo by Brandon Diamond/CDOW).

conditions (e.g., high-fenced pastures with known ungulate numbers), although application to landscape management requires knowledge of ungulate interactions under free-ranging conditions. A number of studies have evaluated competitive interactions between livestock and deer (e.g., Mackie 1970, Kie et al. 1991, Loft et al. 1991, Beck and Peek 2005) or between elk and deer (Lindzey et al. 1997, Wisdom et al. 2004b). Intraspecific competition among deer and detection of density dependence has also been a focus of research (e.g., Caughley 1970, Bartmann et al. 1992, White and Bartmann 1998a, McCullough 1999, Bishop et al. 2005, Forsythe and Caley 2006). Understanding direct competition among ungulates is fundamental to management of multi-grazer systems.

Long-term, indirect impacts of grazing are difficult to study and even more difficult to predict. Some evidence suggests livestock grazing has positive long-term impacts for deer by maintaining shrubs in habitats that would revert back to grass-forb communities in the absence of grazing (Austin and Urness 1998). Ample research has demonstrated that forage selection varies among grazer species, and therefore removal of 1 grazer altogether is likely to alter long-term community composition (Mackie 1970, Holechek et al. 2001, Beck and Peek 2005). If a range were managed for only 1 grazer species, the resulting long-term trend is likely to be undesirable for that species. Management for multiple species at appropriate stocking levels may result in overall higher animal and plant productivity (Holechek et al. 2001). Long-term, indirect impacts of grazing could therefore be beneficial to mule deer in this ecoregion. Alternatively, overgrazing leads to a host of ecological problems, including invasion and perpetuation of noxious weeds. These changes can have long-term negative impacts on community composition and rangeland condition that may not be easily reversed (Fleischner 1994, Pieper 1994).

Therefore, whether dealing with direct or indirect impacts, heavy grazing pressure is likely to be detrimental while low to moderate grazing is likely to be tolerable or possibly beneficial if managed appropriately (Peek and Krausman 1996). What constitutes 'low' or 'heavy' grazing pressure varies by ecoregion and habitat type, and is typically determined by resulting conditions (i.e., stubble height guidelines or other use measures).

Mechanisms of Competition
Understanding the mechanisms of direct competition between mule deer, livestock, and other grazers is necessary to develop successful grazing plans. The degree of direct competition depends on available habitat, diet selection, and foraging behavior. Specifically, diet selection and foraging behavior interact at differing temporal and spatial scales to explain niche overlap (Stewart et al. 2002, Torstenson et al. 2006). The degree of overlap then determines the likelihood that the composite ungulate grazing will have negative effects on mule deer and their habitats.

Diet selection.
Cattle and sheep are well-adapted to bulk diets high in cellulose whereas mule deer prefer diets high in cellular content (Hanley 1982). Typically, mature grasses have thick cell walls that are high in cellulose whereas current annual browse growth, forbs, and immature grasses have thinner cell walls and therefore higher cell contents. Overall quality of mule deer diets is often low, however, because cell walls of browse species have high amounts of lignin, which is not digestible. Mule deer attempt to meet their dietary needs by capitalizing on the readily-digestible cellular content, increasing gut fill to maintain forage intake rates as lignin consumption increases, and efficiently passing the lignin (Hobbs et al. 1983, Baker and Hobbs 1987). Traits facilitating this strategy include various oral-

Figure 11. Mule deer have a variety of anatomical adaptations that allow them to forage selectively. Wintering deer attempt to select diets that are relatively high in cell solubles but also high in lignin. The non-digestible lignin is quickly passed, allowing them to "skim" the quality constituents from forage that is low in overall quality. In comparison, elk and livestock are more adapted to diets high in cellulose (e.g., mature grass) because of their larger rumens (in proportion to body size) and slower rates of passage. (Photos by R. Bruce Gill/CDOW, retired).

pharyngeal adaptations that favor selective feeding capability, the need to fill only a portion of the rumen when consuming moderate to high quality diets, a low rumino-reticular volume to body weight ratio (i.e., relatively small rumen), a comparatively slow rate of cellulose digestion, and a high rate of passage (Fig. 11, Hanley 1982, Hobbs et al. 1983, Baker and Hobbs 1987, Hofmann 1989). Elk select diets intermediate to those of deer and livestock. The variable foraging strategies and digestive physiology among ruminants have led to divergent food niches (Hobbs et al. 1983), which is why total ruminant biomass supported on a given range will typically be higher with multiple grazer species rather than a single species (Holechek et al. 2001). Although overall diet composition will vary among different types of grazers, direct competition will occur if livestock or elk consume forage items that are limited in supply but preferred by mule deer.

Kufeld et al. (1973) provided a general description of mule deer forage use by summarizing numerous food habits studies across the West. Shrubs and trees comprised 74% of winter diets, 49% of spring and summer diets, and 60% of fall diets. Forb consumption ranged from 15% during winter to 49% during summer, and graminoid consumption ranged from 3% during summer to 26% during spring. These broad generalizations indicate why mule deer have been coined 'browsers,' yet they also indicate the importance of forbs and grasses in the diet. The relative importance of shrubs, forbs, and grasses varies spatially as well as temporally. In Middle Park, Colorado, on the eastern edge of the CPE, grasses were found to comprise as much as 80% of the diet during mid-winter (Carpenter et al. 1979). Conversely, mid-winter diets in pinyon-juniper winter range in northwestern Colorado consisted of 80 – 99% shrubs and trees (Bartmann 1983). In central Arizona, winter diet composition ranged from 48% browse and 34% grass in treated ponderosa pine habitat to 94% browse and 1% grass in untreated alligator juniper (*Juniperus deppeana*) stands, while forb consumption was highest (39%) in untreated Utah juniper habitat. Summer diet composition ranged from 22% browse in treated alligator juniper stands to 75% browse in untreated Utah juniper stands, and 24% forbs in untreated Utah juniper habitat to 69% forbs in treated alligator juniper habitat (Neff 1974). Clearly, deer diet selection is driven in large part by site-specific factors, rendering broad generalizations inadequate for understanding competitive relationships between deer and other ungulates.

Elk food habits are extremely variable. Kufeld (1973) and Cook (2002) described elk diets by summarizing food habits studies from across the West. Winter diets were dominated by shrubs or grasses, depending on forage availability. Grasses were important during spring and forbs became particularly important in summer, although diets were highly variable depending on forage availability. Fall diets were typically dominated by grasses, although some studies found shrubs to be the principal component. Interpretations of the relative importance of grasses, forbs, and shrubs vary widely, even when considering only studies in the CPE. For example, in New Mexico, Lang (1958) found that shrubs comprised 77% and 95% of fall and winter diets, whereas Sandoval et al. (2005) found that grasses and forbs comprised nearly 70% of fall and winter diets. However, Sandoval et al. (2005) observed 61% shrubs in elk diets during summer, which is when others have typically seen increased use of forbs. In northern Colorado elk herds, Boyd (1970) and Hobbs et al. (1981) observed gradual transitions from grass to browse as winter progressed, with forbs comprising little of the diet. In southern Colorado, Hansen and Reid (1975) found that grasses comprised the majority of elk diets in all seasons. Of note, Hobbs et al. (1981) observed that elk diet composition during winter was not strictly related to availability, finding that elk maintained a mix of browse and grass even in habitats where 1 forage class dominated. Elk can exhibit even more diverse and variable diet selection than deer, which makes it difficult to draw broad conclusions.

Cattle diets are generally dominated by grasses, but cattle also consume substantial amounts of shrubs and forbs (Peek and Krausman 1996). Holechek et al. (2001) provided a summary of cattle diets based on various studies across the West. Average diet composition was 60% grasses, 18% forbs, and 22% shrubs. Sheep diets comprise substantial amounts of each of the major forage classes, and can be quite variable depending on availability (Peek and Krausman 1996). Holechek et al. (2001) summarized sheep diets, finding an average composition of 48% grasses, 32% forbs, and 21% shrubs. Given the dietary breadth of mule deer, elk, and livestock, site-specific assessments incorporating each species are needed to evaluate potential for direct competition.

Riordan (1956) conducted 1 of the first experiments evaluating simultaneous use of forage by deer, cattle, and sheep. He controlled livestock and deer stocking rates across 6 experimental pastures in northwest Colorado and measured fall-winter use of 3 indicator species: mountain mahogany, Utah serviceberry (*Amelanchier utahensis*), and bluebunch wheatgrass (*Pseudoroegneria spicata*). Browse was used heavily by deer, moderately by sheep, and minimally by cattle. Conversely, bluebunch wheatgrass was used heavily by cattle and sheep and minimally by deer. The results indicated a low probability of competition between cattle and deer based on these 'key' forage species. Riordan (1956) also demonstrated the potential importance of site-specific, intraspecies variation in forage quality and use. Laycock and Price (1970) provide a good review of environmental factors influencing intraspecies plant

variation at local scales. Mackie (1970) evaluated relationships of cattle, deer, and elk in the Missouri River Breaks in Montana. Results indicated a low potential for direct competition between mule deer and cattle based on differing food and range-use habits. However, there was a potential for competition between deer and elk during spring through early fall given overlapping forage use. Hansen and Reid (1975) evaluated forage relationships of cattle, deer, and elk in southern Colorado. Dietary overlap between deer and elk ranged from 3% during winter to 48% during summer. Interactions with cattle occurred only during summer months, when dietary overlap between deer and cattle ranged from 12% to 38%.

Need for additional understanding of simultaneous foraging relationships among deer, elk, and livestock is evident from several studies published within the past 2 years. Beck and Peek (2005) evaluated interactions among cattle, sheep, deer, and elk on summer range in northeastern Nevada. Dietary overlap was lowest between deer and cattle, consistent with past research. Deer diets comprised 30% browse, 64 − 72% forbs, and 2 − 5% graminoids, while cattle diets comprised ≥ 92% graminoids. Potential for competition was higher between deer and sheep, and deer and elk. Overall, potential for ungulate forage competition was highest for forbs in aspen communities. Therefore, monitoring forbs was identified as the key component of a multi-species grazing management system in this area. In Wyoming, Torstenson et al. (2006) found elk and cattle diets to be dominated by grasses while mule deer consumed more forbs and shrubs. Greatest dietary overlap occurred between mule deer and elk during spring and between elk and cattle during multiple seasons. Findholt et al. (2004) observed considerable dietary overlap among mule deer, elk, and cattle, indicating a potential for competition. Overlap between elk and deer was consistently ~ 60% under various grazing history scenarios. Sandoval et al. (2005) evaluated elk and mule deer diets in north-central New Mexico where livestock grazing had been absent for 60 years. They observed an overall dietary overlap of 64% between deer and elk, indicating a high potential for competition.

Foraging behavior.
Understanding spatial and temporal foraging patterns of ungulates is critical for managing grazers on the landscape and evaluating competitive interactions. Foraging ungulates must find plants best capable of meeting their nutritional needs while avoiding plant toxins, negotiating plant physical defenses, and responding to ever-changing plant biochemistry and landscape changes (Provenza 2003). Large-scale distribution patterns are determined by abiotic factors such as distance to water, terrain, cover relationships, and snow depth (Fig. 12, Bailey et al. 1996). Finer-scale distribution patterns are driven by biotic factors,

primarily forage quality and quantity, which dictate patch selection and time spent feeding in a given patch (Bailey et al. 1996).

Considerable research has focused on cognitive abilities of livestock and to what extent feed-patch selection is influenced by spatial memory (Senft et al. 1987, Bailey 1988, Peeples 1991, Bailey et al. 1996, Provenza et al. 1998, Keil 2001). Maze studies with forage stations (i.e., food patches) have been used to test ungulates' ability to distinguish relative food availabilities by recalling which portions of a maze had already been visited (Bailey 1988, Bailey et al. 1989, Peeples 1991). For the most part, these studies have demonstrated that ungulates have spatial memories of specific food patches lasting hours to weeks, which improve foraging efficiency (Bailey et al. 1996). Enhanced foraging efficiency via spatial memory has also been demonstrated under experimental conditions in deer (Gillingham and Bunnell 1989).

Another important cognitive ability is learned behavior, often developed early in life from dams (Provenza 2003). Howery et al. (1998) demonstrated that cattle placed in an allotment used areas that directly overlapped those used by their dams years earlier. In effect, they demonstrated that cattle distribution was affected by experiences gained early in life. Similarly, female mule deer are known to occupy home ranges that overlap or are adjacent to those of their dams. Mule deer in general exhibit strong site fidelity even when landscape disturbances dramatically alter habitat in their home ranges. Radio-collared adult female deer exhibited nearly identical spatial and temporal use patterns within summer home ranges during repeated years (C. J. Bishop, CDOW, unpublished data). Young ungulates learn forage selection and avoidance from their dams, which may have a life-long influence on foraging behavior. Foraging behaviors are also learned through trial-and-error and from

Figure 12. Water sources influence livestock distribution on the landscape. Areas immediately adjacent to water often receive heavy use. (Photo by Jim K. Garner/CDOW).

social interactions in herd animals (Provenza and Balph 1988, Provenza et al. 1992, Scott et al. 1995, Provenza 2003). Herd 'leaders' can have considerable influence on distribution and foraging patterns of other animals in the herd (Bailey et al. 1996). This social dynamic is well-recognized in livestock and elk.

Foraging dynamics models have been developed based on knowledge of abiotic and biotic factors, foraging functional responses, spatial memory, and learned behaviors. Bailey et al. (1996) described a landscape model that integrated abiotic factors (e.g., slope, distance to water) and spatial memory in predicting feeding site selection. The model is species-specific and has a memory decay function. Models at this scale are potentially useful for applied management of multi-species grazing systems to avoid or resolve overgrazing problems and habitat degradation.

Finally, knowledge of foraging behavior is a fundamental prerequisite to behavior modification, which can be useful for addressing competitive interactions among ungulates and for using livestock as ecosystem management tools (Vavra and Ganskopp 1998, Provenza 2003). Perhaps the best example is redistribution of cattle from riparian areas to uplands without fencing.

Evidence of Ungulate Competition

Competition Between Livestock and Mule Deer.
Although dietary overlap between cattle and deer is typically low, multiple studies have demonstrated that cattle grazing altered mule deer habitat use or foraging behavior, consistent with hypotheses of direct competition. Loft et al. (1991) found that mule deer and cattle selected the same habitats, but as cattle grazing occurred, deer shifted use toward habitats avoided by cattle. Other studies also found that deer avoided areas used by cattle (Stewart et al. 2002, Coe et al. 2004). Two separate studies demonstrated that increased cattle stocking rates caused deer to increase time spent feeding (Kie et al. 1991, Kie 1996). Cattle grazing also reduced hiding cover for deer (Loft et al. 1987). Other studies demonstrated deer preferred ungrazed habitats to grazed habitats (Austin et al. 1983, Bowyer and Bleich 1984, Austin and Urness 1986, Wallace and Krausman 1987, Ragotzkie and Bailey 1991). Generally speaking, studies conducted in semi-arid or arid regions provided evidence that cattle negatively interfere with mule deer foraging and habitat use, which could potentially depress deer productivity.

Competition Between Elk and Mule Deer.
Elk and deer populations overlap extensively throughout most of the CPE (Fig. 13). During the past 20 years, elk numbers increased at the same time deer numbers generally decreased (Lindzey et al. 1997, Gill et al. 2001, Keegan and Wakeling 2003). Deer diets overlap with those of elk more than with cattle, creating a higher potential for forage competition. Elk are less selective than deer and capable of using a greater 3-dimensional foraging space, giving them an apparent competitive advantage (Hofmann 1989, Gill et al. 2001, Keegan and Wakeling 2003). Multiple studies indicate a relatively high potential for competition based on habitat use and dietary overlap (Mackie 1970, Hansen and Reid 1975, Findholt et al. 2004, Beck and Peek 2005, Sandoval et al. 2005). However, concrete evidence of deer-elk competition from experimental studies is lacking, at least when competition is defined in terms of negative effects on population productivity (Lindzey et al. 1997).

The impact of elk populations on deer populations remains poorly understood at best. A majority of western states observed recent declines in deer populations where elk were not present or not a factor (Lindzey et al. 1997). In Colorado, where most deer populations overlap abundant elk, high elk harvests (used as a crude index to elk abundance) were just as likely to be associated with productive deer populations as they were unproductive deer populations (Gill et al. 2001). Available data do not indicate that increasing elk populations caused widespread declines

Figure 13. Deer and elk foraging on a pinyon-juniper-sagebrush winter range. Large populations of deer and elk are sympatric throughout much of the CPE. Elk populations have generally increased while deer populations decreased during the past few decades, although it is unclear whether elk have negatively impacted deer populations. (Photo by Chad J. Bishop/CDOW).

Figure 14. Mule deer on heavily used sagebrush winter range. Understanding density dependence is a key aspect of deer population and habitat management. (Photo by David J. Freddy/CDOW).

dependence is difficult. Modern ungulate management paradigms are rooted in density dependence theory even though empirical evidence is limited. One problem has been an overall failure to estimate population size with defensible scientific methods (Gill 1976, Wolfe 1976). As a result, we have typically struggled to quantify what population changes actually occurred, let alone adequately determine the underlying mechanisms. Density dependence is further complicated by fluctuating abiotic conditions such as temperature and precipitation (Milner et al. 1999, Aanes et al. 2000, Coulson et al. 2001, Wang et al. 2006). For example, as the frequency of severe winters and/or drought increases, environmental carrying capacity becomes increasingly variable and the population becomes less stable. In fact, a deterministic density-dependent paradigm is unreasonable in ecosystems that essentially lack equilibrial forage conditions due to frequent, stochastic abiotic perturbations (DeAngelis and Waterhouse 1987, Ellis and Swift 1988). Elucidating the degree of density-dependent population regulation in various systems is necessary because perspectives on density dependence influence everything from hunting license allocations to which limiting factors should be managed (Fig. 14).

Documentation of density dependence requires some type of density manipulation and corresponding measurements of population parameters (Fowler et al. 2006). Several key studies in the CPE have documented density-dependent population regulation. Studies in northwest Colorado, conducted in typical CPE pinyon-juniper winter range, documented compensatory fawn mortality and density-dependent fawn survival by manipulating coyote (*Canis latrans*) and deer densities, respectively (Bartmann et al. 1992, White and Bartmann 1998a). A recent study in southwest Colorado documented density dependence by manipulating deer nutrition and measuring the rate of population change (Bishop et al. 2005). These studies provide empirical evidence of density dependence in pinyon-juniper habitats, which has fundamental implications for habitat management in the CPE given observed declines in deer populations. Specifically, successional management of habitats (see Successional Changes), in combination with population management of deer and competing herbivores, can be expected to largely dictate deer population performance. Temporal variability in weather intensifies density-dependent feedback to population growth whereas spatial variability in habitat resources weakens density-dependent feedback (Wang et al. 2006). The latter finding suggests that management for habitat mosaics (i.e., resource heterogeneity) might have positive effects not only on individual foraging but on long-term population stability and growth rates.

in deer, particularly when other factors such as habitat loss and deterioration are considered. However, it is likely that some deer populations are impacted by competitive interactions with elk. Stewart et al. (2002) observed resource partitioning between deer and elk, while others found that deer avoided areas used by elk (Johnson et al. 2000, Coe et al. 2004).

Competition Between Deer and Other Native Ungulates. There are minimal concerns regarding competition between deer and other native ungulates across the CPE (excluding elk). White-tailed deer are rare or absent across most of the CPE (Baker 1984). Pronghorn are common across east-central Arizona, but they typically occur in small, scattered populations across the remainder of the CPE (Yoakum 2004a). Where deer and pronghorn are sympatric, potential for competition is typically low due to spatial segregation (Mackie 1981, Yoakum 2004b). Bighorn sheep are common across the CPE and are generally sympatric with mule deer, but they occupy only a small portion of overall mule deer habitat and are often spatially segregated (Mackie 1981).

Intraspecific Competition.
Understanding interspecific relationships between deer and other ungulates is important primarily to the extent that it exacerbates intraspecific factors, which are ultimately tied to density-dependent population regulation. Understanding density dependence in the context of temporal and spatial environmental variation is arguably the most important aspect of deer population and habitat management. Density dependence is fundamentally tied to carrying capacity concepts and is often assumed to be the mechanistic explanation of observed deer population changes. In fact, the widespread decline in deer during the 1960s and early 1970s is believed to have occurred primarily because deer populations exceeded carrying capacity (Workman and Low 1976). While this is probably true, detecting density

Optimal management of deer and other herbivores requires quantitative knowledge of species-specific animal numbers

and habitat capability. This management approach has been an objective of wildlife management agencies for many years, yet optimal population management continues to be a significant challenge. State agencies have put forth considerable efforts to monitor deer and elk populations in the CPE and elsewhere, and these efforts will likely persist or intensify (Mason et al. 2006). Estimates of deer and elk numbers are fundamentally important to objective-based population management, just as identification of appropriate stocking rates is fundamental to livestock grazing management. Population estimates can then be compared to population objectives for making management decisions. Determining population objectives with scientific rigor is no easier than estimating abundance, in part because social and economic factors must be considered. Even if socio-economics are ignored, determining the number of deer that can be supported on a landscape is a tremendous challenge. Carrying capacity estimates depend on the appropriate integration of ruminant nutrient requirements, foraging behavior and diet selection, and forage quality and availability under free-ranging conditions (Swift 1983, Hobbs and Swift 1985). Nutrient-based carrying capacities of ungulates have been estimated by relating total forage supply to herbivore energy and nitrogen requirements (Wallmo et al. 1977, Hobbs et al. 1982). Hobbs and Swift (1985) developed a more robust procedure for estimating carrying capacity by integrating forage availability and forage quality rather than treating the two as independent variables. Their approach allows estimation of carrying capacity for differing levels of herbivore nutritional status, which is useful for evaluating habitat management scenarios and herbivore population objectives. Temporal variation adds an additional layer of complexity when estimating and interpreting carrying capacity. While applicable, techniques to rigorously estimate nutrient-based carrying capacity have been deemed too intensive and costly for routine population management.

Summary.
There is overwhelming evidence that deer experience competitive interactions with livestock and elk across the CPE and elsewhere. The magnitude of these effects in terms of deer productivity is poorly understood. The extent to which cattle, sheep, or elk may be responsible for observed declines in deer is unknown. Perhaps more importantly, even the best landscape herbivory models will likely fail to adequately predict deer population responses to reductions in cattle and/or elk. An improved understanding of deer competition with elk, livestock, or other ungulates will invariably require more rigorous experiments where deer population responses are quantified. There is a distinct need for additional research in the CPE because the potential for excessive herbivory is high, especially across winter range habitats.

Intraspecific competition among deer has been documented in pinyon-juniper habitats of the CPE. Elk and livestock likely exert additional density dependent feedback on deer, although the magnitude undoubtedly varies spatially and temporally. These findings suggest a need for improved successional management of habitats as well as enhanced herbivore population management. Carrying capacity models have been developed for specific deer and elk populations by integrating nutritional requirements with measurements of forage resources and knowledge of foraging behavior. Recent efforts have focused on integrated ecosystem models (e.g., SAVANNA) that incorporate spatial and temporal heterogeneity in determining carrying capacities for multiple grazers (Coughenour 1993, Weisberg et al. 2002). These models represent the state-of-the-art in multi-grazer management; however, data requirements are too resource-intensive to be used for routine population management. The future may depend on simplified models that optimize the tradeoff between scientific rigor and funding to allow routine assessments of habitat conditions and forage allocation (Roath et al. 2006). Ultimately, such models could effectively bridge the gap between rangeland health and herbivore population management in applied State and Federal agency programs.

Indirect Impacts of Grazing
Indirect impacts refer to long-term changes in plant community composition and structure caused by grazing (Holechek et al. 2001). Concerns over indirect impacts have traditionally focused on effects of cattle and sheep grazing, although effects of elk are becoming relevant in some areas given high elk abundance in recent decades. A common explanation for increases in deer populations during the mid-20th century is that excessive livestock grazing in combination with drought conditions favored a conversion of many grassland habitats to shrub-dominated habitats (Julander 1962, Julander and Low 1976, Urness 1976, Robinette et al. 1977). This theory is based on research primarily conducted in Utah. Recent research in northern Utah indicated that livestock grazing continued to have positive long-term impacts by maintaining shrubs in habitats that would have otherwise reverted back to grass-forb communities (Austin and Urness 1998). However, it is equally apparent that long-term grazing pressures in other ecosystems have had negative impacts on deer.

Ungulates modify the ecosystems they inhabit by affecting nutrient cycling, productivity, and disturbance regimes (Hobbs 1996). Predictions of how ecosystem processes will be affected depend on evolutionary relationships, habitat type, abiotic factors such as climate and soils, distribution and prevalence of invasive plant species, and combinations of grazing species and grazing intensities (Augustine and McNaughton 1998, Holechek et al. 2001, Milchunas 2006). The role of abiotic factors is apparent at broad spatial scales

when considering the effect of annual precipitation and soils on ecosystem resilience to grazing. Desert ecosystems of the Southwest are easily impacted by grazing while tallgrass prairie systems in the central United States evolved with bison (*Bison bison*) grazing and are extremely resilient (Holechek et al. 2001). Herbivore diet selection and foraging behavior are predictors of long-term, direct and indirect grazing impacts on ecosystems (Senft et al. 1987, Coughenour 1991, Bailey et al. 1996). Theories of plant succession are helpful for understanding indirect grazing impacts (Briske et al. 2005, Cingolani et al. 2005).

Considering winter range habitat in the CPE, mule deer are highly vulnerable to negative impacts of long-term grazing because of low precipitation and the dominance of pinyon-juniper habitat, which is highly susceptible to grazing impacts. Holechek et al. (2001) identified pinyon-juniper as 1 of the most depleted range types in the West and 1 of the slowest to recover from overgrazing. As already discussed, the exception to this general rule is in northern Utah sagebrush-grass ranges, where reductions in grazing have lowered shrub prevalence. Austin and Urness (1998) identify the loss of shrub forage as a major factor in declining deer populations. Future condition of northern Utah foothill ranges is particularly uncertain because of noxious weeds. Summer ranges across the CPE are also susceptible to negative, indirect impacts of grazing, particularly in the southern portions of the ecoregion. Increasing elk populations could exacerbate negative, long-term impacts on deer because of their foraging similarities with cattle. Whether considering direct or indirect impacts, grazing must be carefully managed across the CPE to avoid negative impacts to deer, especially on pinyon-juniper range. At the same time, understanding and manipulating livestock and elk impacts may be valuable for positively affecting long-term ecosystem management for deer.

Grazing Systems

The major grazing strategies employed in the CPE include: 1) continuous, 2) deferred-rotation, 3) seasonal-suitability, 4) rest-rotation, and 5) short-duration (Kruse and Jemison 2000, Holechek et al. 2001). As the name implies, animals under continuous grazing are placed in a single, large pasture or allotment for the entire grazing season. Under deferred-rotation grazing, the range is divided into multiple pastures and each pasture receives deferment every 2 – 4 years. Deferment means that grazing is delayed until key forages reach seed maturity. Seasonal-suitability systems redistribute livestock throughout the grazing season based on differing vegetation classes, thereby attempting to optimally mesh plant community and livestock requirements. Rest-rotation is similar to deferred-rotation except rested pastures receive no use for the entire year. Short-duration grazing (i.e., Savory or holistic method) involves the repeated transfer of large numbers of livestock

among numerous pastures (i.e., ≥ 8). Livestock are typically moved after ≤ 5 days of use, resulting in a labor-intensive system (Kruse and Jemison 2000, Holechek et al. 2001).

Livestock distribution can become a serious problem with continuous grazing systems. This is especially true in CPE habitats where terrain is often rugged, water sources may be limited, and sensitive riparian areas can easily receive excessive use. From the standpoint of mule deer and other wildlife in the CPE, manipulated grazing systems are almost always preferable to passive, season-long grazing (Holechek et al. 2001). Manipulations include periodic redistribution of cattle or adjustments in timing and intensity of use. In some cases, behavioral modification can be used to address livestock distribution issues (Provenza 2003). Naturally, grazing systems that simultaneously consider wildlife, rangeland health, and profitability will be preferable to those that only focus on the latter. However, wildlife-friendly grazing systems that are complex or intensive to implement may not be used even when they would be economically feasible for the operator (Coppock and Birkenfeld 1999). Grazing systems that are cooperatively developed through input from wildlife and rangeland managers and livestock operators are likely to be the most beneficial because there is ownership by all interests.

Livestock Stocking Rate

Stocking rates are the most critical factor of grazing management (Holechek et al. 2001). Heavy grazing intensities will generally be damaging in CPE habitats regardless of the grazing system used, whereas light or light-moderate grazing intensities will typically be sustainable under any grazing system (Fig. 15). Stocking rates are often evaluated in terms of percent utilization.

Figure 15. A comparison of ungrazed (left) and grazed (right) rangeland following seasonal use under a managed grazing system with light-moderate stocking levels. The grazing plan meets livestock management objectives while maintaining habitat value for mule deer and other wildlife. (Photo by Jim K. Garner/CDOW).

Table 1. Summary of authorized use of Bureau of Land Management grazing district lands and grazing lease lands, Fiscal Year 2004 (October 2003 – September 2004).*

STATE	NUMBER OF AUTHORIZATIONS			ANIMAL UNIT MONTHS AUTHORIZED		
	CATTLE AND BISON	SHEEP AND GOATS	HORSES AND BURROS	CATTLE AND BISON	SHEEP AND GOATS	HORSES AND BURROS
ARIZONA	608	2	69	308,109	842	2,630
COLORADO	1,178	115	56	209,047	47,738	1,514
NEW MEXICO	1,957	254	376	963,942	96,496	13,370
UTAH	1,053	147	46	415,602	116,913	1,499
TOTAL	4,796	518	547	1,896,700	261,989	19,013

* Data source: USDI (2004).

Figure 16. Riparian degradation caused by excessive livestock use. (Photo courtesy of BLM).

Based on a compilation of studies across North America, average use of forage was 57% under heavy grazing intensity, 43% under moderate grazing, and 32% under light grazing (Holechek et al. 2001). Ecological condition declined under heavy grazing scenarios and improved under light grazing, and net profits per acre were the lowest under heavy grazing intensities. Based on studies across the Southwest, net profits are maximized when stocking rates result in about 30 – 45% forage use (Pearson 1973, Holechek 1994, Winder et al. 2000, Holechek et al. 2001). When factoring in periodic drought, conservative stocking rates (i.e., 35% forage use) will likely maximize long-term profitability and sustainability of semi-arid rangelands across the CPE.

Precipitation and spatial use patterns are arguably the 2 most important factors to consider when setting and adjusting stocking rates. Temporal precipitation patterns

demand that stocking rates be dynamic, particularly in the CPE where drought conditions can quickly reduce grazing capacity (Thurow and Taylor 1999, Horn et al. 2003). Spatial use patterns are important because not all forage is equally available or equally used (Bailey et al. 1996). Forage distant from water or on steep slopes may be functionally unusable for livestock, particularly cattle (Holechek 1988, Bailey et al. 1996). Even if behavioral management is used as part of a complex grazing system to optimize distributional use patterns, some forage will be underused or not used at all. Therefore, total forage estimates are often not reflective of grazing capacity.

Stocking rates in the CPE were reduced in the early 2000s in response to drought conditions. Total authorized livestock Animal Unit Months (AUMs) by the BLM in Colorado, Utah, New Mexico, and Arizona decreased 38% from 3,496,445 in 2000 to 2,177,702 in 2004 (USDI 2000, 2004). In 2004, cattle and bison represented 87% of the authorized AUMs (Table 1).

Grazing in Riparian and Xeroriparian Habitats
Grazing in riparian areas in the CPE is of special concern because of the high potential for habitat degradation (Belsky et al. 1999). Riparian and xeroriparian (i.e., habitat immediately adjacent to dry or ephemeral watercourses) habitats are very attractive to livestock and can receive a disproportionate amount of use relative to other habitats (Fig. 16). The importance of riparian and xeroriparian ecosystems for mule deer tends to increase with aridity. These habitats often provide critical water, forage, and cover resources that are otherwise unavailable in desert or semi-desert environments.

Most livestock grazing in arid or semiarid riparian areas is ecologically harmful (Belsky et al. 1999). The challenge

Figure 17. Healthy riparian habitat following seasonal grazing (left of fence), demonstrating how responsible grazing management can maintain or restore riparian habitat while meeting livestock management goals. (Photo by Jim K. Garner/CDOW).

facing livestock managers is to design grazing systems that prevent riparian degradation (Fig. 17). A common goal of federal land managers and ranchers is to graze the landscape and simultaneously restore degraded riparian ecosystems. The most common strategy has been livestock exclusion through the use of fencing, which has been an effective restoration tool (Elmore and Kauffman 1994). However, stream corridor fencing as a routine management strategy is not practical nor always desirable. Ultimately, riparian restoration depends on grazing systems that integrate reduced stocking rates, rest, careful timing, and redistribution of livestock away from riparian areas (Elmore and Kauffman 1994). The potential and need for livestock behavioral management is apparent (Provenza 2003). Stubble height guidelines are presently used as a standardized way to manage and monitor grazing impacts in riparian ecosystems (Clary and Leininger 2000, Boyd and Svejcar 2004).

Improving Habitat with Livestock
An important element of the environmental debate over livestock grazing in recent years has been whether livestock are beneficial to wildlife and rangeland health when managed appropriately. As just discussed, grazing in riparian ecosystems is not beneficial largely because the likelihood for damage far exceeds any potential positive outcomes. In other habitats, though, some suggest livestock can be important habitat management tools. As with any complex issue, there is no right or wrong answer. Rather, livestock can have negative, neutral, or positive effects on plants, ecosystems, and wildlife depending on the objectives of the grazing system and how effects are defined. For example, a specific cattle grazing system could have positive effects on mule deer by increasing vigor of certain deer forages (Paige 1992, Noy-Meir 1993) and maintaining shrubs in what were historically grassland

habitats (Austin and Urness 1998). The same grazing system evaluated at a different temporal or spatial scale could have negative effects by altering deer foraging behavior and reducing cover (Loft et al. 1987, Kie et al. 1991, Kie 1996). Also, an action creating positive effects for 1 species may be detrimental to another species (Severson and Urness 1994, Holechek et al. 2001, Vavra 2005).

Livestock impact habitat by modifying nutritive quality, productivity, structure, and/or availability of plant species and individual plants, which often lead to alterations in community composition (Severson and Urness 1994, Vavra 2005). Theoretically, livestock can be used to manipulate habitat in desired ways by carefully managing these impacts. Successful applications depend on a thorough understanding of livestock foraging behaviors and predicted plant responses. For example, whether grazing causes reduced growth or compensatory growth of plants depends on plant species, site-specific environmental factors, and level of herbivory (Paige 1992, Noy-Meir 1993, Painter and Belsky 1993). There are relatively few examples where livestock were used explicitly to improve habitat for wildlife. Anderson and Scherzinger (1975) documented improvements in forage quality for elk using prescribed cattle grazing. Reiner and Urness (1982) used horses to increase bitterbrush production on a sagebrush-grass range in northern Utah. Holechek et al. (2001) provided a summary of grazing approaches that have been used in an attempt to improve habitat for deer and other wild ungulates.

There are various ways livestock could be managed to potentially benefit wildlife. Specific management objectives and desired outcomes must be explicitly stated prior to the grazing treatment, and impacts on other desirable wildlife species should also be considered. Additional research on herbivore foraging behavior and herbivore-plant interactions should improve our ability to prescribe grazing treatments that accomplish intended objectives (Vavra 2005). However, increasingly specialized grazing management for wildlife will likely fail to maximize economic returns for livestock producers, thereby limiting the application of such grazing treatments (Holechek et al. 2001). The appeal of using livestock to manipulate wildlife habitat rests largely with federal and state agencies attempting to conduct habitat treatments at minimal expense. A realistic objective for typical livestock producers is to lessen or minimize negative impacts of grazing on habitat rather than employ systems designed exclusively for wildlife. In many cases, a grazing system will need to be developed or restructured to meet this objective, indicating the importance of communication and collaboration among rangeland managers and livestock operators (Coppock and Birkenfeld 1999).

GUIDELINES

A. Management Approach

State wildlife agencies responsible for managing mule deer and elk populations do not have regulatory control over livestock grazing on federal or private lands, yet most mule deer habitat is under federal or private ownership. The majority of federal lands are managed by the BLM or the USFS, both of which operate under multiple-use mandates that include livestock grazing. Private ranches typically graze livestock for some type of economic benefit, unless they have been converted to urban or exurban developments. Guidelines intended to lessen impacts of livestock grazing on mule deer habitat must embrace these realities to be effective. Three principal guidelines should be applied:

1. State wildlife agencies should establish collaborative partnerships with private and federal entities to develop and implement herd-specific grazing plans and to coordinate habitat management. The state will likely need to bring monetary resources to the table to establish the partnership. State dollars can be used for habitat improvement on private and federal lands and for reducing wildlife impacts on private lands. Once established, partnerships can facilitate integrated land management and resource planning that simultaneously considers rangeland ecosystem health, wildlife, and livestock. Examples include:

   - Colorado Habitat Partnership Program (HPP)
   The HPP is a state program that seeks to develop partnerships between landowners, land managers, sportsmen, the public, and the Division of Wildlife to resolve big game conflicts with agricultural interests (http://wildlife.state.co.us/LandWater/PrivateLand Program/HPP/). The HPP is funded using a percentage of big game license revenues.

   - Arizona Habitat Partnership Committee (HPC)
   The HPC is a state program designed to facilitate local decision making regarding wildlife habitat issues and to act as a vehicle for developing partnerships among private, state, and federal entities (http://www.azgfd. gov/w_c/). The HPC is funded by the sale of special big game licenses as well as other sources.

   - The Uncompahgre Plateau Project (UP)
   The UP is a partnership in western Colorado whose purpose is to "develop a collaborative approach to restore and maintain the ecosystem health of the Uncompahgre Plateau, using best science and public input" (http://www.upproject.org/). The UP is a collaborative among the BLM, USFS, CDOW, and the Public Lands Partnership (PLP) and is funded by grants and the participating agencies.

2. Livestock and wild ungulate carrying capacities should be evaluated holistically and be used to guide stocking rate decisions and wild ungulate population objectives. Public and private resource managers should collectively apply the best scientific principles currently available regarding differential herbivore diet selection and foraging behaviors to manage multiple herbivores in an ecosystem context (Weisberg et al. 2002). This type of an approach is the only viable way to positively influence herbivory levels on managed landscapes where regulatory control of use is spread among multiple public and private entities with different agendas. Domestic stocking rates and wild ungulate population objectives should not be fixed targets. Rather, they should be developed as ranges to reflect the wide variance in carrying capacity over time due to climatic variation. There is not a "best" approach for determining how many wild and domestic herbivores can be supported on the landscape under differing management scenarios. However, 2 main components are needed:

   - An ecosystem grazing model incorporating spatial and temporal variation – The model should integrate climate and habitat variables with herbivore-specific forage selection and behavior to establish forage allocation scenarios and to approximate overall carrying capacity of the landscape. The purpose of the model is to coherently summarize available information to guide decision-making. There are 2 options currently available:

      • SAVANNA model (Coughenour 1993, Weisberg et al. 2002) – SAVANNA is a comprehensive but very data-intensive model with limited practical application except for intensively studied landscapes where optimal estimates of joint herbivore capacities are needed.

      • Habitat Assessment Model (HAM, Roath et al. 2006) The HAM is a simplified version of the SAVANNA model that can be practically implemented as part of routine management. Herd-specific model development and implementation requires approximately 1 year. Although the HAM makes numerous assumptions, it can provide a reasonable framework for guiding decision-making.

   - Evaluation of relative deer and elk forage utilization - Just as livestock producers are expected to know their stocking rates, wildlife managers should have a method to quantitatively assess deer and elk forage utilization. This can be accomplished directly by monitoring forage condition, availability, and use by deer and elk in the absence of livestock grazing or indirectly by developing defensible deer and elk population estimates. Failure to establish a quantitative basis for evaluating deer and elk foraging effects may compromise state agency credibility when requesting livestock reductions. Forage condition, availability, and use in areas where livestock are excluded by wildlife-friendly fencing can be monitored using fixed transects, clip plots, and caged plots (McNaughton et al. 1996, Stohlgren et al. 1998). Population estimates can be obtained using aerial

surveys and/or population models incorporating measured inputs (Samuel et al. 1987, Unsworth et al. 1990, Freddy 1994, White and Bartmann 1998b, White and Lubow 2002, Mason et al. 2006).

3. State wildlife areas (SWAs) should be optimally managed for deer, elk, and other wildlife habitat. Grazing should not be allowed unless specifically managed to enhance wildlife habitat. Irrigated forage production should be principally managed for wild ungulates on the property. Production of grass hay on SWAs provides little benefit for deer. Upland habitats should provide examples of optimal management for adjacent private and federal lands. Emphasis should be placed on managing for desirable native plant species and locally-adapted ecotypes. Woven-wire and other unfriendly wildlife fences should be removed or replaced. In short, state agencies must demonstrate healthy ecosystem management on their own properties to have success influencing land use on federal and private lands.

B. Grazing System
There is no single grazing system that is optimal in all ecosystems under all conditions. Specialized grazing systems should be developed to minimize direct competition between livestock and deer, maintain ecosystem health, and accommodate basic needs of livestock operators. The following guidelines should be considered when developing a grazing system:

1. Avoid passive, season-long grazing (Holechek et al. 2001).
2. Prevent excessive use and degradation of riparian and xeroriparian areas using behavioral management, wildlife-friendly fencing, and/or upland water developments (Elmore and Kauffman 1994, Provenza 2003).
3. Rest-rotation grazing systems in combination with deferment should be considered in most CPE habitats to minimize potential for habitat degradation and to lessen competition.
4. Principal summer range habitats of mule deer should not be grazed during late May and June when deer give birth to fawns (i.e., grazing should be deferred), particularly when spatial overlap among livestock and deer is expected (Loft et al. 1987, 1991; Kie et al. 1991). May and June is typically a good time to graze deer winter range because deer have already migrated, herbaceous forage is most available, and it should have little effect on habitat quality the following winter (Jensen et al. 1972, Smith et al. 1979, Austin 2000).
5. Winter range habitats should not be grazed during late summer and fall to avoid excessive use of desirable shrubs (Jensen et al. 1972, Austin 2000).
6. The grazing system should be responsive to range trend monitoring. Flexibility is therefore needed so that reductions in stocking rate or pasture rest may be accommodated without harming the livestock operator.

Consideration should be given to setting some allotments aside as grass banks that can be used for livestock grazing if rest is deemed necessary in existing allotments.

C. Stocking Rate/Utilization and Range Monitoring
1. Stocking rates should be responsive to drought and range monitoring to avoid habitat degradation.
2. Stocking rate plans should be developed that specify grazing reductions and alternative management actions to be implemented under differing drought scenarios (Thurow and Taylor 1999, Horn et al. 2003). Such planning will minimize response times, preserve rangeland health or at least minimize degradation, and help the livestock operator minimize any financial losses that may be incurred during drought conditions.
3. The influence of terrain and distance to water on livestock forage use should be factored into stocking rate decisions because some portions of the landscape will receive minimal or no use by livestock, especially cattle. Holechek (1988) made the following recommendations: 1) a 30% reduction in cattle grazing capacity for $11-30\%$ slopes; 2) a 60% reduction in grazing capacity for $31-60\%$ slopes; 3) slopes exceeding 60% should be considered ungrazable by cattle; 4) a 50% reduction in cattle grazing capacity for range $1-2$ miles from water; and 5) range >2 miles from water should be considered ungrazable by cattle.
4. Site-specific stocking rates should be developed for each herd unit based on soils, habitat types, current ecological conditions, wild ungulate populations, and landscape features. However, realizing this may not always be feasible, we offer the following general guideline based on research conducted in the southwest (Holechek et al. 2001, Roath et al. 2006): use from all herbivores in aggregate should not exceed $\sim 35-45\%$ of the forage commonly available to all species, or $\sim 25-35\%$ of the total forage on the landscape (which accounts for some forage being unavailable to livestock).
5. Range monitoring is a requirement for responsible management and provides the basis for subsequent decision-making (SRM Task Group 1995, Austin 2000, Clary and Leininger 2000, Holechek et al. 2001, Pyke et al. 2002):
- Monitoring sites should be identified by "ecological site" as a standard procedure for evaluating site potential and site characteristics. Ecological site is defined by the SRM Task Group (1995) as "a kind of land with specific physical characteristics which differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its response to management".
- The monitoring plan should clearly specify the desired plant community (DPC) for each site based on deer habitat management and/or various other habitat management objectives.

- Key monitoring components include erosion status, forage use, ecological condition, and trend data evaluated with respect to the DPC.
- In riparian areas, measurements of stubble height (i.e., use) should be taken as a guard against overgrazing. Clary and Leininger (2000) recommend a residual stubble height of 10 cm as an initial threshold for monitoring.

D. Habitat Manipulations

1. Successional management via habitat manipulations should be considered as a technique for increasing overall herbivore capacity on ranges where natural disturbance regimes have been eliminated or greatly altered (see Successional Changes).
2. Livestock and elk herds are often attracted to newly treated areas because of the herbaceous response, which may compromise ultimate success of the habitat treatment. For best results, particularly when treatments are designed for mule deer, the following steps should be taken:
   - Rest pastures containing habitat treatments from livestock grazing for ≥1 year immediately following treatment.
   - Pair mule deer winter range treatments with higher-elevation treatments designed for elk.
   - Design and implement a complex of habitat treatments on a landscape to help minimize an ungulate swamping effect.
   - In areas with high deer and elk densities, shrub establishment may require shrubs to be planted as seedlings using nursery stock. In extreme cases, temporary high fence may be required to exclude wild ungulates (in addition to livestock) until shrubs have been successfully established.

E. Fencing

1. Fences should be constructed to accommodate wildlife passage. Wildlife-friendly fencing will save livestock operators money by reducing fence repairs, particularly in areas with elk.
2. Fences that are not wildlife-friendly should be removed or replaced.
3. State wildlife agencies should offer support to private operators willing to replace wildlife-unfriendly fences.
4. Mule deer cross fences by jumping over the top strand, crawling underneath the bottom strand, or crossing between strands. Mule deer and elk neonates must cross underneath fences during the first weeks of life. Therefore, wire fences with ≥5 strands and woven-wire fences (i.e., net-wire fences) should be avoided at all costs, especially on summer range. Unfortunately, sheep allotments often use woven-wire fencing.
5. Wildlife-friendly wire fences comprise:
   - Three or 4 total strands
   - A smooth bottom wire ≥16 inches above the ground

- A smooth top wire, that ideally is vinyl coated for increased visibility, ≤42 inches above the ground (38-40 inches is preferred)
- At least 12 inches between the top 2 wires
6. Wildlife-friendly rail fences should include a maximum of 3 rounded rails separated by 16 inches with a maximum height of 48 inches. This allows passage underneath, through, and over the fence.

SUCCESSIONAL CHANGES

BACKGROUND

The impact of plant succession on mule deer habitat in the CPE varies with a number of correlated factors including elevation, climate, soils, and ultimately, vegetation type. Higher elevation habitat types in the CPE are primarily composed of deciduous and coniferous forests. Non-riparian deciduous forests are typically a monoculture of aspen, whereas coniferous forests are composed of spruce-fir and some lodgepole pine (*Pinus contorta*) communities that transition into ponderosa pine at mid-elevations. Descending in elevation, the primary vegetation types shift to Gambel oak and other mountain shrub species (i.e. mountain mahogany, serviceberry, and snowberry). The lowest vegetation communities, which typically serve as mule deer winter range, are primarily composed of pinyon-juniper woodlands and sagebrush, blackbrush (*Coleogyne ramosissima*), or saltbush (*Atriplex* spp.) shrublands.

Many of the deer in this ecoregion migrate between relatively moist higher elevation, summer range habitats and lower, drier, foothill-basin wintering areas (Carpenter and Wallmo 1981, Kie and Czech 2000). In most of the CPE, this movement primarily occurs in April and May and again in October and November. In many areas, deer making seasonal movements will use mid-elevation, mountain shrub/Gambel oak transitional ranges that can provide high quality forage. During mild winters (i.e., minimal amounts of snow), deer will use transitional range for extended periods.

As was noted by Carpenter and Wallmo (1981), throughout much of the intermountain west and specifically in the northern part of the CPE, mule deer are primarily limited by forage quality and quantity on winter range. Summer range resource limitation is also possible in some areas, especially in the southern part of the CPE where aspen and mountain shrub communities are limited. While there is less evidence indicating that transitional ranges play a limiting role for mule deer in the CPE, they can provide abundant, high quality forage that can improve the condition of deer prior to arriving on winter ranges and help deer regain condition more quickly in the spring.

In general, as plants mature and reach late seral stages, they have inherently established themselves and have thus out-competed other plants for resources. However, even when the dominant plant types are highly useful to mule deer, the overall impact may not be positive. There is often an inverse relationship between plant age and forage value for ungulates. As such, younger and more diverse plant communities are often most beneficial to mule deer (Wallmo 1978, Stevens 2004a).

Research has shown that both vegetation and deer can respond positively to disturbance. For instance, Shepherd (1971) concluded, through experimental browse clipping research, that consistent removal of 50-60 percent of current annual growth was sustainable for serviceberry, oakbrush, antelope bitterbrush, and big sagebrush while removal rates ≤80% were sustainable for mountain mahogany. Additionally, he concluded that at moderate removal rates (20-30%), browsing was invigorating and decreased die-off of leaders.

Kufeld (1983) demonstrated large increases in forb and grass production following treatment of Gambel oak communities. Regardless of treatment type (burning, chaining, and spraying), forb production increasd from 14% to 27% 2 years post treatment. Similarly, large increases in grass and shrub production were measured post treatment on all treatment types except for burned areas.

Figure 18. Aerial herbicide treatments are often necessary or preferred in areas with rough, uneven terrain or in areas where road access is limited. (Photo by Eric J. Bergman/CDOW).

Ultimately (10 years post treatment), production of forbs and grasses increased on all treatment types. Production of shrubs increased due to chaining, but decreased due to spraying and burning. Despite changes in plant production, no changes in deer use were detected for 2 years post treatment. However, 5 years post treatment and extending through 10 years post treatment, deer showed preference for sprayed units.

Anderson (1969) found that areas of sagebrush that were partially treated with 2,4-D (as opposed to areas of complete treatment, or no treatment at all) had higher forage yields. However, in an effort to achieve the same result via a different approach, Carpenter and Wallmo (1981) recommended treating sagebrush communities by stimulating the growth of other forage types, in lieu of destroying sagebrush.

In some circumstances, use of herbicides may be the most effective way to impact vegetation in mule deer habitat.For instance, not all landscapes are conducive to implementing mechanical disturbance (e.g., steep slopes, rocky sites, heavily timbered areas, high moisture areas, roadless areas). In situations such as these, application of herbicides may be the only cost-effective approach. Aerial application of herbicides allows for quick treatment of large areas, as well as areas that are not accessible to other equipment (Fig. 18). Additionally, in easily accessible areas, herbicides can be applied with standard farm equipment

Although, availability, selectivity, and cost of herbicides continue to improve, certain herbicides have become mainstays. In particular to the CPE, 2,4-D (several trade names) and Tebuthiron (Spike) have primarily been used to control sagebrush in areas where it has out-competed other plants. As a broad-spectrum herbicide, 2,4-D is used to control broadleaf plants, whereas Spike is more specific for woody species. There are a number of other herbicides currently in use. However, most of these are primarily used to control noxious weeds (Table 2). Application of the herbicides of interest is traditionally done through foliage spray (both aerially and via ground delivery). Aerial application allows for treatment of large areas

Table 2. Common herbicides used for mule deer habitat improvement in the Colorado Plateau Ecoregion.

COMMON NAME	TRADE NAME	TARGET GROUP
2,4-D	Various	Broadleaf plants
Tebuthiron	Spike	Woody plants
Glyphosphate	Roundup	Grasses and forbs
Oust	Oust	Annual grasses
Imazapic	Plateau	Annual grasses

in short periods of time and, if effective application is possible using fixed-wing aircraft, can be very cost effective. Ground delivery can be labor intensive.

Another type of habitat modification involves mechanical disturbance. While use of equipment to reset seral stage has occurred for the past 4-5 decades, the actual processes and methods employed have changed. Primary changes have come in the form of refinement of equipment, thereby allowing greater flexibility and selectivity in the delivery of disturbance. In light of these changes, economics surrounding mechanical disturbance have remained relatively unchanged. Mechanical disturbance is usually relatively expensive and may not always be an option.

However, when treatments are carefully designed and implemented they can successfully re-establish key browse species. During the past decade, use of mechanical equipment for resetting seral stage on pinyon-juniper winter range has focused on 2 primary techniques. The least expensive of these is the roller-chopper, a bulldozer pulling a large, water-filled, steel drum with perpendicular blades (Fig. 19). Depending on the needs at specific sites, a seeder can be placed between the bulldozer and the drum, thereby allowing seeding to occur in the same process. Seed dribblers that deposit seed onto the tracks of the bulldozer can also be used for some browse species that require deeper planting (e.g., four-wing saltbush, mountain mahogany, antelope bitterbrush). The objectives of a roller-chopper are 5-fold: 1) uproot or break and chop larger diameter trees at ground-level, 2) distribute and create seed beds for desirable species, 3) provide cover for seed, 4) create water catchments, and 5) stimulate existing forage by trimming (Stevens and Monsen 2004a).

The second primary type of mechanical disturbance is the hydro-ax. The hydro-ax is a 6-8 foot wide, boom-mounted mulcher, affixed to a reticulated tractor (Fig. 20). While the cost per acre for hydro-ax treatments is considerably higher than that for roller-chopper treatments, there are some advantages to hydro-ax treatments. Due to the mulching nature of the hydro-ax, overstory vegetation ultimately ends up as ground cover which facilitates establishment of desired seedlings by trapping moisture and providing optimum microclimates for seedling establishment. Also, depending on the time of year employed, soil scarification by hydro-axes can be greatly minimized, thereby reducing the potential for establishment of noxious weeds. Hydro-ax treatments are usually more aesthetically pleasing than roller-chopping or anchor-chaining treatments and therefore often more acceptable to the public.

ISSUES AND CONCERNS

As vegetation communities age, their utility to deer changes. Forage production decreases dramatically when aspen communities are replaced by conifers because

understory productivity is reduced due to shading. As pinyon-juniper stands reach late seral stages their value as escape and thermal cover increases, but understory

Figure 19. Roller choppers offer a more affordable method of mechanically treating large areas. (Photo by Bruce Watkins/CDOW).

Figure 20. Hydro-ax treatments, while typically more expensive than roller-chopper treatments, allow high mosaic variability to be built into small treatments and are often more aesthetically appealing to the public. (Photos by Bruce Watkins/CDOW).

Figure 21. Understory productivity typically diminishes over time in pinyon-juniper woodlands. (Photo by Jim K. Garner/CDOW).

Figure 22. Reduced understory productivity in a late seral sagebrush community is often a product of age and competition. (Photo by Jim K. Garner/CDOW).

vegetation is reduced by shading effects and reduced water availability (Fig. 21). Late seral Gambel oak and mountain shrub communities can become so dense that deer movement is restricted and forage production and available leader growth are reduced. However, older and taller sagebrush plants can improve habitat quality by functioning as wind and snow breaks, thus providing refuge from harsh winter conditions and breaking up snow pack which enhances foraging efficiency. Late seral stage sagebrush can also out-compete surrounding vegetation, resulting in little or no understory growth (Fig. 22).

As plants mature, their quality as forage for mule deer generally declines. During early, pre-senescent stages, the majority of current annual growth occurs as leaders (Short and Reagor 1970). Leaders are typically more digestible for deer and are a highly preferred plant part. As plants age and reach later seral stages, they tend to produce fewer leaders (Hormay 1943), cell walls tend to thicken and

become less digestible, and anti-herbivory responses become more developed. Anti-herbivory responses are physiological or morphological changes such as increased production of secondary compounds (e.g., volatile oils, tannins, alkaloids) or structures (e.g., spines, thorns, sharp awns) that reduce palatability and foraging selection. Thus, whereas many habitat improvement efforts are intended to replace undesirable species, others are intended to replace overly-mature plants with younger, more useful plants of the same species.

Another common concern surrounding winter range habitat quality across the CPE pertains to encroachment of pinyon-juniper forests into surrounding areas (Figs. 23 and 24). Although mature pinyon-juniper forests provide high quality thermal and escape cover for mule deer, expansion of these forests into surrounding grass and sagebrush parks leads to further reduction of browse. As pinyon-juniper forests expand and age, they eliminate understory vegetation by depriving other plants of sunlight and nutrients, and by intercepting moisture. A primary source of annual moisture for winter range vegetation comes via winter snowfall. As pinyon-juniper forests reach later seral stages, canopy cover can approach 100%. During winter months, dense canopy cover prevents snow from reaching the ground. By holding snow above ground, sublimation occurs, thereby minimizing the amount of moisture that reaches ground level via melting. Pinyon-juniper expansion along stand edges is largely a function of animal species that serve as dispersal agents, physical structure adjacent to the woodland, and availability of nurse plants in surrounding edge communities (Schupp et al. 1999). Eisenhart (2004) concluded that cycles of pinyon-juniper expansion and thinning follow an ebb and flow pattern that is strongly related to drought and pluvial periods.

Similar to pinyon-juniper forests, mature sagebrush can also greatly reduce understory vegetation. Encompassing a large proportion of deer winter range in the CPE, the sagebrush-steppe habitat type has been subject to widely varying attitudes about its value. Sagebrush often out-competes grasses, and has thereby been subject to various forms of eradication or control in attempts to increase forage availability for livestock (Carpenter and Wallmo, 1981). Deer use and reliance upon sagebrush during winter is well documented. However, deer cannot subsist on an exclusive diet of sagebrush for extended periods of time (Carpenter and Wallmo 1981). As such, the ideal structure of sagebrush communities includes adequate amounts of other herbaceous forage.

Regardless of habitat type, quality of typical winter range diets is inadequate to prevent catabolism and weight loss in mule deer. However, the rate of weight loss can be reduced by improving winter range forage conditions. In addition to

Table 3. Common, native, winter and transition range shrubs, forbs, and grasses used by mule deer in the CPE. List compiled from Kufeld et al. (1973), Carpenter et al. (1979), Milchunas et al. (1978), Wallmo and Regelin (1981), and Bartmann (1983). Scientific names are provided in Appendix A.

SHRUBS	FORBS	GRAMINOIDS
Big sagebrush	Aster	Indian ricegrass
Serviceberry	Sagewort	Needle and thread grass
Mountain mahogany	Phlox	Great Basin wildrye
Snowberry	Snakeweed	Sandberg bluegrass
Rabbitbrush	Cryptantha	Blue grama
Bitterbrush	Globemallow	Bottlebrush squirreltail
Gambel oak	Buckwheat	Junegrass
Rose	Penstemon	Needle grass
Chokecherry	Fringed sage	Idaho fescue
Aspen	Goldenweed	Bluebunch wheatgrass
Ponderosa pine	Arrowleaf balsamroot	Western wheatgrass
Cliffrose	Lupine	Mutton bluegrass

Figure 23. Sagebrush park prior to pinyon-juniper woodland expansion. (Photo by C. Whitman Cross/U.S. Geological Survey [USGS]).

Figure 24. Sagebrush park after pinyon-juniper woodland expansion. (Photo by David Bradford/USFS).

sagebrush, important shrub species on winter range in the CPE include serviceberry, bitterbrush, mountain mahogany, and cliffrose.. Important forbs include buckwheat (*Eriogonum* spp.), fringed sage (*Artemisia frigida*), and phlox (*Phlox* spp.). Useful grasses include blue grama (*Bouteloua gracilis*), wheatgrass (*Agropyron* spp., *Pseudoroegneria* spp.), fescue (*Festuca* spp.), and bluegrass (*Poa* spp., Table 3).

Habitat treatment efforts typically focus on increasing the abundance of desirable plants and/or reducing abundance of undesirable plants. Dependent upon the primary objective, different habitat improvement techniques should be used accordingly (Monsen 2004). Within these objectives, there are 4 categories of disturbance treatments:

fire, harvest treatment, chemical treatment, and mechanical disturbance. Not all treatment methods are useful in all habitat types.

Use of fire as a disturbance technique is losing momentum due to social and political constraints. As the density and frequency of urban and exurban development increases in mule deer habitat, use of fire will be increasingly restricted due to potential for inadvertent destruction of structures and infrastructures and because of air quality issues. In particular, there has been a pulse of human development in recent years in the pinyon-juniper foothills that surround many communities. Whereas fire was a natural occurrence in these habitats prior to Euro-American settlement, its current presence (whether natural or artificial) is seldom

tolerated. Despite these issues, fire still has a role in areas with little or no human development. Unfortunately, these areas primarily occur at higher elevations in areas where structure and quality of habitat is often satisfactory. An alternative to controlled burning is application of timber

Figure 25. Examples of large-scale treatments with low edge/treatment ratios that are of questionable value for mule deer. Note the less distinct mosaic patterns in the bottom 2 photographs that were created using a roller-chopper 30 years after the large areas were anchor-chained in the 1970's. Also, note the lack of sight barriers left along roads and the geometric shapes of the treatments. (Photos by Bruce Watkins/CDOW).

harvest treatments. Timber harvest often meets the multi-use mandates of land management agencies as it can allow for resource use and be beneficial to wildlife. However, as is the case with fire, areas most conducive to timber harvest occur at higher elevations. Pinyon and juniper trees have little value as timber and are often only harvested for firewood or fence posts although there is increasing interest in possible use of pinyon and juniper as biomass fuels.

Use of chemicals as a habitat treatment varies in appropriateness depending on landscape, land ownership, time of year, and vegetation to be treated. Under some circumstances, use of chemicals can provide the best alternative for achieving desired results. Chemicals can be used to set back succession and/or to remove undesirable species. As was highlighted by Vallentine (2004), chemical treatments 1) can be used where mechanical methods are not feasible, 2) provide a selective means of killing sprouting plants that are unaffected by top removal, 3) are generally less expensive than mechanical methods, 4) maintain vegetal litter, 5) do not disturb soil or expose it to erosion, and 6) can often be applied via equipment and machinery that is readily available. Potential negative aspects of chemical treatments are that no single chemical is effective on all plants, non-target plant species can be negatively impacted, and effectiveness may not always be realized on lands of low potential (Vallentine 2004).

Mechanical habitat treatments include use of roller-choppers, hydro-axes, flails, anchor chains, Dixie harrows, brush beaters, aerators, and disks. As is the case with chemical treatment, there are both distinct advantages and disadvantages with mechanical treatment. Mechanical treatments can be implemented in close proximity to developed areas where fire and chemicals may not be tolerated, seeding operations can be more effectively incorporated, and they are often conducive to subsequent assessment or follow-up treatments. Disadvantages include terrain and access constraints for equipment (e.g., steep, rocky slopes), relatively high cost, creation of future access for motorized vehicles, soil compaction, and soil disturbance that can lead to erosion and noxious weed invasion.

Efforts to improve mule deer habitat by resetting vegetative seral stage can yield unintended negative results. A major concern is invasion of undesirable plant species following treatment. In the CPE, cheatgrass invasion is a major threat to any winter range habitat treatment. With few exceptions, disturbance treatments on winter range must be reseeded to reduce the probability that cheatgrass and other undesirable species will become established or proliferate following disturbance. Other considerations are treatment scale, design, and juxtaposition. Treatments that are too small can easily be overwhelmed and ultimately produce

unsatisfactory results because of excessive use, not only by deer, but also by elk and livestock. Elk often appear to be more attracted to habitat treatments than deer and winter range treatments intended for mule deer can sometimes draw elk from their more traditional wintering areas. Whenever feasible, habitat treatments primarily intended for mule deer should be combined with higher elevation treatments that will be attractive to elk. Large-scale treatments that have a low edge:treatment ratio may receive little use and be largely ineffective for mule deer because of a lack of escape and thermal cover (Fig. 25).

GUIDELINES

To positively influence and change the impacts of plant maturation and successional development across mule deer range, necessary steps can be grouped into 3 stages: planning, treatment delivery, and post-treatment assessment.

A. Planning

Prior to delivery of any habitat treatment, careful consideration of treatment design and capacity needs to occur. There are a number of issues surrounding habitat treatments that, if not considered during the design phase, could ultimately result in effectively reducing the quality of habitat in treatment areas.

1. Identification of highest priority areas - Across much of the CPE, winter range appears to be the most limiting habitat type. However, this may not always be the case. Prior to conducting habitat treatments for deer, habitat components that are most likely limiting the deer population in the area should be identified and assessed.

2. Development of a comprehensive habitat treatment plan - Prior to initiating treatments, a landscape level treatment plan that coordinates treatment efforts over many years is necessary. Without a comprehensive plan, treatments are likely to occur in piece-meal efforts and will not be integrated with one another. The potential for reducing effectiveness increases greatly without *a priori* planning on the landscape level. Ideally, the treatment plan should be based on ecological attributes rather than exclusively on land ownership and administrative boundaries.

3. Treatment scale and design - Treatments should be large enough that they are not overwhelmed by ungulate use. This is best accomplished by conducting many smaller treatments separated by cover rather than conterminous large treatments. A high edge:treated area ratio with irregular edges and visual barriers should be maintained (i.e., avoid geometric shapes). In particular, Reynolds (1966) demonstrated that deer use of treated areas decreased beyond 590 feet from an edge. Thomas et al. (1979) predicted that smaller treatment areas (~ 5 acres) would receive more use than larger areas (≥ 25 acres) (Fig. 26).

4. Consideration of competition - Treatments should not be considered in areas where they are likely to receive heavy livestock use. Although some livestock grazing can be beneficial by stimulating regrowth of grasses or actually used as part of a treatment (e.g., salting oak brush so cattle will break it down; using domestic sheep, goats, and cattle to help control noxious species), the unintended increase in livestock use can reduce deer use to less than pre-treatment levels.

B. Treatment Delivery

As discussed above, habitat treatments are typically either mechanical or chemical in nature. The appropriateness of each technique is heavily dependent on the landscape suitability and the desired effect. However, regardless of primary treatment type, there are several key aspects that should be addressed.

1. Reseeding - Most mechanical treatments and prescribed burns on winter ranges with < 15 inches of annual precipitation should be reseeded to prevent non-native weed invasion. In areas with > 15" of annual precipitation, reseeding may not be imperative, but might improve the treatment effect. In a best-case scenario,

Figure 26. Examples of smaller treatment areas recommended for mule deer. The top photo shows hydro-ax treatments and the bottom photo shows roller-chopper treatments used to open dense pinyon-juniper canopy. (Top photo by Jim K. Garner/CDOW, bottom photo by Bruce Watkins/CDOW).

reseeding can be used in conjunction with planting seedlings of preferred species. Efforts to reestablish preferred species are a necessity from a plant recovery standpoint.

2. Seed type and quality - Diverse seed mixtures of native species, preferably locally adapted ecotypes, should be used when reseeding. Use of a seed mix increases community structure and function, initiates natural succession processes, increases the probability of success, improves ground cover and watershed stability, and increases habitat diversity (Stevens 2004). Non-spreading, non-native forbs with high palatability (e.g., alfalfa [*Medicago sativa*], small burnet [*Sanguisorba minor*], sainfoin [*Onobrychis viciaefolia*]) can also be used along with native species. Non-native grasses (e.g., Fairway crested wheatgrass [*Agropyron cristatum*], smooth brome [*Bromus inermis*], orchardgrass [*Dactylis glomerata*]) are not recommended and should only be used as a last resort for soil stabilization or to prevent site-dominance by invasive exotic species. Agencies should be proactive in the development of native seed sources for habitat projects. Utah has been a leader in this regard. Prior to treatment, a seed mixture should be in hand and tested for quality. Seeds of some common native grass and forb species (e.g., western wheatgrass [*Agropyron smithii*], bluebunch wheatgrass, Great Basin wildrye [*Elymus cinereus*], Indian ricegrass [*Oryzopsis hymenoides*]) are commercially available (Jorgensen and Stevens 2004). Seed should be as similar as possible to that of native species in the vicinity of the treatment area. Date, method, depth of seeding, germination rates, and compatibility of different species should also be considered (Monsen and Stevens 2004, Stevens and Monsen 2004b). Finally, prior to distributing seed, effectiveness of the delivery mechanism to be employed should be evaluated for each type of seed in the mix. Seeds establish at different rates and thereby need to be distributed at different rates (Stevens 2004a).

3. Browse establishment - One of Wallmo's (1978) axioms of mule deer habitat management was that more browse is preferable to less browse. Most winter range treatments should be done with the intention of increasing useable browse for deer. Reseeding shrubs, shrub transplants, and stimulating leader growth of extant shrubs should be priorities for most winter range treatments. Unfortunately, with the exception of sagebrush, four-wing saltbush, and bitterbrush, browse seed is often not as readily available as seed for some grasses and forbs. As such, it will likely be necessary to harvest seed from the local area for several years prior to delivery of treatments (Jorgensen and Stevens 2004).

4. Road avoidance - Treatment areas should be well screened from roads whenever possible by leaving tress and shrubs along travel corridors. Roads into treatment areas should be blocked whenever possible.

C. Post-Treatment Assessment

1. Assessment - The treatment plan should include monitoring to evaluate treatment results. This should include pre-treatment and periodic post-treatment vegetation measurements to evaluate species composition and abundance. Ideally this assessment should also include some measure of use (e.g., cage clipping studies). Pellet counts are commonly used but are probably of questionable value for assessing use.

2. Follow-up - In the event that post-treatment assessment indicates treatment results are unsatisfactory (e.g., seeding is ineffective, invasion of noxious weeds), an a priori commitment should be made to conduct follow-up treatments. In most circumstances, follow-up treatments will involve further seeding and/or herbicide application to control undesirable species.

NON-NATIVE INVASIVE SPECIES

BACKGROUND

Prior to European settlement, native vegetative communities were affected primarily by natural processes such as drought, fire (lightning-caused and Native American-caused), and native ungulate grazing. Non-native plant species had little or no impacts on the dynamics of rangeland plant communities. During the first 60 years of western settlement, a combination of overgrazing by livestock and introductions of competitive exotic plants set the stage for dramatic changes in plant communities (Miller et al. 1994). Many of the introductions of non-native invasive plants into the CPE likely occurred through crop seed and hay contamination or through the attachment of seeds on livestock brought by settlers to the region. Livestock numbers in the Intermountain West peaked in the early 1900s (Young et al. 1976). During this period, not only were animal densities high, but in many areas they were present throughout the year. This high-intensity grazing coupled with the introduction of exotic plant species, led to dramatically altered plant community compositions.

By the 1920s and 1930s, many sagebrush rangelands had exotic plant species introduced. Following difficult years of drought and depression, many dry-land farmers went bankrupt and abandoned their farms (Yensen 1981). The abandoned farms were quickly colonized by exotic plant species from Europe and Asia brought in with crop seeds. Species such as cheatgrass, Russian thistle (*Salsola tragus*), and tumblemustard (*Sisymbrium altissimum*) became prominent at this time (Pyke 1999). The passage of state and federal seed laws, such as the Federal Seed Act of 1939, helped reduce the transport and spread of exotic species in the United States. However, species such as cheatgrass had already become established throughout

western rangelands by the time these laws were enacted. Today, nonnative invasive weeds are spreading at an accelerated rate on public and private lands throughout the Colorado Plateau. These invasive plant species compete with native plant communities for scarce resources and limit the productivity of rangelands. Invading exotic plant species can have several negative environmental impacts, such as increased erosion and fire frequency, and decreased plant species diversity.

Wildlife populations, including mule deer, are adapted to native plant communities and can be adversely affected by altered plant communities caused by invasive weeds. Individual mule deer exhibit high site fidelity toward their selected home ranges, especially ranges used during summer and winter. Alterations in these preferred ranges caused by invasive exotic plant species can have dramatic impacts to mule deer habitat selection, survival, and sustained population size.

Mule deer throughout the CPE are highly dependent on sagebrush-steppe plant communities during certain periods of the year. Sagebrush-steppe communities are also among the ecosystems most vulnerable to invasion and degradation by invasive weeds. The exotic plant species of most concern that are well suited to rangelands within the CPE include cheatgrass, Russian, spotted, and diffuse knapweed (*Centaurea repens*, *C. maculosa*, and *C. diffusa*), Canada, musk, and scotch thistle (*Cirsium arvense*, *Cardus nutans*, and *Onopordum acanthium*), Dalmatian and yellow toadflax (*Linaria dalmatica*, *L. vulgaris*), hoarycress (*Cardaria draba*), leafy spurge (*Euphorbia esula*), houndstongue (*Cynoglossum officinale*), and tamarisk (*Tamarix ramosissima*). These exotic plant species have invaded tens of thousands of acres throughout the Colorado Plateau, causing detrimental changes to native plant communities and watersheds. The abundance and quality of mule deer habitat across the CPE has therefore been diminished by the aggressive invasion of these plant species. While several of these exotic species have had significant impacts to rangeland health, there is no species that has caused a more widespread negative affect on mule deer habitat than cheatgrass.

Cheatgrass

Cheatgrass is an exotic annual grass that was accidentally introduced into North America in the late 1800s. It is native to Southern Europe and Central Asia where it had invaded rangelands and agricultural fields (Young and Allen 1997). Cheatgrass likely entered the United States as seed in grain and hay as well as carried on livestock. One of the first accounts documenting the presence of cheatgrass in the West was in 1883 in Washington (Warg 1938). By 1902, cheatgrass had been recorded in Washington, British Columbia, Utah, and Oregon (Mack 1981). By the mid

1900s, cheatgrass had spread throughout rangelands in the western North America.

Cheatgrass can quickly become established after existing plant cover is disturbed by natural or artificial means. Cheatgrass is a prolific seed producer, which accelerates its rate of spreading into new sites. Disturbances such as mechanical soil manipulation, fire, and heavy livestock grazing all give cheatgrass added opportunity to colonize

Figure 27. Cheatgrass invading a sagebrush and native grass site in Utah over a 10 year period. (Photos courtesy of Utah Division of Wildlife Resources [UDWR]).

and invade new plant communities. Once present in an area, cheatgrass can readily invade healthy sites, even in the absence of disturbance (Fig. 27). Throughout the Colorado Plateau, cheatgrass has invaded plant communities in salt desert shrub, sagebrush zones, pinyon-juniper woodlands, and some ponderosa pine, and Douglas fir zones (Hull and Pechanec 1947). Mack (1981) estimated the complete range of cheatgrass in the Intermountain West at 99 million acres. Millions more acres in Utah, Colorado, Wyoming, New Mexico, and Arizona are currently at risk of invasion by cheatgrass.

Cheatgrass has several traits that enable it to aggressively compete with native grasses and forbs for valuable and limited resources. Cheatgrass is highly adapted to germinate under a wide range of temperatures. Seeds can germinate at temperatures as low as 32 F and as high as 104 F (Young and Evans 1985). This variability in germination allows cheatgrass to germinate earlier in the fall than most native grass species and during late winter when temperatures are too cold for native seeds to germinate. Cheatgrass seedlings have very rapid and elongated root development which is

1986 Before Fire

2005 After Fire

Figure 28. A sagebrush and perennial grass community in Utah before and after cheatgrass-fueled wildfire. (Photos courtesy of UDWR).

superior in advancement when compared to slower developing native grasses. The developed root system provides a competitive edge to cheatgrass seedlings in the spring when temperatures allow for shoot growth (Young et al. 1987). This early germination and rapid root growth increases the ability for cheatgrass to compete successfully for available nutrients and water (Harris 1967, 1977). Cheatgrass can produce more seeds per unit area than many native grasses, even when cheatgrass densities are low (Young and Allen 1997). These abundant seeds can lay dormant for >1 year until moisture and soil nitrogen conditions are suitable for germination (Pyke and Novak 1994). These adaptations allow cheatgrass to quickly become established, persist, and out-compete native grasses during periods of environmental stress.

ISSUES AND CONCERNS
Impacts of Cheatgrass on Mule Deer Habitat
Cheatgrass is a major concern for wildlife and land managers throughout the CPE. In some portions of the CPE where human-induced habitat alterations are of minimal magnitude, habitat deterioration caused by cheatgrass invasion is the primary factor threatening productive mule deer populations.

Cheatgrass does provide some nutritive value for mule deer during the green stage by providing similar moisture content, crude protein, crude fat and fiber, and lignin content to that of desirable grasses (Pyke 1999). In many areas throughout the CPE, cheatgrass is used heavily by mule deer after germination in the fall and again during green-up in the spring. During mild winters, young cheatgrass can be used by mule deer throughout the winter on ranges free of snow. During normal or heavy winters, cheatgrass provides little or no forage value to mule deer except for the brief periods after germination in the late fall and in the spring before cheatgrass reaches maturation. In general, cheatgrass provides some short-term forage benefits for mule deer while in the early stages of growth, but lacks the ability to provide high quality forage during most of the year.

The first impact to mule deer habitat that results from cheatgrass invasion is a reduction or eventual elimination of perennial grasses and forbs that are more palatable and nutritious. Once native grasses and forbs are greatly reduced by cheatgrass, the ability and likelihood for recovery of these communities is low. Cheatgrass exhibits typical morphological characteristics of grazing-tolerant plants that allow them to re-grow following defoliation (Archer and Pyke 1991). Its short, "green-feed" period, high variability in annual biomass production, and potentially injurious awns make cheatgrass a hazardous species on which to base livestock production (Stewart and Young 1939). Livestock tend to select more productive and

palatable native grass species, thus giving a competitive advantage to cheatgrass. This leads to the depression of native grasses and proliferation of cheatgrass on rangelands, particularly on ranges susceptible to drought. Livestock have been used to control cheatgrass, but effective treatment required use until plants reached the purple stage over several years (Mosley 1996). Daubenmire (1940) noted that if grazing is not continued, cheatgrass would quickly return. Using livestock to suppress or control cheatgrass on large-scale areas of infestation throughout the CPE is not only impractical, but likely impossible. The inability to impact cheatgrass abundance on a large scale through livestock grazing, in a manner similar to native grass communities, has resulted in continued expansion of cheatgrass into valuable mule deer habitat.

Secondly, cheatgrass increases the likelihood and intensity of fires on native rangelands. These fires are often in the mid- to lower elevational winter ranges of mule deer habitat and destroy native shrublands that provide critical forage and cover for mule deer (Fig. 28). Prior to European settlement, fire, both lightning caused and Native American caused, was an important environmental force that manipulated ecosystems toward perennial grasslands. During the intervals between fires, succession allowed shrub and tree recovery. Fires would typically occur every 20 to 100 years, with interval length depending on the plant community type (Miller et al. 1994). Shrubland ecosystems did not evolve with the frequency and intensity at which fires burn today (Robertson 1954, Billings 1994). With annuals like cheatgrass in an ecosystem, the return interval has shortened to as few as 5 years under some conditions (Whisenant 1990). Leopold (1941) realized and stressed that the increased chance of ignition and rate of spread of wildfires fueled by cheatgrass would prove very harmful to wildlife populations. This has certainly occurred through areas of the CPE. Large areas of antelope bitterbrush- and sagebrush-dominated winter ranges have burned or are at risk of burning due to cheatgrass prevalence. Cheatgrass invasion and the increased risk of fires on low elevation shrublands is one of the primary threats in maintaining mule deer winter habitat in eastern Utah, southwestern Colorado, southern Wyoming, northern New Mexico, and areas in northern Arizona.

Impacts of Other Non-native, Invasive Plant Species
Exotic, invasive plant species such as Russian, spotted, and diffuse knapweed, Canada, musk, and scotch thistle, Dalmatian and yellow toadflax, hoary cress, leafy spurge, houndstongue, and tamarisk all pose serious threats to mule deer habitat throughout the CPE. Unlike cheatgrass, these species are largely found in localized infestations and are not affecting areas on a landscape scale. They are very adaptable plant species that interfere with the natural propagation and succession of native grass and shrubland communities. Species such as Russian knapweed, spotted knapweed, diffuse knapweed, leafy spurge, and hoary cress are capable of infesting large tracts of rangelands up to several thousand acres (Figs. 29 and 30). These invasive species have a variety of morphological characteristics such as waxy leaves, toxic compounds, thorns, and spines that reduce their

Figure 29. Hoary cress, also known as whitetop, is an unpalatable invasive species that can dominate large areas such as this site on mule deer winter range in northwestern Colorado. (Photo by Rod Cook/La Plata County).
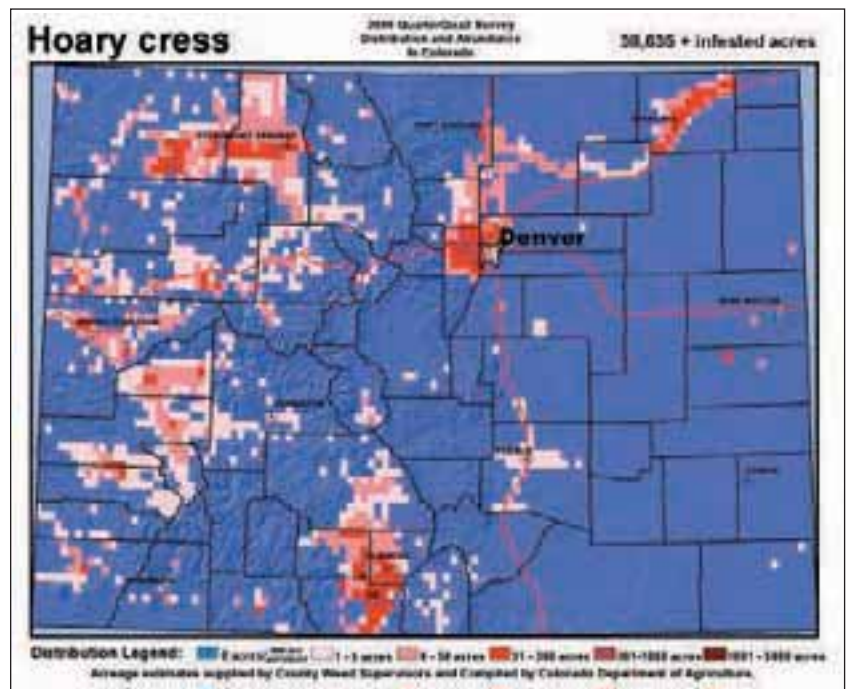

Figure 30. The extent of hoary cress infestation in Colorado shows how widespread invasive species can become. Wide distribution greatly reduces the effectiveness of localized control efforts. (Map courtesy of the Colorado Department of Agriculture).

Figure 31. Leafy spurge infestations, such as this one in southwestern Colorado, can affect large areas and can alter a site's ecological complexion. Although deer will consume some leafy spurge, it has little forage value compared to a diversity of native species. (Photo by Rod Cook/La Plata County).

palatability to mule deer and livestock. Avoidance by livestock and many wildlife species gives these species added advantages to develop, reproduce, and spread. Many invasive species also have superior root systems that allow them to out-compete native plants for soil nutrients and moisture and endure periods of extended drought. These attributes allow invasive species to accelerate their prevalence on landscapes, while native grasses and forbs gradually lose dominance and become sparse occupiers of the plant community (Fig. 31).

Tamarisk can pose serious threats to riparian areas and other moist sites in the drier zones of the Plateau. Unlike native riparian vegetation, tamarisk is not an efficient water user. A single large tamarisk tree may transpire 200 gallons of water per day (Hoddenbach 1987). Tamarisk can spread rapidly and replace desirable riparian vegetation that could otherwise serve as productive forage for mule deer (Fig. 32). Tamarisk often lives in the drier regions of the Colorado Plateau, but is nearly always associated with water sources. Water sources are often

Figure 32. Mule deer winter range in Colorado before and after tamarisk removal. Although tamarisk is usually associated with riparian corridors (top photos) it can also occur on more mesic sites (bottom photos). (Top photos by Randy Hampton/CDOW; bottom photos by Dave Hale/CDOW).

scarce in these drier regions, thus making them highly valuable to a variety of wildlife, including mule deer. Tamarisk can dramatically reduce the amount of surface water at springs and small water sources, limiting their use by mule deer.

GUIDELINES

A. The Management Plan

An initial inventory of habitat condition to determine presence and abundance of invasive plant species must be made in areas critical to mule deer. Invasive plant species of concern should be identified and prioritized according to their perceived threats to mule deer habitat. Throughout the Colorado Plateau, there is a wide range of varying topographic and soil types, elevation, plant communities, and different mule deer habitat types. Distinctions between these varied habitats should be created in an attempt to group similar habitat types and areas with similar invasive species concerns.

Once an assessment of invasive plant species distribution and abundance has been conducted, a prioritization of areas with invasive species should be made. Areas with highly valuable mule deer habitat and threats of current or future invasion should receive close attention. Efforts to establish range trend monitoring sites should be made to observe changes in invasive species density, distribution, and rates of invasion. These monitoring sites can also measure changes in vegetative composition as a result of the colonization and establishment of exotic plants. Data derived from these monitoring sites should be quantifiable and correlated to mule deer habitat quality. Mule deer population parameters and management objectives should be clearly defined for each high priority area of concern before prescribing vegetative treatments and invasive species control measures.

Wildlife managers, land managers, and habitat biologists must work closely together to define clear goals and objectives for areas of mule deer habitat in need of treatment. Historical trend data for vegetation and mule deer populations should be used to help determine where habitat manipulation is needed most. Areas needing vegetative manipulation and or invasive species control should be identified collectively and prescriptions made in concert with other wildlife and land use practices. Consideration must be given to private and Tribal lands, taking advantage of opportunities to inventory, monitor, and treat mule deer habitat within these areas. Agencies must seek opportunities to establish partnerships with a wide array of public and private organizations. This will prove valuable in gaining public support and securing adequate funding to conduct vegetation treatments.

B. Specific Guidelines

1. Mitigate the spread of non-native invasive plant species by using proper livestock grazing practices including rotational grazing systems, appropriate stocking rates, and altering season of use.
2. Use a variety of mechanical, cultural, chemical, and biological control methods to reduce the threats of invasive plant species and improve habitat for mule deer (Fig. 33).
3. Promote native grass, forb, and shrub communities by managing proper functioning communities for long-term sustainability and manipulating communities where plant species diversity is lacking.
4. To limit the dominance of cheatgrass, quickly rehabilitate rangelands burned by wildfire by seeding native or, if necessary, non-native species that will quickly establish and provide forage and cover for mule deer.
5. Using non-native seeded species following fire or in vegetation treatments should be considered when there is a threat of cheatgrass dominance immediately

Figure 33. Sprayer-mounted all terrain vehicles or aircraft are often the most the effective ways to distribute herbicides across rangelands. (Left photo by Dave Hale/CDOW; right photo by Bruce Watkins/CDOW).

Figure 34. A small stand of Russian knapweed (center) on big game winter range in Colorado is flagged to be sprayed. Early detection and eradication is the most effective method to control invasive plants. (Photo by Dave Hale/CDOW).

Figure 35. Most pinyon-juniper-sagebrush winter ranges in the CPE have some snow present during much of the winter. As snow melts, deer will often drift towards higher elevations. (Photo by Bruce Watkins/CDOW).

Figure 36. Irrigated agricultural lands have had the greatest positive impact on mule deer of any water-related developments in the CPE. (Photo by Bruce Watkins/CDOW).

following the disturbance. Although non-native species such as crested wheatgrass are not generally recommended, they are preferable to cheatgrass. Ideally, managers should proactively develop native seed sources for rangeland rehabilitation.

6. Proactively identify and treat high priority mule deer habitat that is at risk or being threatened by invasive species before exotic species become dominant on the landscape (Fig. 34).

7. Consider the potential for non-native plant invasions before new disturbances such as road construction, mineral development, prescribed fire, and recreational activities.

8. Support and implement new research and methods to reduce prevalence of cheatgrass in critical mule deer habitat.

9. Support efforts by public land managers that require certified weed free hay for feeding livestock on public lands.

10. Although total eradication of non-native invasive plant species is unlikely, goals should be made to reduce their rate of infestation, increase native plant diversity, and create stable plant communities capable of providing high quality mule deer habitat.

WATER AVAILABILITY

BACKGROUND

The CPE includes a wide variety of habitat types and life zones ranging from Upper Sonoran desert to alpine tundra. Average annual precipitation in the CPE varies from < 8 inches to > 40 inches depending primarily on elevation. Water availability is normally not a limiting factor for mule deer across most of the CPE. Although water can be in short supply in some lower elevation areas during summer, many deer in this ecoregion migrate to higher elevations in late spring where water is often relatively abundant during summer. Monsoonal flow in mid- to late summer often brings additional moisture to mountain areas during the hottest months. In the late fall, deer in the CPE typically concentrate on low elevation, low precipitation ranges to spend the winter. The majority of mule deer in the CPE winter in pinyon-juniper and sagebrush steppe habitats where annual precipitation typically averages 9-15 inches (Goodrich et al. 1999). Snow is often available in these areas during winter and low evaporation rates due to cold temperatures result in greater persistence of temporary water sources (Fig. 35). Based on water turnover rates, water requirements of mule deer would be expected to be lowest during winter (Longhurst et al. 1970).

There is little documented evidence that wildlife water developments in the CPE can have a positive effect on mule deer populations. Deer that winter in areas with little snow

or resident deer populations in some arid, lower elevation areas would be the most likely to benefit from increased water availability. By far, the greatest effect of non-domestic water developments on mule deer in the CPE has resulted from the creation of irrigated agricultural lands (Fig. 36). Although it does not appear that water is a major limiting factor for mule deer in the CPE, it is possible that severe drought conditions could create critical, short-term needs for free water sources. For example, deer appeared to become very dependent on artificial water sources during unusually dry conditions on Arizona's Kaibab winter range in 1989 (Heffelfinger 2006). Prolonged, extreme drought conditions would likely result in reduced forage production which could obviate the benefits of additional free water.

ISSUES AND CONCERNS

Water Requirements

Water is essential for all animals. Sources of water for mule deer include preformed moisture in forage, metabolic water derived through cellular oxidative processes, and free water including snow and dew (Robbins 1983, Cain et al. 2005). Mule deer in the CPE often occur in areas where snow is the only source of free water available during winter. To what degree free water availability can affect survival and productivity of mule deer is still a matter of debate (Severson and Medina 1983, Broyles 1995; Kie and Czech 2000). The amount of free water required depends on moisture content of forage, dry matter composition and intake, physiological status, activity, and environmental factors. Free water intake by deer and other ruminants is inversely related to forage moisture content (Verme and Ullrey 1984). High moisture forages can greatly reduce or eliminate the need for free water in many desert-adapted ungulates (Cain et al. 2005). Conversely, a lack of free water can result in reduced food intake and weight loss when low moisture forages are consumed (Lautier et al. 1988). When forage moisture content is low and ambient temperatures are high, it is reasonable to assume that survival and productivity of mule deer could be negatively affected by a lack of free water (Severson and Medina 1983). The highest demand for water in mule deer likely occurs during late gestation and peak lactation. At these times (May-Jul), moisture content of forage is usually relatively high.

Average water turnover rates in mule and black-tailed deer have been reported to vary from 53-104 ml/kg/day with the highest rates occurring during summer and under high ambient temperatures (Knox et al. 1969, Longhurst et al. 1970). Average free water consumption rates for mule deer have been reported to vary from 1.6 to 6.3 quarts/day with the highest rates occurring during the summer months (Hervert and Krausman 1986, Hazam and Krausman 1988). Does may consume more water during summer than bucks to meet lactation demands (Hazam and Krausman 1988).

Deer Movements and Distribution in Relation to Water Sources

Water sources can have a major influence on distribution and movements of mule deer in arid environments (Hervert and Krausman 1986, Boroski and Mossman 1996). Mule deer often seek free liquid water sources even when snow is available or the moisture content of forage is relatively high. In many cases, this behavior is likely due more to preference than need. Mule deer will regularly move up to 1.5 mi to use free water sources (Boroski and Mossman 1996). During summer, does are often distributed closer to water than bucks presumably because of their increased need for water during lactation (Hervert and Krausman 1986, Main and Cobletz 1996).

Potential Benefits of Water Developments

Little information is available on effects of wildlife water developments on deer in the CPE. Water developments in the Southwest Deserts Ecoregion (Lower Sonoran, Mojave, and Chihuahuan deserts) appeared to increase mule deer populations in some cases, indicating water developments can be beneficial in arid areas when adequate forage is available (Rosenstock et al. 1999). Mule deer will readily use water developments, with highest use occurring during hot, dry periods (Rosenstock et al. 2004).

Mule deer distribution in arid environments can be highly influenced by water availability (Hervert and Krausman 1986, Boroski and Mossman 1996). Water developments can be used to more evenly distribute deer across suitable habitat and encourage more optimal use of forage resources (Fig. 37).

Deer will often negotiate hazards such as wildlife-unfriendly fences, residential areas, or highways to get to water. In such cases, the benefits of water developments may have more to do with minimizing risky behavior than

Figure 37. In semi-arid areas with adequate forage and cover, water sources can have a major influence on the distribution and movements of deer. (Photo by Bruce Watkins/CDOW).

Figure 38. Large areas of the CPE are desert or semi-desert habitats that provide little forage or cover for deer. Increased water availability would do little to improve these areas for mule deer. (Photo by Bruce Watkins/CDOW).

Figure 39. Many low elevation areas in the CPE would be unusable by livestock without water developments. (Photo by Bruce Watkins/CDOW).

Figure 40. Big game water catchment and drinking tank in Northern Arizona. (Photo by Luke Thompson/AGFD).

meeting a physiological requirement. Creating additional water sources to minimize hazardous movements by deer may be justified even if water availability, per se, is not a limiting factor.

In some areas, developing low-elevation water sources could result in an increase in deer that remain at low elevations rather than migrating. Sedentary, low-elevation deer herds in the CPE are usually closely associated with water sources.

Potential Negative Impacts of Water Developments
Wildlife water developments can potentially have little benefit for mule deer and other wildlife. Some have even expressed concerns that wildlife water developments can be detrimental (Broyles 1995, Stevens 2004b). Before proceeding with any wildlife water developments, consideration should be given to the cost/benefit potential and the possibility of unintended negative consequences.

The more arid areas of the CPE typically produce little forage for mule deer during the driest months. Food and cover are likely more limiting than water in many of these areas and water developments would provide little benefit (Fig. 38). Available forage resources should always be evaluated before proceeding with water developments

Some water developments could have an adverse affect on deer distribution and movements. For example, the benefit of developing low-elevation water sources may be questionable if migratory deer winter in the same area where water developments could result in more resident deer using limited forage resources year-round.

A major concern with water developments is their potential influence on livestock distribution and habitat use. Cattle are much more dependent on free water sources than deer (NRC 1981). Creating water sources that can also be used by cattle could be counter-productive for deer if the result is heavy cattle grazing in areas that previously received little livestock use (Fig. 39, Mackie 1981, Stevens 2004b).

Broyles (1995) speculated water developments in arid areas might increase predation rates by concentrating prey. Although conclusive evidence is not available, increased predation on deer might occur in some situations where terrain and cover proximate to limited water sources are conducive to predation.

Water quality issues and disease transmission should also be considered with regard to water developments (Broyles 1995). Wildlife water developments often contain warm, stagnant water that can result in the proliferation of algae and other micro-organisms. In addition, some water sources can contain high concentrations of dissolved solids and

potentially toxic minerals. Fortunately, available evidence indicates water quality is not likely to be a major health issue with most wildlife water developments in the Southwest (Rosenstock et al. 2004).

Increased disease transmission can potentially result from contaminated water, increased animal densities in proximity to water, and water-dependent insect vectors. Dead animals can contaminate water sources resulting in subsequent morbidity of other animals that drink the water (Swift et al. 2000). In addition, water developments can provide breeding habitat for biting midges (*Culicoides* spp.) that can be vectors for bluetongue and epizootic hemorrhagic disease. However, *Culicoides* midges have been found to be widely distributed irrespective of water sources and water developments with little mud or silt along the water margin do not appear to provide suitable breeding habitat (Rosenstock et al. 2004).

Additional potential negative effects of wildlife water developments include drowning (Swift et al. 2000), construction of access roads, and increased human activity. Water developments that dry up during the hottest and driest part of the year can be counterproductive by forcing deer to leave established home ranges to seek water during critical periods (Hervert and Krausman 1986).

GUIDELINES
A. Need
Before proceeding with wildlife water developments for mule deer, the need for additional free water sources should be assessed. In winter range areas where snow is typically available most of the winter, water developments would be unlikely to have a positive population effect. Water developments may be beneficial in some arid summer and fall transition ranges or possibly some winter range areas where snow is infrequent. Questions to evaluate free water needs include:
1. Are forage resources adequate to support more deer in the area?
2. Are sources of free water well distributed in relation to typical deer movement patterns (i.e., are free water sources available within 3 mi. of one another)?
3. Could water developments be used to more effectively distribute deer in relation to forage?
4. Could water developments disrupt established migratory patterns of mule deer in the area?
5. Are movements to water causing conflicts? Is it likely these conflicts could be reduced by developing additional water sources?

B. Design and Capacity
There are 4 basic types of big game water developments: 1) artificial collection systems and diversions, 2) natural collection systems, 3) wells, and 4) spring enhancements.

Artificial collection systems use man-made catchment surfaces (e.g., concrete, asphalt, metal, or polyethylene aprons and basins) to collect water and store it in lined basins, tanks, or cisterns (Fig. 40). Diversions bring water from other drainages making use of elevation gradients. Natural collection systems increase water retention and storage capacity of natural drainages through the use of dams and dikes. Wells use devices such windmills or solar pumps to draw water from the ground. Spring enhancements usually involve construction of a reservoir or tank to retain water. There are many different designs for each type (Yoakum et al. 1980, Bleich et al. 1982, Bleich and Weaver 1983, Brigham and Stevenson 1997, AGFD 2004, Rice 2004, USDI 2005a). The most appropriate type and design will depend on a variety of conditions and available water sources.
1. Water developments intended to benefit mule deer and other wildlife should be fenced with wildlife-friendly fencing to restrict use by domestic livestock. Cattle should be excluded using a 3 or 4-strand fence that does not exceed 42" (see Excessive Herbivory, Guidelines, E. Fencing).
2. For artificial collection systems, an underground storage tank that provides water to a separate drinking tank or allows access to only a small part of the storage tank is generally recommended. Drinking tanks should be small (< 4 feet in diameter or length) to prevent drowning and minimize insect breeding and evaporation. The waterline between storage and drinking tanks should be screened to prevent debris from clogging float valves that regulate water flow to the drinking tank.
3. Exposed water sources should be shaded whenever possible to reduce evaporation and algae growth.
4. Larger water developments should be designed to allow easy egress to help prevent drowning (e.g., gentle slopes or escape ramps).
5. Water developments should be designed and maintained to reduce the amount of silt and mud at the water margin to reduce breeding habitat for *Culicoides* midges.
6. Whenever possible, storage reservoirs and tanks should be designed to allow periodic flushing.
7. Water developments should not be allowed to go dry during periods they would be needed the most. Developments should be designed with sufficient catchment surface and storage capacity to ensure that water is available during critical periods. Assuming 100% collection efficiency, each square foot of catchment surface will collect 0.623 gallons of water for each inch of rain.

C. Other Considerations
1. Wildlife water developments require regular maintenance to ensure proper function, to remove silt, algal mats, and other debris, and to fix problems associated with wear and tear and vandalism. There should be a commitment

to provide long-term maintenance before developing water sources for deer and other wildlife.

2. Water quality should be tested if problems are suspected (e.g., dead animals in the vicinity, foul smell, little evidence of use). Rosenstock et al. (2004) can be used as a guide for appropriate water quality tests.
3. Whenever possible, water developments should be placed in areas with good visibility and low relief to minimize predation risks.
4. Water developments should be placed where deer and other wildlife will not need to cross busy roads or negotiate other hazards to gain access.
5. Access roads into water developments should be blocked to prevent unnecessary human activity.

Human Encroachment

Background

Human activity can impact habitat suitability in 3 ways: displacing wildlife through habitat occupation (e.g., construction of buildings), reducing habitat suitability by altering physical characteristics of that habitat (e.g., habitat damage resulting from off highway vehicle use), or displacing wildlife by altering wildlife perception of the suitability of habitat through other than physical alteration (e.g., noise, activity).

Issues and Concerns

Displacement by Occupation

Wildlife habitat is appealing in many ways to humans. Because of the appealing nature of landscapes occupied by wildlife, humans are increasingly moving to these habitats to live (Fig. 41). The occupation of these habitats brings with it construction of homes, fencing, roadways, agriculture, and supporting infrastructures, such as communities, stores, and health facilities. People who occupy these areas frequently bring domestic dogs and livestock that may jeopardize wildlife through direct mortality or disease transmission. These communities are often located in habitats that fill critical wildlife needs during periods of migration or winter stress. When people move to habitats that contain wildlife, the resultant development destroys many of the features that initially attracted people to the area. This is the greatest impact of human disturbance on wildlife populations. During the mid 1990s alone, development occupied 5.4 million acres of open space in the West (Lutz et al. 2003).

However, human occupation may provide some advantages to local wildlife populations (Tucker et al. 2004). Wildlife in some urban areas may have more water from artificial sites (e.g., pools, ponds) and enhanced forage (e.g., lawns, plantings, golf courses, agricultural fields) than in surrounding areas. The reduction of predators in these habitats can also reduce mortality for wildlife that inhabits the area.

Enhanced forage conditions and decreased predation may result in unhealthy densities of wildlife that will be susceptible to diseases or might actually increase the probability that predators will move into the urban area from surrounding areas to prey on naïve wildlife. Ultimately, these predators may prey on domestic pets as well. Incidences of predators preying on humans in these environments are increasing (Beier 1991, Torres et al. 1996).

A major concern for mule deer is encroachment upon, and development within, important habitats. A primary example of this is the impact of land development on winter range. Improved forage and decreased predation notwithstanding, increased housing density can result in decreased mule deer abundance (Vogel 1989, Fig. 42). Mineral exploration-extraction or urban development can preclude use of winter ranges that are critically important to migratory deer herds during severe winters (Lutz et al. 2003). Road development can limit mule deer access to important habitat as well. Agricultural developments often make habitats more desirable to mule deer; however, these

Figure 41. Humans, along with their pets and livestock, continue to move into mule deer habitat. (Photo by Bruce Watkins/CDOW).

same developments sometimes include efforts by those managing agricultural lands to limit wildlife use of the area (Fig. 43).

Reduction of Habitat Suitability

Human activity has the ability to alter habitat suitability through direct alteration of habitat characteristics, thereby influencing habitat quality. Improper use of off highway vehicles (OHVs) can alter habitat characteristics through destruction of vegetation, compacting soil, and increasing erosion. Perry and Overly (1977) found roads through meadow habitats reduced deer use, whereas roads through forested habitat had less effect.

The most obvious negative impact on habitat suitability is elimination of linkages between important habitats. These impacts may be the result of actual development or road proliferation and improvement.

Recognition and understanding of the impact of highways on wildlife populations have increased dramatically in the past decade (Forman et al. 2003). In fact, highway-associated impacts have been characterized as being among the most prevalent and widespread forces affecting natural ecosystems and habitats in the U.S. (Noss and Cooperrider 1994, Trombulak and Frissell 2000, Farrell et al. 2002). These impacts are especially severe in the western states where rapid human population growth and development are occurring at a time when deer populations are depressed. Human population growth has resulted in increased traffic volume on highways, upgrading of existing highways, and construction of new highways, all serving to further exacerbate highway impact to mule deer and other wildlife.

Direct loss of deer and other wildlife due to collisions with motor vehicles is a substantial source of mortality affecting populations. Romin and Bissonette (1996) conservatively estimated > 500,000 deer of all species are killed each year in the U.S.; Schwabe and Schuhmann (2002) estimated this loss at 700,000 deer/year, whereas Conover et al. (1995) estimated > 1.5 million deer-vehicle collisions occur annually. In addition to effects on populations, many human injuries and loss of life occur with deer-wildlife collisions annually. Conover et al. (1995) estimated that collisions involved 29,000 injuries and 200 deaths to humans annually. There is substantial loss of recreational opportunity and revenue associated with deer hunting, and damage to property from collisions is tremendous (Romin and Bissionette 1996, Reed et al. 1982). Deer-vehicle collisions are particularly a problem on winter ranges where deer are concentrated and along migration routes to and from winter ranges (e.g., Gordon and Anderson 2003). The problem is further compounded by the dramatic explosion of human residential and other development within mule

deer winter ranges in the Intermountain West. Additionally, roadways fragment habitat and impede movements for migratory herds (Lutz et al. 2003). Some highway transportation departments have used overpasses and underpasses for wildlife to mitigate highways as impediments. Recently, temporary warning signs have been demonstrated to be effective in reducing collisions during short duration migration events (Sullivan et al. 2004). Quick fixes such as Swareflex reflectors have often proven ineffective (Farrell et al. 2002, Fig. 44).

Of all the impacts associated with highways, the most important to mule deer and other wildlife species is

Figure 42. Mule deer often use residential areas that supplant their historic habitat. Although ornamental forage and refuge from hunting and predators can attract deer to these areas, there is often a greater risk of vehicle accidents, harassment, and un-natural hazards such as fences, Christmas lights, and volleyball nets. (Photo by Jerrie McKee/CDOW).

Figure 43. Irrigated agricultural lands often have a positive effect on mule deer in the CPE but can lead to game damage conflicts. (Photo by Randy Hampton/CDOW).

attributable to barrier and fragmentation effects (Noss and Cooperrider 1994, Forman and Alexander 1998, Forman 2000, Forman et al. 2003). Highways alone act as barriers to animals moving freely between seasonal ranges and to special or vital habitat areas. This barrier effect fragments habitats and populations, reduces genetic interchange among populations or herds, and limits dispersal of young; all serve to ultimately disrupt the processes that maintain viable mule deer herds and populations. Furthermore, effects of long-term fragmentation and isolation render populations more vulnerable to the influences of stochastic events, and may lead to extirpations of localized or restricted populations of mule deer. These effects are greatly exacerbated when impermeable, ungulate-proof high fences are used to prevent deer from entering roadways (Fig. 45). Other human impacts directly tied to increased roads

Figure 44. Swareflex reflectors placed along a Colorado highway appeared to do little to reduce deer-vehicle collisions. (Photo by Bruce Watkins/CDOW).

Figure 45. Earthen ramps are often used in the CPE to allow trapped deer and elk to exit roadways protected with high fences. Although high fences are the most effective way to reduce deer-vehicle collisions, they can be very detrimental to migrating deer populations if adequate passage structures are not provided. (Photo by Bruce Watkins/CDOW).

include increased poaching, unregulated off-highway travel, and ignition of wildfires. Highways also serve as corridors for dispersal of invasive plants that degrade habitats (White and Ernst 2003).

In the past, efforts to address highway impact were typically approached as single-species mitigation measures (Reed et al. 1975). Today, the focus is more on preserving ecosystem integrity and landscape connectivity benefiting multiple species (Clevenger and Waltho 2000). Farrell et al. (2002) provide an excellent synopsis of strategies to address ungulate-highway conflicts.

Several states have made major commitments to early multi-disciplinary planning, including Washington (Quan and Teachout 2003), Colorado (Wostl 2003), and Oregon; some receive funding for dedicated personnel within resource agencies to facilitate highway planning. Florida's Internet-based environmental screening tool is currently a national model for integrated planning (Roaza 2003). To be most effective, managers must provide scientifically credible information to support recommendations, identifying important linkage areas, special habitats, and wildlife-vehicle collision hotspots (Endries et al. 2003).

There is a major need for states to complete large-scale connectivity and linkage analyses to identify priority areas for protection or enhancement in association with highway planning and construction. Such large-scale connectivity analyses, already accomplished in southern California (Ng et al. 2004), New Mexico, Arizona, and Colorado, serve as a foundation for improved highway planning to address wildlife permeability needs. More refined analyses of wildlife connectivity needs, particularly to identify locations for passage structures are of tremendous benefit, and run the gamut from relatively simple GIS-based "rapid assessment" of linkage needs (Reudiger and Lloyd 2003) to more complex modeling of wildlife permeability (Singleton et al. 2002). Strategies for maintaining connectivity may include land acquisition (Neal et al. 2003) or conservation easements.

Structures designed to promote wildlife permeability across highways are increasing throughout North America, especially large bridged structures (e.g., underpasses or overpasses) designed specifically for ungulate and large predator passage (Clevenger and Waltho 2000, 2003). Whereas early passage structures were approached as single-species mitigation measures (Reed et al. 1975), their use today is focused more on preserving ecosystem integrity and landscape connectivity benefiting many species (Clevenger and Waltho 2000, 2003). Transportation agencies are increasingly receptive to integrating passage structures into new or upgraded highway construction to address both highway safety and ecological needs (Farrell et al. 2002).

However, there is increasing expectation that such structures will indeed yield benefit to multiple species and enhance connectivity (Clevenger and Waltho 2000), and that scientifically sound monitoring and evaluation of wildlife response will occur to improve future passage structure effectiveness (Clevenger and Waltho 2003, Hardy et al. 2003).

Displacement through Disturbance
Research has documented that wildlife modify their behavior to avoid activities that they perceive as threatening, such as avoidance of higher traffic roads by elk. However, this avoidance is generally temporary and once removed, wildlife return to their prior routine. Extensive research has failed to document population level responses (e.g., decreased fitness, recruitment, or conception) as a direct result of disturbance. White-tailed deer in the eastern U.S. have acclimated to relatively high densities of people and disturbance. Even direct and frequent disturbance during breeding season has not yielded any population level responses (Bristow 1998).

Information regarding the response of deer to roads and vehicular traffic is limited. Perry and Overly (1977) found that main roads had the greatest impact on mule deer, and primitive roads the least impact. Proximity to roads and trails has a greater correlation with deer distribution than does crude calculations of mean road densities (Johnson et al. 2000). Off road recreation is increasing rapidly on public lands. The U.S. Forest Service estimates that OHV use has increased sevenfold during the past 20 years (Wisdom et al. 2005). Use of OHVs has a greater impact on avoidance behavior than does hiking or horseback riding, especially for elk (Wisdom et al. 2005), especially for elk.

GUIDELINES

A. Planning and Coordination
1. Develop and maintain interagency coordination in land planning activities to protect important habitats.
2. Land and wildlife management agencies should play a proactive role in city and county planning, zoning, and development.
3. Identify important habitats, seasonal use areas, migration routes, and important populations of mule deer.
4. Coordinate with agricultural producers to consider wildlife needs in the selection of crops, locations, and rotations. Identify acceptable wildlife use.
5. Analyze linkages and connectivity of habitats to identify likely areas for impact hazards as new roads are developed or altered for higher speed and greater volume traffic.

B. Minimizing Negative Effects of Human Encroachment
1. Develop consistent regulations for OHV use.
2. Maintain interagency coordination in the enforcement of OHV regulations.
3. Designate areas where vehicles may be legally operated off road.
4. Encourage use of native vegetation in landscaping human developments to minimize loss of usable habitat.
5. Examine records of road-killed deer locations to determine where major impact areas exist and evaluate the need for wildlife passage structures.
6. Construct overpasses and underpasses along wildlife corridors known to be mule deer travel routes.
7. Monitor activities that may unduly stress deer at important times of the year. Reduce-regulate disturbance if deemed detrimental.
8. Enhance alternate habitats to mitigate for habitat loss, including components like water availability.
9. Provide ungulate-proof fencing to direct wildlife to right-of-way passage structures or away from areas of high deer-vehicle collisions.
10. Encourage the use of wildlife-friendly fence (permeable) in appropriate areas to minimize habitat fragmentation.
11. Coordinate with agencies to provide private landowner incentives, such as conservation easements, for protecting habitat.

C. Wildlife Passage Structures
1. To maximize use by deer and other wildlife, passage structures should be located away from areas of high human activity and disturbance. For established passage structures in place > 10 years, Clevenger and Waltho (2000) found that structural design characteristics were of secondary influence to ungulate use compared to human activity.
2. Locate passage structures in proximity to existing or traditional travel corridors or routes (Singer and Doherty 1985, Bruinderink and Hazebroek 1996), and in proximity to natural habitat (Foster and Humphrey 1995, Servheen et al. 2003, Ng et al. 2004).
3. Spacing between structures is dependent on local factors (e.g., known deer crossing locations, deer-vehicle collision "hotspots," deer densities adjacent to highways, proximity to important habitats).
4. Where appropriate and available, use models and other tools to assist in location of passage structures (Clevenger et al. 2002, Barnum 2003, Claar et al. 2003).
5. Passage structures should be designed to maximize structural openness (Reed 1981, Foster and Humphrey 1995, Clevenger and Waltho 2003, Ng et al. 2004, Reudiger 2001). The openness ratio (width x height/length) should be > 0.6 (Reed et al. 1979), and preferably > 0.8 (Gordon and Anderson 2003). Reductions in underpass width influence mule deer passage more than height (Gordon and Anderson 2003, Clevenger and Waltho 2000).
6. Underpasses designed specifically for mule deer should be at least 20 feet wide and 8 feet high (Gordon and

Anderson 2003, Forman et al. 2003). Gordon and Anderson (2003) and Foster and Humphrey (1995) stressed the importance of animals being able to see the horizon as they negotiate underpasses. Mule deer make minimal use of small passage structures such as livestock and machinery box-culverts (Gordon and Anderson 2003, Ng et al. 2004).

7. More natural conditions within underpass structures (e.g., earthen sides and naturally vegetated) has been found to promote use by ungulates (Dodd et al. 2006). In Banff National Park, Alberta, deer strongly preferred (10x more use) crossing at vegetated overpasses compared to open-span bridged underpasses (Forman et al. 2003).

8. Use ungulate-proof fencing in conjunction with passage structures to reduce deer-vehicle collisions (Clevenger et al. 2001, Farrell et al. 2002). Caution should be exercised when applying extensive ungulate-proof fencing without sufficient passage structures to avoid creating barriers to free deer movement.

9. Where possible, fences should be tied into existing natural passage barriers (e.g., large cut slopes, canyons; Puglisi et al. 1974).

10. When fencing is not appropriate to reduce deer-vehicle collisions, alternatives include enhanced signage to alert motorists (Farrell et al. 2002), Swareflex reflectors (with generally inconclusive results [Farrell et al. 2002]), deer crosswalks (Lehnert and Bissonette 1997), and electronic roadway animal detection systems (RADS, Huijser and McGowen 2003).

ENERGY AND MINERAL DEVELOPMENT

BACKGROUND

Energy consumption and production continues to be a major part of our nation's overall energy policy. According to the National Energy Policy (2001), "…if energy production increases at the same rate as during the last decade our projected energy needs will far outstrip expected levels of production. This imbalance, if allowed to continue, will inevitably undermine our economy, our standard of living, and our national security". Even as recent as 2006, President Bush stated, "America is addicted to oil…" He has set a new national goal of replacing more than 75% of the United States' oil imports from the Middle East by 2025.

As pressure mounts to explore new energy initiatives and develop more areas (i.e., Arctic National Wildlife Refuge, Raton Basin, San Juan Basin, Uinta-Piceance Basin, etc.), careful attention must be given to how this industry can expand to satisfy increasing energy demands. A national debate must focus on identifying practical means of moving forward with energy independence while at the same time recognizing the importance of a healthy environment in terms of the diversity of economies, recreation, and inherent aesthetics it supports and provides.

Some success with providing protection to wildlife habitat from energy development occurred in late 2006 when President Bush signed legislation to protect the Valle Vidal in north-central New Mexico from drilling and mining.

Figure 46. The Valle Vidal (left), now protected from drilling, borders the Vermejo Park Ranch (right) where coal bed methane is in production. (Photo courtesy of New Mexico Department of Game and Fish [NMDGF]).

The Valle Vidal, a 102,000 acre tract within the Carson National Forest, is well known for its aesthetic landscapes. It is also known to be important year-round elk range. Until this recent action, portions of the Valle Vidal were being considered for coal bed methane (CBM) development similar to what is occurring within the neighboring Vermejo Park Ranch (Fig. 46).

Because much of the CPE is comprised of high elevation forests and low elevation shrub and grasslands, mule deer are dependent upon separate ranges for summer and winter seasons. Migratory routes are necessary for transitioning between these critical areas. Energy and mineral development activities not only remove productive habitat from these ranges, but also create barriers preventing migration and use of remaining habitats.

Coincidentally, much of the CPE contains significant accumulations of natural gas and coal deposits. CBM, methane gas trapped within coal beds, is becoming a predominant energy alternative within the CPE. Reserves of CBM can be found throughout much of the Rocky Mountains and the CPE (Fig. 47). Unfortunately, development and extraction activities associated with CBM tend to be aggressive and therefore have potential for more profound and long-term impacts on the environment.

Tessmann et al. (2004) reported that exploration and extraction of non-renewable oil and gas resources has and continues to cause a range of adverse effects. All disturbances to the landscape constitute an impact at some level. The severity of the impact to mule deer depends upon the amount and intensity of the disturbance, specific locations and arrangements of disturbance, and ecological importance of the habitats affected. Small, isolated disturbances within non-limiting habitats are of minor consequence within most ecosystems. However, larger-scale developments within habitats limiting the abundance and productivity of mule deer are of significant concern to managers because such impacts cannot be relieved or absorbed by surrounding, unaltered habitats. Impacts, both direct and indirect, associated with energy and mineral development has the potential to affect ungulate population dynamics, especially when impacts are concentrated on winter ranges (Sawyer et al. 2002).

For the purpose of this discussion, oil and gas development includes those activities used to extract all hydro-carbon compounds such as natural gas, crude oil, CBM, and oil shale. Many industries depend upon other materials (e.g., copper, uranium, vanadium) for their products and services and extracting these raw materials can have the same effects on wildlife and the environment as oil and gas development. Therefore, although the issues and concerns as well as guidelines discussed in this section focus predominantly upon oil and gas development, in most circumstances they are relevant and applicable to mineral extraction activities.
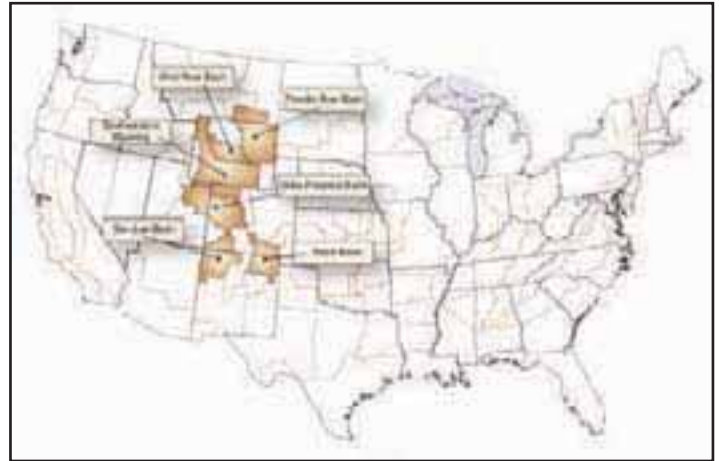

Figure 47. Coal Bed Methane locations within the Rocky Mountain region. (Map courtesy of USGS).

Figure 48. A typical gas well; 1-2 acre footprint. (Photo courtesy of NMDGF).

Figure 49. This compressor station is an example of other facilities that directly remove habitat. (Photo courtesy of NMDGF).

Table 4. Categories of impact on mule deer from energy and mineral extraction activities (Tessman et al. 2004)

MODERATE	HIGH	EXTREME
Impacts can be minimized or avoided through effective management practices & habitat treatments	Impacts are increasingly difficult to mitigate and may not be completely offset by management and habitat treatments	Habitat function is substantially impaired and cannot generally be recovered through management or habitat treatments
1-4 wells and ≤20 acres disturbance per section	5-16 wells and 20-80 acres disturbance per section	> 16 wells or > 80 acres disturbance per section

Impact Thresholds

Impact thresholds, as defined by Tessman et al. (2004), are levels of development or disturbance that impair key habitat functions by directly eliminating habitat, disrupting access to habitat, or causing avoidance and stress. For this discussion, impact thresholds are based upon 2 quantitative measures – density of well locations (pads) and cumulative disturbance per section (640 acres). Density of well locations has bearing on the intensity of disturbances associated with oil and gas field operations whereas the cumulative area of disturbance measures direct loss of habitat.

In addition to well pads, a typical oil and gas field includes many other facilities and associated activities that affect wildlife: roads, tanks, equipment staging areas, compressor stations, shops, pipelines, power supplies, traffic, human activity, etc. (Figs. 48 and 49). Densities of well pads can be viewed as a general index to well field development and activities. However, thresholds based upon well pad densities and cumulative acreage alone may under-represent the actual level of disturbance.

Measures to reduce impacts should be considered when well densities exceed 4 wells per section or when road density exceeds 3 miles of road per section (USDI 1999). The following describe and define relative degrees of impact (Table 4).

Moderate Impact
Habitat effectiveness is reduced within a zone surrounding each well, facility, and road corridor through human presence, vehicle traffic, and equipment activity.

High Impact
At this range of development, impact zones surrounding each well pad, facility, and road corridor begin to overlap, thereby reducing habitat effectiveness over much larger, contiguous areas. Human, equipment, and vehicular activity, and noise and dust are also more frequent and intensive. This amount of development will impair the ability of animals to use critical areas (winter range, fawning grounds, etc.) and impacts will be much more

difficult to mitigate. Fully mitigating impacts caused by higher well densities may not be possible, particularly by developing habitat treatments on site. Habitat treatments will then generally be located in areas near, rather than within well fields to maintain the function and effectiveness of critical areas.

Extreme Impact
Function and effectiveness of habitat would be severely compromised (Fig. 50). With CBM, a single well may only be capable of removing a small amount of the gas contained within the coal bed. Consequently, many hundreds to thousands of wells may be required to recover the available gas (USDI 2005b). The long-term consequences are continued fragmentation and disintegration of habitat leading to decreased survival, productivity, and ultimately, loss of carrying capacity for the herd. This will result in a loss of ecological functions, recreation, opportunity, and income to the economy. An additional consequence may include permanent loss of migration memory from large segments of unique, migratory mule deer herds.

Impacts to mule deer from energy and mineral development can be divided into the following general categories: 1) direct loss of habitat; 2) physiological stresses; 3) disturbance and displacement; 4) habitat fragmentation and isolation; and 5) other secondary effects (Tessman et al. 2004). Each of these, alone or in conjunction with others, has the potential to significantly influence whether deer can maintain some reasonable existence in the developed area or abandon it altogether.

ISSUES AND CONCERNS

Direct Loss of Habitat
Direct loss of habitat results primarily from construction and production phases of development. The presence of well pads, open pits, roads, pipelines, compressor stations, and out buildings directly removes habitat from use (Fig. 51). Production activities require pervasive infrastructure and depending upon scale, density, and arrangement of the developed area, collateral loss of habitat could be extensive (USDI 1999). As an example, within the Big Piney-LeBarge

oil and gas field in Wyoming, the actual physical area of structures, roads, pipelines, pads, etc. covers approximately 7 square miles. However, the entire 166 square mile landscape is within 0.5 mile of a road, and 160 square miles (97% of landscape) is within 0.25 mile of a road or other structure (Stalling 2003). Furthermore, Bartis et al. (2005) reported that oil shale development has the likelihood of removing a portion of land over the Green River Formation, withdrawing it from current uses, with possible permanent topographic changes and impacts on flora and fauna.

Generally, while 50% of a disturbed area could be minimally reclaimed within a 3-5 year period after construction, development of a fully productive habitat (proper species composition, diversity, and age) could require up to 20 years (Fig. 52). The remaining 50%, which constitutes the working surfaces of roads, well pads, and other facilities, could represent an even greater long-term habitat loss (USDI 1999). Reclamation of sagebrush communities is tenuous at best because success is highly dependent upon amount and timing of moisture; reseeding is usually required if initial efforts are conducted > 1 year post-disturbance.

Physiological Stress

Physiological stresses occur when energy expenditures by an animal are increased due to alarm or avoidance movements. These are generally attributed to interactions with humans or activities associated with human presence (e.g., traffic, noise, pets). During winter months, stress-related energy expenditure can be particularly important because mule deer are already in a negative energy balance. In addition, stress can be detrimental during other critical periods such as gestation and lactation. Kuck et al. (1985) suggested, in their simulated mine disturbance experiment, that increased energy costs of movement, escape, and stress caused by frequent and unpredictable disturbance may have been detrimental to elk calf growth. An EIS on oil and gas development in the Glenwood Springs Resource Area in Colorado determined these impacts could ultimately have population effects through reduced production, survival, and recruitment (USDI 1999).

Disturbance and Displacement

Increased travel by humans within the area, equipment operation, vehicle traffic, and noise related to wells and compressor stations are primary factors leading to avoidance of the developed area by wildlife. These avoidance responses by mule deer (indirect habitat loss) extend the influence of each well pad, road, and facility to surrounding areas. Zones of negative response can reach a 0.25-mile radius for mule deer (Freddy et al. 1986).

Significant differences in elk distribution between construction and non-construction periods were observed

Figure 50. The Rosa gas field in northwestern New Mexico--extreme impact. (Photo courtesy of NMDGF).

Figure 51. Drilling rig; construction phase. (Photo courtesy of NMDGF).

Figure 52. Successful reclamation of open pit coal mine in northeastern New Mexico; 15-20 years of reclamation effort. (Photo courtesy of NMDGF).

Figure 53. Improperly fenced collection pits pose risks to wildlife in search of water. (Photo courtesy of CDOW).

Figure 54. Erosion resulting from increased use and runoff. (Photo courtesy of CDOW).

by Johnson et al. (1990) in the Snider Basin calving area of western Wyoming. Elk moved away from construction activities during calving season, but returned the following year when no construction activities occurred. Furthermore, these elk not only avoided areas near drill sites, but also areas visible from access routes.

During all phases, roads tend to be of significant concern because they often remain open to unregulated use. Open roads contribute to noise and increased human presence within the development area. Rost and Bailey (1979) found an inverse relationship to habitat use by deer and elk with distance to roads. This displacement can result in under-use of the habitat near disturbances while overuse may occur in other locations. This has the added potential for creating depredation problems with nearby agricultural properties. Added consequences from human presence include, but are not limited to, mortality and injury due to vehicle collisions, illegal hunting, and harassment

Habitat Fragmentation and Isolation
Associated with displacement is the greater impact of fragmentation. Meffe et al. (1997) suggested the largest single threat to biological diversity is the outright destruction of habitat along with habitat alteration and fragmentation of large habitats into smaller patches. As stated earlier, road networks have a cumulative effect when considering total amount of habitat lost. This is especially evident in their contribution to habitat fragmentation. USDI (1997) stated: "As road density increases, the influence on habitat effectiveness increases exponentially, such that at road densities of 3 miles per square mile, habitat effectiveness is reduced by about 30 percent."

Should development occur within or proximate to migration corridors, isolation may result. Isolation could lead to adverse genetic effects such as inbreeding depression and decreased genetic diversity. Without an ability to move into or from areas critical to normal needs or life stages (e.g., fawning areas, winter range) abandonment could ultimately result.

Habitat fragmentation creates landscapes made of altered habitats or developed areas fundamentally different from those shaped by natural disturbances that species have adapted to over evolutionary time (Noss and Cooperrider 1994). These changes likely manifest themselves as changes in vegetative composition, often to weedy and invasive species. This, in turn, changes the type and quality of the food base as well as habitat structure. Increased interface between developed and undeveloped areas often results in reduced forage quality and security cover, potentially increasing deer susceptibility to predation.

Use of migration corridors also depends on factors such as aspect, slope, and weather. Therefore when planning

developments, it is critical to consider impacts to these corridors and how to mitigate them to facilitate migration of mule deer (Merrill et al. 1994). Flexibility in movement across ranges can be ultimately reflected in the survival and productivity of a deer population and likely enhances their ability to recover from population declines.

Secondary Effects
Secondary effects may be as significant as those direct effects described above. Activities associated with the support and service industries linked to development can aggravate adverse impacts. These impacts can be similar to those that occur during construction and operations–only intensified. Vehicular traffic to support operations would likely increase significantly which may result in increased deer-vehicle collisions. Additional human presence from increased support industries as well as community expansion will contribute to human-wildlife interactions and declines in mule deer habitat availability and quality.

Roads, pipelines, and transmission corridors not only directly remove habitat, but also have the potential to contaminate ground and surface water supplies. Noxious weeds can infiltrate roadside impact zones and bring negative impacts such as non-native bacteria, viruses, insect pests, and chemical defense compounds with toxic or allergenic properties (NMDGF 2004).

Activities occurring at the well site (e.g., drilling, pumping) or as a result of transporting the product to other destinations via pipeline or vehicle, may lead to release of a variety of toxic compounds. These compounds are common by-products and pose serious health risks not only to employees, but also the environment and the wildlife inhabiting the locality (Fig. 53).

Water quality is a major concern with both CBM and oil shale development. During CBM extraction, water is pumped to the surface in large volumes to release the gas trapped in coal seams. Quantities of water removed from wells depend upon the physical nature of the gas reservoir and depths drilled. In portions of the Fruitland formation of the San Juan Basin, average daily water production from a single CBM well was 250 barrels (10,500 gallons), whereas in portions of the Fort Union Formation of the Powder River Basin, between 200 to 500 barrels (8,400 to 21,000 gallons) were produced daily. Deeper wells produced > 1,000 barrels (42,000 gallons, USDI 2003).

Water produced during CBM extraction can be highly saline and, as a result, may not be suitable for agriculture and wildlife (Elcock et al. 1999). With respect to oil shale operations, surface water contamination from waste piles may result from leaching of salts and toxins from spent shale (Bartis et al. 2005). Furthermore, with removal of

so much groundwater, hydrology of local natural springs can be seriously impacted.

Erosion of sediment from roads and pipeline corridors increases surface runoff into watercourses, reduces infiltration, lowers water tables, and results in lower rangeland productivity (Fig. 54). These problems will increase if some of the recommendations outlined in the National Energy Policy, such as building an additional 38,000 miles of new gas pipelines, are implemented (National Energy Policy 2001).

All these events can reduce the amount of area available to mule deer and other wildlife. The potential exists for rendering an area useless to wildlife for an indeterminable amount of time unless careful consideration is given to planning and implementing quality mitigation and reclamation programs.

GUIDELINES
To minimize impacts of energy and mineral development activities on mule deer and their habitat, several recommendations are provided for consideration and implementation. These recommendations are compiled from a number of sources and support the principles for prudent and responsible development as stated in the National Energy Policy (2001). When energy development is proposed, the federal government has the dual responsibilities of facilitating such energy development and conserving our natural resource legacy.

A. Pre-planning and Scoping
1. Consult appropriate state and federal wildlife agencies during pre-planning exercises.
2. Design configurations of oil and gas development to avoid or reduce unnecessary disturbances, wildlife conflicts, and habitat impacts. Where possible, coordinate planning among companies operating in the same oil and gas field.
3. Identify important, sensitive, or unique habitats and wildlife in the area. To the extent feasible, incorporate mitigation practices that minimize impacts to these habitats and resources.
4. Where practical, implement timing limitation stipulations that minimize or prohibit activities during certain, critical portions of the year (e.g., when deer are on winter range, fawning periods).
5. Prepare a water management plan in those regions and for those operations that generate surplus quantities of water of questionable quality (e.g., CBM).
6. Plan the pattern and rate of development to avoid the most important habitats and generally reduce the extent and severity of impacts. To the extent practicable, implement phased development in smaller increments.
7. Cluster drill pads, roads, and facilities in specific, low-impact areas.

Figure 55. Well pads located on prominent sites contribute to increased visual and auditory impacts. (Photo courtesy of NMDGF).

Figure 56. Preferable locations for buildings and other facilities. (Photo courtesy of NMDGF).

Figure 57. Close roads to reduce unnecessary access. (Photo courtesy of NMDGF).

8. Locate drill pads, roads, and facilities below ridgelines or behind topographic features, where possible, to minimize visual and auditory effects, but away from streams, drainages, and riparian areas, as well as important sources of forage, cover, and habitats important to different life cycle events (e.g., reproduction, winter, parturition, and rearing) (Figs. 55 and 56).
9. Develop and implement "Best Management Practices" for all energy development activities.

B. Roads
1. Use existing roads and 2-tracks if they are sufficient and not within environmentally sensitive areas.
2. If new roads are needed, close existing roads that provide access to the same area but impact important mule deer habitat (Fig. 57).
3. Construct the minimum number and length of roads necessary.
4. Use common roads to the extent practical.
5. Coordinate road construction and use among companies operating in the same oil and gas field.
6. Design roads to an appropriate standard no higher than necessary to accommodate their intended purpose.
7. Design roads with adequate structures or features to prohibit or discourage vehicles from leaving the roads.
8. Roads should be gated and property fenced to preclude unauthorized use by vehicles.
9. Roads should be closed and reclaimed as soon as they are no longer needed.

C. Wells
1. Drill multiple wells from the same pad using directional (horizontal) drilling technologies (Fig. 58).
2. Disturb the minimum area (footprint) necessary to efficiently drill and operate a well.
3. Where soil type is conducive, consider use of oak mats to cover the well pad during construction and operation (Figs. 59 and 60). This technique is designed to minimize impacts to soil and plant resources without the need to remove, and eventually replace, top-soil. Oak mats are constructed in sections from three layers of 2 inch by 10 inch oak boards arranged perpendicular with 1inch spacing between the boards. With this configuration, precipitation is capable of reaching the soil underneath. Each mat section is 8 feet wide, 12 feet long and 6 inches thick and connected to neighboring mats using the tongue-and-groove method. This technique is undergoing evaluation in the Jonah Field in Pinedale, Wyoming (EnCana 2005).

D. Ancillary Facilities
1. Use existing utility, road, and pipeline corridors to the extent feasible.
2. Bury all power lines in or adjacent to roads.

E. Noise
1. Minimize noise to the extent possible. All compressors, vehicles, and other sources of noise should be equipped with effective mufflers or noise suppression systems (e.g., "hospital mufflers").
2. Whenever possible, use electric power instead of diesel to power compression equipment.
3. Use topography to conceal or hide facilities from areas of known importance to mule deer.

F. Traffic
1. Develop a travel plan that minimizes the amount of vehicular traffic needed to monitor and maintain wells and other facilities.
2. Limit traffic to the extent possible during high wildlife use hours (within 3 hours of sunrise and sunset).
3. Use pipelines to transport condensates off site.
4. Transmit instrumentation readings from remote monitoring stations to reduce maintenance traffic (Fig. 61).
5. Post speed limits on all access and maintenance roads to reduce wildlife collisions and limit dust (30-40 mph is adequate in most cases).

G. Human Activity
1. Employees should be instructed to avoid walking away from vehicles or facilities into areas used by wildlife, especially during winter months.
2. Institute a corporate-funded reward program for information leading to conviction of poachers, especially on winter range.

H. Pollutants, Toxic Substances, Fugitive Dust, Erosion, and Sedimentation
1. Avoid exposing or dumping hydrocarbon products on the surface. Oil pits should not be used, but if absolutely necessary, they should be enclosed in netting and small-mesh fence. All netting and fence must be maintained and kept in serviceable condition.
2. Produced water should not be pumped onto the surface except when water quality standards for wildlife and livestock are met.
3. Produced water should not be pumped onto the surface within big game crucial ranges. However, produced water of suitable quality may be used for supplemental irrigation to improve reclamation success.
4. Re-injection of water into CBM sites should be considered when water quality is of concern.
5. Hydrogen sulfide should not be released into the environment.
6. Use dust abatement procedures including reduced speed limits, and application of an environmentally compatible chemical retardant or suitable quality water.

I. Monitoring and Environmental Response
1. Monitor conditions or events that may indicate environmental problems (e.g., water quality in nearby rivers, streams, and wells). Such conditions or events can include any significant chemical spill or leak, detection of multiple wildlife mortalities, sections of roads with frequent and recurrent wildlife collisions, poaching and harassment incidents, severe erosion into tributary drainages, migration impediments, wildlife entrapment, sick or injured wildlife, or other unusual observations.
2. Immediately report observations of potential wildlife problems to the state wildlife agency and, when applicable, federal agencies such as U.S. Fish and Wildlife Service or Environmental Protection Agency.
3. Apply GIS technologies to monitor the extent of disturbance annually and document the progression and footprint of disturbances. Release compilations of this information to state and federal resource agencies at least annually.

J. Research and Special Studies
1. Where questions or uncertainties exist about the degree of impact to specific resources, or the effectiveness of

Figure 58. Well site with 2 wells. (Photo courtesy of NMDGF).

Figure 59. Oak mats being placed directly over drill site to protect top soil. (Photo courtesy of EnCana Oil and Gas).

mitigation, industries and companies should fund special studies to collect data for evaluation and documentation.

Figure 60. Drilling rig positioned on oak mats. (Photo courtesy of EnCana Oil and Gas).

Figure 61. Remote monitoring reduces traffic around well sites. (Photo courtesy of NMDGF).

K. Noxious Weeds
1. Control noxious and invasive plants that appear along roads, on well pads, or adjacent to other facilities.
2. Clean and sanitize all equipment brought in from other regions. Seeds and propagules of noxious plants are commonly imported by equipment.
3. Request that employees clean mud from boots and work shoes before traveling to the work site to prevent importation of noxious weeds.

L. Interim Reclamation
1. Establish effective, interim reclamation on all surfaces disturbed throughout the operational phase of the well field.
2. Where practical, salvage topsoil from all construction and re-apply during interim reclamation.
3. Approved mulch application should be used in sensitive areas (e.g., dry, sandy, steep slopes).
4. A variety of native grasses and forbs should be used. Non-native vegetation is unacceptable for any purpose, including surface stabilization. Continue to monitor and treat reclaimed surfaces until satisfactory plant cover is established (Fig. 62).

M. Final Reclamation
1. Salvage topsoil during decommissioning operations and reapply to all reclaimed surfaces including access roads (Fig. 63).
2. Use mat drilling to eliminate top-soil removal.
3. Replant a mixture of forbs, grasses, and shrubs that are native to the area and suitable for the specific ecological site.
4. Restore vegetation cover, composition, and diversity to achieve quantitative standards that are commensurate with the ecological site.
5. Do not allow livestock grazing on re-vegetated sites until plants are established and can withstand herbivory.
6. Continue to monitor and treat reclaimed areas until plant cover, composition, and diversity standards have been met.
7. Reevaluate the existing system of bonding. Bonds should be set at a level that is adequate to cover the company's liability for reclamation of the entire well field.

Appropriate planning and careful implementation of these guidelines can mean the difference between an unproductive remnant or a restored habitat capable of supporting wildlife and its associated recreational, economic and aesthetic benefits (Figs. 64 and 65).

Figure 62. Successful interim reclamation of gas well. (Photo courtesy of NMDGF).).

Figure 63. Reclamation of roads is necessary to eliminate permanent access and disturbance. (Photo courtesy of NMDGF).

Figure 64. Failed reclamation of a gas well. (Photo courtesy of BLM).

Figure 65. Successful final reclamation of a gas well. (Photo courtesy of NMDGF).

SUMMARY

Mule deer habitats in the CPE are highly variable, complex, and under considerable pressures from man's activities. There are a number of key concerns that the mule deer habitat manager must address.

Herbivory by wild and domestic ungulates has been and continues to be a major factor in shaping vegetation complexes throughout the CPE. Impacts of herbivory are always greater in times of drought. Much of the CPE has experienced severe droughts during the past 2 decades which have resulted in significant changes to vegetative composition and structure. Many of these changes have not been beneficial for mule deer. A constant theme voiced by managers is lack of habitat monitoring information that documents this dilemma. This is a critical need facing mule deer managers and efforts should be aimed at development of monitoring processes that provide information on basic rangeland conditions.

Once basic habitat information is in hand, and much is known about impacts of herbivore grazing, an obvious first step is development of a plan or assessment of the situation for the management area of concern. All pertinent monitoring information including maps (historic and current) and existing herbivore use data must be consolidated and land and wildlife managers must work effectively and collaboratively to design livestock and wild herbivore management plans to address habitat capabilities.

Droughts may exacerbate impacts of herbivores. To aid in assessing this impact various computer models are now available that rapidly integrate climatic and habitat variables and produce alternative forage allocation scenarios. Managers should take advantage of these tools and use resulting outputs to guide herd management and livestock stocking decisions.

In some situations, vegetative complexes have degraded to the point that it will be necessary to apply a vegetation management treatment before the system can begin to recover. These treatment efforts must be well designed and implemented. Effective control of existing wild and domestic grazing pressures is a major component of successful treatments and managers must be willing to take the steps needed to achieve those controls.

A closely related factor impacting mule deer habitats in the CPE is plant succession. As plants mature and reach late seral stages, they become well established and out-compete other plant species. A classic example in the CPE is encroachment of pinyon-juniper woodlands into sagebrush or grassland habitats. This is an often subtle and slow change that typically results in reduced plant diversity and less nutritious plants. Managers must first analyze and assess the magnitude of the problem by inspecting all pertinent information (both historic and recent). Once this is done, areas of highest priority for mule deer must be identified and appropriate treatment plans developed. Because fiscal resources will be limited, it is imperative that the manager select specific management strategies with potential to maximize positive results. Cost of treatment compared to benefits expected must be carefully analyzed. Treatments must also be designed collaboratively with public land managers to ensure all users are amenable and other natural resources (including other wildlife species) are considered before treatments are applied.

Another threat to mule deer ranges often overlooked and closely associated with man's activities, is nonnative, invasive plant species. Numerous invasives are spreading at an accelerated rate on public and private lands throughout the CPE. As the name implies, invasive plants often invade native plant communities and replace native plant species that are important as mule deer forage or cover. One of the worst offenders in the CPE is cheatgrass. This annual grass typically increases to the point that it is the dominant plant on the landscape. Cheatgrass also increases the frequency and intensity of fires on native rangelands. Repeated and frequent fires can greatly reduce the abundance of shrubs that are important for cover and forage for mule deer. Once cheatgrass is established, it is very difficult to remove and the threat of repeated fires increases with each fire event. In severe infestations costly and aggressive rehabilitation efforts may be necessary. Land and wildlife managers must work together to proactively address invasive plant species before they become dominant on the landscape.

In a dry environment, such as the CPE, water availability could potentially be a limiting factor for mule deer. As summer temperatures increase and snow melts faster in the spring as a result of global climate change, water availability will decrease. Some springs and streams that were historically permanent will become ephemeral. Management strategies that are aimed at adjustments in livestock grazing systems and riparian protection that will enhance water flows will be especially important in the future. In some cases, the need for water could justify water developments. However, there is disagreement on how beneficial water developments are for mule deer as well as other wildlife species. The manager must be aware that there are potential negative effects associated with wildlife water developments and benefits may not be sufficient to justify the project. The manager must also be aware that once a water development is built, there are on-going maintenance costs to consider.

Human encroachment within the CPE is a large and growing concern. Humans have discovered that many of the geologic, topographic, and habitat factors that characterize this ecoregion are attractive for a wide variety of human uses. Human activities that usurp and preclude mule deer occupation are the most detrimental. These include urban and suburban developments and all associated impacts like lawns, golf courses, and highways. However, human recreational activities can also be a concern, especially when large numbers of people are involved. Important habitats may not be lost directly, but become unavailable because of real or perceived suitability of the habitat. Unfortunately, there are few management options available to the manager to address these impacts. The most promising approach is to work to positively influence land management planning and zoning decisions so they better consider wildlife habitat values. For activities regulated by land and wildlife agencies, restrictions on human uses should be considered (e.g., using seasonal restrictions or closures) to minimize impacts on critical habitats.

A fast growing and major concern in the CPE is energy development. For decades, mineral and energy developments have occurred throughout the CPE. However, the growing national need for energy is now focusing development of these resources in the West. Unfortunately, much of the ecoregion overlays vast oil and gas resources and a major problem that mule deer and land managers face is lack of understanding of how development of these resources will impact mule deer. Little is known, but recent research has shown that large-scale and intensive developments are detrimental to mule deer. Common sense dictates that approaches involving staged and phased developments would be less detrimental than unplanned and poorly planned developments. In many situations, associated impacts such as roads, traffic patterns, noise, and human activities are as detrimental as, or more so, than wells that produce the energy. Land management agencies and industry must be made aware of critical mule deer habitats proposed for development.

To be most effective, the mule deer manager must strive to identify all possible alternative development plans that could lead to a lessened impact on those habitats. Responsible energy companies are demonstrating a willingness to work with land managers to lessen their foot prints. However, it is the responsibility of the mule deer manager to identify major impacts and suggest potential ameliorating or mitigating approaches. "Best management practices" for all aspects of energy development must be developed and implemented for all phases of an energy development process. Mule deer managers must be involved from the beginning of a project,

and on federal lands, especially so during the initial leasing process.Unfortunately, the many demands placed on today's mule deer managers have allowed less time to spend in the field directly observing and working with mule deer. As described in the preceding pages, the future mule deer manager must be accomplished in all things related to humans and their demands upon the land base. To do this effectively, the manager must be well informed biologically, ecologically, and sociologically. How to become informed is often a key question. We hope these guidelines will aid the manager in meeting this need. They were written with that goal in mind.

Aanes, R., B. E. Saether, and N. A. Oritsland. 2000. Fluctuations of an introduced population of Svalbard reindeer: the effects of density dependence and climatic variation. Ecography 23:437 − 443.

Ammann, A. P., R. L. Cowan, C. L. Mothershead, and B. R. Baumgardt. 1973. Dry matter and energy intake in relation to digestibility in white-tailed deer. Journal of Wildlife Management 37:195 − 201.

Anderson, A. A. 1969. 2,4-D, sagebrush, and mule deer-cattle use of upper winter range. Special Report Number 21, Colorado Division of Game, Fish and Parks, Fort Collins, Colorado, USA.

Anderson, E. W., and R. J. Scherzinger. 1975. Improving quality of winter forage for elk by cattle grazing. Journal of Range Management 28:120 − 125.

Archer, S., and D. A. Pyke. 1991. Plant-animal interactions affecting plant establishment and persistence on revegetated rangeland. Journal of Range Management 44:558-565.

Arizona Game and Fish Department (AGFD). 2004. Wildlife water development standards. Arizona Game and Fish Department, Phoenix, Arizona, USA.

Augustine, D. J., and S. J. McNaughton. 1998. Ungulate effects on the functional species composition of plant communities: herbivore selectivity and plant tolerance. Journal of Wildlife Management 62:1165 − 1183.

Austin, D. D. 2000. Managing livestock grazing for mule deer (Odocoileus hemionus) on winter range in the Great Basin. Western North American Naturalist 60:198 − 203.

Austin, D. D., and P. J. Urness. 1986. Effects of cattle grazing on mule deer diet and area selection. Journal of Range Management 39:18 − 21.

Austin, D. D., and P. J. Urness. 1998. Vegetal change on a northern Utah foothill range in the absence of livestock grazing between 1948 and 1982. Great Basin Naturalist 58:188 − 191.

Austin, D. D., P. J. Urness, and L. C. Fierro. 1983. Spring livestock grazing affects crested wheatgrass regrowth and winter use by mule deer. Journal of Range Management 36:589 − 593.

Bailey, D. W. 1988. Characteristics of spatial memory and foraging behavior in cattle. Dissertation, Colorado State University, Fort Collins, Colorado, USA.

Bailey, D. W., J. E. Gross, E. A. Laca, L. R. Rittenhouse, M. B. Coughenour, D. M. Swift, and P. L. Sims. 1996. Synthesis paper: mechanisms that result in large herbivore grazing distribution patterns. Journal of Range Management 49:386 − 400.

Bailey, D. W., L. R. Rittenhouse, R. H. Hart, D. M. Swift, and R. W. Richards. 1989. Association of relative food availabilities and locations by cattle. Journal of Range Management 42:480 − 482.

Baker, D. L., and N. T. Hobbs. 1987. Strategies of digestion: digestive efficiency and retention time of forage diets in montane ungulates. Canadian Journal of Zoology 65:1978 − 1984.

Baker, R. H. 1984. Origin, classification, and distribution. Pages 1 − 18 in L. K. Halls, editor. White-tailed deer: ecology and management. Wildlife Management Institute, Washington, D.C., and Stackpole Books, Harrisburg, Pennsylvania, USA.

Barnum, S. A. 2003. Identifying the best locations to provide safe highway crossing opportunities for wildlife. Pages 246-259 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Bartis, J. T., T. LaTourrette, L. Dixon, D. J. Peterson, and G. Cecchine. 2005. Oil shale development in the United States-prospects and policy issues (summary). Rand Corporation, Santa Monica, California, USA.

Bartmann, R. M. 1983. Composition and quality of mule deer diets on pinyon-juniper winter range, Colorado. Journal of Range Management 36:534 − 541.

Bartmann, R. M., G. C. White, and L. H. Carpenter. 1992. Compensatory mortality in a Colorado mule deer population. Wildlife Monographs 121:1 − 39.

Beck, J. L., and J. M. Peek. 2005. Diet composition, forage selection, and potential for forage competition among elk, deer, and livestock on aspen-sagebrush summer range. Rangeland Ecology and Management 58:135 − 147.

Behan, B., and B. L. Welch. 1985. Black sagebrush: mule deer winter preference and monoterpenoid content. Journal of Range Management 38:278 − 280.

Beier, P. 1991. Cougar attacks on humans in the United States and Canada. Wildlife Society Bulletin 19:403–412.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419 − 431.

Bender, L. C. 2003. Identification of factors limiting mule deer populations and development of corrective management strategies along the Upper Santa Fe Trail, New Mexico. New Mexico Cooperative Fish and Wildlife Research Unit, Las Cruces, New Mexico, USA.

Billings, W. D. 1994. Ecological impacts of cheatgrass and resultant fire on ecosystems in the Great Basin. Pages 22-30 in Proceedings: Symposium on ecology, management, and restoration of Intermountain annual rangelands. 18-22 May 1992, Boise, Idaho, USA.

Bishop, C. J., G. C. White, D. J. Freddy, and B. E. Watkins. 2005. Effect of nutrition and habitat enhancements on mule deer recruitment and survival rates. Federal Aid in Wildlife Restoration Project W − 185 − R, Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Bleich, V. C., L. J. Coombes, and J. H. Davis. 1982. Horizontal wells as a wildlife habitat improvement technique. Wildlife Society Bulletin 10:324 – 328.

Bleich, V. C., and R. A. Weaver. 1983. "Improved" sand dams for wildlife habitat management. Journal of Range Management 36:133.

Boroski, B. B., and A. S. Mossman. 1996. Distribution of mule deer in relation to water sources in Northern California. Journal of Wildlife Management. 60:7770-776.

Bowyer, R. T., and V. C. Bleich. 1984. Effects of cattle grazing on selected habitats of southern mule deer. California Fish and Game 70:240 – 247.

Boyd, C. S., and T. J. Svejcar. 2004. Regrowth and production of herbaceous riparian vegetation following defoliation. Journal of Range Management 57:448 – 454.

Boyd, R. J. 1970. Elk of the White River Plateau, Colorado. Technical Publication Number 25, Colorado Division of Game, Fish and Parks, Fort Collins, Colorado, USA.

Brigham, W. R., and C. Stevenson. 1997. Wildlife water catchment construction in Nevada. Bureau of Land Management Technical Note 397. Denver, Colorado, USA.

Briske, D. D., S. D. Fuhlendorf, and F. E. Smeins. 2005. State-and-transition models, thresholds, and rangeland health: a synthesis of ecological concepts and perspectives. Rangeland Ecology and Management 58:1 – 10.

Bristow, K. 1998. Effect of disturbance on reproduction of Coues white-tailed deer. Pages 39-46 in J. C. deVos, Jr., editor. Proceedings of the 1997 Deer/Elk Workshop, Arizona Game and Fish Department, Phoenix, USA.

Brown, D. E., and N. B. Carmony, editors.1995. Aldo Leopold's southwest. University of New Mexico Press, Albuquerque, New Mexico, USA.

Brown, R. L. 1994. Elk seasonal ranges and migrations in Arizona. Arizona Game and Fish Department Technical Report 15, Phoenix, Arizona, USA.

Broyles, B. 1995. Desert wildlife water developments: questioning use in the Southwest. Wildlife Society Bulletin. 23:663-675.

Bruinderink, G., and E. Hazebroek. 1996. Ungulate traffic collisions in Europe. Conservation Biology 10:1059-1067.

Bunderson, E. D., B. L. Welch, and D. J. Weber. 1986. In vitro digestibility of Juniperus osteosperma (Torr.) Little from 17 Utah sites. Forest Science 32:834 – 840.

Burk, C. J. 1973. The Kaibab deer incident: a long-persisting myth. Bioscience 23:113 – 114.

Cain, J. W.,III, P. R. Krausman, S. S. Rosenstock, and J. C. Turner. 2005. Literature review and annotated bibliography: water requirements of desert ungulates. U.S. Geological Survey Open-File Report 2005-1141. U.S. Geological Survey, Southwest Biological Science Center, Sonoran Desert Research Station, University of Arizona, Tucson, Arizona, USA.

Carmony, N. B., and D. E. Brown, editors. 2001. Man and wildlife in Arizona: the American exploration period, 1824 – 1865. Based on a Master of Science Thesis by Goode P. Davis, Jr. Arizona Game and Fish Department, Phoenix, Arizona, USA.

Carpenter, L. H., O. C. Wallmo, and R. B. Gill. 1979. Forage diversity and dietary selection by wintering mule deer. Journal of Range Management 32:226 – 229.

Carpenter, L. H. and O. C. Wallmo. 1981. Rocky Mountain and Intermountain habitats. Part 2: Habitat evaluation and management. Pages 399-422 in O.C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Carrel, W. K., R. A. Ockenfels, and R. E. Schweinsburg. 1999. An evaluation of annual migration patterns of the Paunsaugunt mule deer herd between Utah and Arizona: a final report. Research Branch Technical Report 29, Arizona Game and Fish Department, Phoenix, Arizona, USA.

Cartron, J. E., S. H. Stoleson, P. L. L. Stoleson, and D. W. Shaw. 2000. Riparian areas. Pages 281 – 327 in R. Jemison and C. Raish, editors. Livestock management in the American southwest: ecology, society, and economics. Elsevier Science B.V., Amsterdam, The Netherlands.

Caughley, G. 1970. Eruption of ungulate populations, with emphasis on Himalayan tahr in New Zealand. Ecology 51:53 – 72.

Cingolani, A. M., I. Noy-Meir, and S. Diaz. 2005. Grazing effects on rangeland diversity: a synthesis of contemporary models. Ecological Applications 15:757 – 773.

Claar, J. J., T. Bertram, R. Naney, N. Warren, and W. Ruediger. 2003. Wildlife linkage areas: an integrated approach for Canada lynx. Pages 234-239 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Clary, W. P., and W. C. Leininger. 2000. Stubble height as a tool for management of riparian areas. Journal of Range Management 53:562 – 573.

Clevenger, A. P., and N. Waltho. 2000. Factors influencing the effectiveness of wildlife underpasses in Banff National Park, Alberta, Canada. Conservation Biology 14:47-56.

Clevenger, A. P., and N. Waltho. 2003. Long-term, year-round monitoring of wildlife crossing structures and the importance of temporal and spatial variability in performance studies. Pages 293-302 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Clevenger, A. P., P. J. Wierzchowski, B. Chruszcz, and K. Gunson. 2002. GIS-generated expert based models for identifying wildlife habitat linkages and mitigation passage planning. Conservation Biology 16:503-514.

Clevenger A. P., B. Chruszcz, and K. Gunson. 2001. Highway mitigation fencing reduces wildlife-vehicle collisions. Wildlife Society Bulletin 29:646-653.

Cluff, L. K., B. L. Welch, J. C. Pederson, and J. D. Brotherson. 1982. Concentration of monoterpenoids in the rumen ingesta of wild mule deer. Journal of Range Management 35:192 − 194.

Coe, P. K., B. K. Johnson, K. M. Stewart, and J. G. Kie. 2004. Spatial and temporal interactions of elk, mule deer and cattle. Transactions of the North American Wildlife and Natural Resources Conference 69:656 − 669.

Connolly, G. E. 1981. Trends in populations and harvests. Pages 225 − 243 in O. C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Conover, M. R., W. C. Pitt, K. K. Kessler, T. J, DuBow, and W. A. Sandborn 1995. Review of human injuries, illnesses, and economic losses caused by wildlife in the United States. Wildlife Society Bulletin 23:407-414.

Cook, J. G. 2002. Nutrition and food. Pages 259 − 349 in D. E. Toweill and J. W. Thomas, editors. North American elk: ecology and management. Wildlife Management Institute, Washington, D.C., and Smithsonian Institution Press, Washington, D.C., USA.

Coppock, D. L., and A. H. Birkenfeld. 1999. Use of livestock and range management practices in Utah. Journal of Range Management 52:7 − 18.

Coughenour, M. B. 1991. Spatial components of plant-herbivore interactions in pastoral, ranching, and native ungulate ecosystems. Journal of Range Management 44:530 − 542.
Coughenour, M. B. 1993. The SAVANNA landscape model − documentation and users guide. Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colorado, USA.

Coulson, T., E. A. Catchpole, S. D. Albon, B. J. T. Morgan, J. M. Pemberton, T. H. Clutton-Brock, M. J. Crawley, and B. T. Grenfell. 2001. Age, sex, density, winter weather, and population crashes in Soay sheep. Science 292:1528 − 1531.

Daubenmire, R. F. 1940. Plant succession due to over-grazing in the Agropyron bunchgrass prairie of southeastern Washington. Ecology 21:55-64.

DeAngelis, D. L., and J. C. Waterhouse. 1987. Equilibrium and nonequilibrium concepts in ecological models. Ecological Monographs 57:1 − 21.

Demment, M. W., and P. J. Van Soest. 1985. A nutritional explanation for body size patterns of ruminant and non-ruminant herbivores. The American Naturalist 125:641 − 672.

Denney, R. N. 1976. Regulations and the mule deer harvest – political and biological management. Pages 87 − 92 in G. W. Workman and J. B. Low, editors. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

deVos, J. C., Jr., M. R. Conover, and N. E. Headrick, editors. 2003. Mule deer conservation: issues and management strategies. Berryman Institute Press. Utah State University, Logan, USA.

Dietz, D. R. 1965. Deer nutrition research in range management. Transactions of the North American Wildlife and Natural Resources Conference 30:274 − 285.

Dodd, N. L., J. W. Gagnon, and R. E. Schweinsburg. 2007. Application of video surveillance to assess wildlife use of highway underpasses in Arizona. Wildlife Society Bulletin 34:in press.

Einarsen, A. S. 1946. Crude protein determination of deer food as an applied management technique. Transactions of the North American Wildlife Conference 11:309 − 312.

Eisenhart, K. 2004. Historic range of variability and stand development in pinyon-juniper woodlands of western Colorado. Dissertation, University of Colorado, Boulder, USA.

Elcock, D., J. Gasper, and D. Moses. 1999. Environmental regulatory drivers for coal bed methane research and development. In Meeting the challenges of the nation's growing natural gas demand, volume I. National Petroleum Council. < http://ipec.utulsa.edu/Conf2002/elcock_gasper_moses/ > . Accessed March 2006.

Ellenberger, J. H. 1999. Colorado. Pages 78 − 81 and 174 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club, Missoula, Montana, and Foundation for North American Wild Sheep, Missoula, Montana, USA.

Ellis, J. E., and D. M. Swift. 1988. Stability of African pastoral ecosystems: alternate paradigms and implications for development. Journal of Range Management 41:450 − 459.

Elmore, W., and B. Kauffman. 1994. Riparian and watershed systems: degradation and restoration. Pages 212 − 231 in M. Vavra, W. A. Laycock, and R. D. Pieper, editors. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.

EnCana Oil and Gas (USA) Inc. 2005. Evaluation of drill pads constructed with oak mats to minimize impacts to soil and plant resources. EnCana Jonah Field Pilot Reclamation Program: 2005 Annual Report.

Endries, M., T. Gilbert, and R. Kautz. 2003. Mapping wildlife needs in Florida: the integrated wildlife habitat ranking system. Pages 525-534 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Farrell, J. E., L. R. Irby, and P. T. McGowen. 2002. Strategies for ungulate-vehicle collision mitigation. Intermountain Journal of Sciences 8:1-18.

Findholt, S. L., B. K. Johnson, D. Damiran, T. DelCurto, and J. G. Kie. 2004. Diet composition, dry matter intake and diet overlap of mule deer, elk and cattle. Transactions of the North American Wildlife and Natural Resources Conference 69:670 – 686.

Fisher, A. S. 1999. New Mexico. Pages 96 – 99 and 180 – 183 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club, Missoula, Montana, and Foundation for North American Wild Sheep, Missoula, Montana, USA.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:629 – 644.

Forman, R. T. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14:31-35.

Forman, R. T. T., and L. E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics 29:207-231.

Forman, R. T. T., D. Sperling, J. A. Bissonette, A. P. Clevenger, C. D. Cutshall, V. H. Dale, L. Fahrig, R. France, C. R. Goldman, K. Heanue, J. A. Jones, F. J. Swanson, T. Turrentine, and T. C. Winter. 2003. Road ecology: science and solutions. Island Press, Washington, D.C., USA.

Forsyth, D. M., and P. Caley. 2006. Testing the irruptive paradigm of large-herbivore dynamics. Ecology 87:297 – 303.

Foster, M. L., and S. R. Humphrey. 1995. Use of highway underpasses by Florida panthers and other wildlife. Wildlife Society Bulletin 23:95-100.

Fowler, N. L., R. D. Overath, and C. M. Pease. 2006. Detection of density dependence requires density manipulations and calculation of   . Ecology 87:655 – 664.

Freddy, D. J. 1985. Quantifying capacity of winter ranges to support deer – evaluation of thermal cover used by deer. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Freddy, D. J. 1994. Estimating survival rates of elk and developing techniques to estimate population size. Federal Aid in Wildlife Restoration Project W-153-R-6-3-9. Wildlife Research Report, Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Freddy, D. J., W. M. Bronaugh, and M. C. Fowler. 1986. Responses of mule deer to disturbance by persons afoot and snowmobiles. Wildlife Society Bulletin 14:63-68.

Garrott, R. A., G. C. White, R. M. Bartmann, L. H. Carpenter, and A. W. Alldredge. 1987. Movements of female mule deer in northwest Colorado. Journal of Wildlife Management 51:634 – 643.

Geist, V. 1981. Behavior: adaptive strategies in mule deer. Pages 157 – 223 in O. C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Gilbert, P. F., O. C. Wallmo, and R. B. Gill. 1970. Effect of snow depth on mule deer in Middle Park, Colorado. Journal of Wildlife Management 34:15 – 23.

Gill, R. B. 1976. Mule deer management myths and the mule deer population decline. Pages 99 – 106 in G. W. Workman and J. B. Low, editors. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

Gill, R. B., T. D. I. Beck, C. J. Bishop, D. J. Freddy, N. T. Hobbs, R. H. Kahn, M. W. Miller, T. M. Pojar, and G. C. White. 2001. Declining mule deer populations in Colorado: reasons and responses. Colorado Division of Wildlife Special Report Number 77, Denver, Colorado, USA.

Gillingham, M. P., and F. L. Bunnell. 1989. Effects of learning on food selection and searching behaviour of deer. Canadian Journal of Zoology 67:24 – 32.

Goodrich, S., E. D. McArthur, and A. H. Winward. 1999. Sagebrush ecotones and average annual precipitation. Pages 88-94 in E. D. McArthur, W. K. Ostler, C. L. Wambolt, and L. Carl, editors. Proceedings: sagebrush ecotones. RMRS-P-11, U.S. Forest Service, Ogden, Utah, USA.

Gordon, K. M., and S. H. Anderson. 2003. Mule deer use of underpasses in western and southeastern Wyoming. Pages 309-318 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Hanley, T. A. 1982. The nutritional basis for food selection by ungulates. Journal of Range Management 35:146 – 151.

Hansen, R. M., and L. D. Reid. 1975. Diet overlap of deer, elk, and cattle in southern Colorado. Journal of Range Management 28:43 – 47.

Hardy, A., A. P. Clevenger, M. Huijser, and G. Neale. 2003. An overview of methods and approaches for evaluating the effectiveness of wildlife crossing structures: emphasizing the science in applied science. Pages 319-330 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Harris, G. A. 1967. Some competitive relationships between Agropyron spicatum and Bromus tectorum. Ecological Monographs 37:89-111.

Harris, G. A. 1977. Root phenology as a factor of competition among grass seedlings. Journal of Range Management 30:172-177.

Hazam, J. E., and P. R. Krausman. 1988. Measuring water consumption of desert mule deer. Journal of Wildlife Management 52:528-534.

Heffelfinger, J. R. 2006. Deer of the southwest: a complete guide to the natural history, biology, and management of southwestern mule deer and white-tailed deer. Texas A&M University Press, College Station, USA.

Heffelfinger, J. R., and T. A. Messmer. 2003. Introduction. Pages 1 – 11 in J. C. deVos, Jr., M. R. Conover, and N. E. Headrick, editors. Mule deer conservation: issues and management strategies. Berryman Institute Press, Utah State University, Logan, USA.

Hervert, J. J., and P. R. Krausman. 1986. Desert mule deer use of water developments in Arizona. Journal of Wildlife Management 50:670-676.

Hobbs, N. T. 1996. Modification of ecosystems by ungulates. Journal of Wildlife Management 60:695 – 713.

Hobbs, N. T., D. L. Baker, J. E. Ellis, and D. M. Swift. 1981. Composition and quality of elk winter diets in Colorado. Journal of Wildlife Management 45:156 – 171.

Hobbs, N. T., D. L. Baker, J. E. Ellis, D. M. Swift, and R. A. Green. 1982. Energy-based and nitrogen-based estimates of elk winter-range carrying-capacity. Journal of Wildlife Management 46:12 – 21.

Hobbs, N. T., D. L. Baker, and R. B. Gill. 1983. Comparative nutritional ecology of montane ungulates during winter. Journal of Wildlife Management 47:1 – 16.

Hobbs, N. T., and D. M. Swift. 1985. Estimates of habitat carrying-capacity incorporating explicit nutritional constraints. Journal of Wildlife Management 49:814 – 822.

Hoddenbach, G. 1987. Tamarix control. Tamarisk control in the southwestern United
States. Cooperative National Park Resources Services Unit, Special Report Number 9:116-125.

Hofmann, R. R. 1989. Evolutionary steps of ecophysiological adaptation and diversification of ruminants: a comparative view of their digestive system. Oecologia 78:443 – 457.

Holechek, J. L. 1994. Financial returns from different grazing management systems in New Mexico. Rangelands 16:237 – 240.

Holechek, J. L. 1988. An approach for setting the stocking rate. Rangelands 10:10 – 14.

Holechek, J. L., R. D. Pieper, and C. H. Herbel. 2001. Range management: principles and practices. Fourth edition. Prentice Hall, Upper Saddle River, New Jersey, USA.

Holter, J. B., H. H. Hayes, and S. H. Smith. 1979. Protein requirement of yearling white-tailed deer. Journal of Wildlife Management 43:872 – 879.

Hormay, A.L. 1943. Bitterbrush in California. U.S. Forest Service, California Forest and Range Experiment Station, Forest Research Note 34, Berkeley, California, USA.

Horn, B. E., C. R. Hart, and S. I. Paisley. 2003. Management of rangeland livestock under drought. Annals of Arid Zone 41:1 – 23.

Howery, L. D., F. D. Provenza, R. E. Banner, and C. B. Scott. 1998. Social and environmental factors influence cattle distribution on rangeland. Applied Animal Behaviour Science 55:231 – 244.

Huijser, M. P., and P. T. McGowen. 2003. Overview of animal detection and animal warning systems in North America and Europe. Pages 368-382 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Hull, A. C., Jr, and J. F. Pechanec. 1947. Cheatgrass – a challenge to range research. Journal of Forestry 45:555-564.

Jensen, C. H., A. D. Smith, and G. W. Scotter. 1972. Guidelines for grazing sheep on rangelands used by big game in winter. Journal of Range Management 25:346 – 352.

Jensen, M. N. 2001. Can cows and conservation mix? BioScience 51:85 – 90.

Jobman, W. G. 1972. Consumption of juniper by deer and inhibition of rumen microorganisms by volatile oils of juniper. Thesis, Colorado State University, Fort Collins, USA.

Johnson, B. K., L. D. Hayden-Wing, and D. C. Lockman. 1990. Responses of elk to development of Exxon's Riley Ridge gas field in western Wyoming. Pages 42-45 in Proceedings of the Western States and Provinces elk workshop. California Department of Fish and Game, Sacramento, California, USA.

Johnson, B. K., J. W. Kern, M. J. Wisdom, S. L. Findholt, and J. G. Kie. 2000. Resource selection and spatial separation of mule deer and elk during spring. Journal of Wildlife Management 64:685 – 697.

Jorgensen, K. R. and R. Stevens. 2004. Seed collection, cleaning, and storage. Pages 699-716 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 3. U.S. Forest Service, Fort Collins, Colorado, USA.

Julander, O. 1962. Range management in relation to mule deer habitat and herd productivity in Utah. Journal of Range Management 15:278 – 281.

Julander, O., and J. B. Low. 1976. A historical account and present status of the mule deer in the West. Pages 3 – 19 in G. W. Workman and J. B. Low, editors. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

Karpowitz, J. 1999. Utah. Pages 192 – 195 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club, Missoula, Montana, and Foundation for North American Wild Sheep, Missoula, Montana, USA.

Keegan, T. W., and B. F. Wakeling. 2003. Elk and deer competition. Pages 139 – 150 in J. C. deVos, Jr., M. R. Conover, and N. E. Headrick,

editors. Mule deer conservation: issues and management strategies. Berryman Institute Press, Utah State University, Logan, USA.

Keil, M. R. 2001. Multi-level spatial decision making by cattle. Thesis, Colorado State University, Fort Collins, USA.

Kie, J. G. 1996. The effects of cattle grazing on optimal foraging in mule deer (Odocoileus hemionus). Forest Ecology and Management 88:131−138.

Kie, J. G., V. C. Bleich, A. L. Medina, J. D. Yoakum, and J. W. Thomas. 1994. Managing rangelands for wildlife. Pages 663−688 in T. A. Bookhout, editor. Research and management techniques for wildlife and habitats. The Wildlife Society, Bethesda, Maryland, USA.

Kie, J. G. and B. Czech. 2000. Mule and black-tailed deer. Pages 629-657 in S. Demarais and P. R. Krausman, editors. Ecology and management of large mammals in North America. Prentice-Hall, Upper Saddle River, New Jersey, USA.

Kie, J. G., C. J. Evans, E. R. Loft, and J. W. Menke. 1991. Foraging behavior by mule deer − the influence of cattle grazing. Journal of Wildlife Management 55:665−674.

Klyza, C. M. 1996. Who controls public lands?: Mining, forestry, and grazing policies, 1870−1990. University of North Carolina Press, Chapel Hill, North Carolina, USA.

Knipe, T. 1977. The Arizona whitetail deer. Special Report Number 6, Arizona Game and Fish Department, Phoenix, Arizona, USA.

Knox, K. L., J. G. Nagy, and R. D. Brown. 1969. Water turnover in mule deer. Journal of Wildlife Management 33:389-393.

Kruse, W. H., and R. Jemison. 2000. Grazing systems of the southwest. Pages 27−52 in R. Jemison and C. Raish, editors. Livestock management in the American southwest: ecology, society, and economics. Elsevier Science B.V., Amsterdam, The Netherlands.

Kuck, L., G. Hompland, and E. H. Merrill. 1985. Elk calf response to simulated mine disturbance in southeast Idaho. Journal of Wildlife Management 49:751-757.

Kufeld, R. C. 1973. Foods eaten by the Rocky Mountain elk. Journal of Range Management 26:106−113.

Kufeld, R. C. 1979. History and current status of the mule deer population on the east side of the Uncompahgre Plateau (Game Mgmt Unit 62). Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Kufeld, R.C. 1983. Responses of elk, mule deer, cattle, and vegetation to burning, spraying, and chaining of Gambel oak rangeland. Technical Publication 34, Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Kufeld, R. C., O. C. Wallmo, and C. Feddema. 1973. Foods of the Rocky Mountain mule deer. U.S. Forest Service Research Paper RM−111, Fort Collins, Colorado, USA.

Lang, E. M. 1958. Elk of New Mexico. Bulletin Number 8, New Mexico Department of Fish and Game, Albuquerque, New Mexico, USA.

Lautier, J. K., T. V. Dailey, and R. D. Brown. 1988. Effect of water restriction on feed intake of white-tailed deer. Journal of Wildlife Management 52:602-606.

Laycock, W. A., and D. A. Price. 1970. Environmental influences on nutritional value of forage plants. Pages 37−47 in H. A. Paulsen, Jr., E. H. Reid, and K. W. Parker, editors. Range and wildlife habitat evaluation − a research symposium. U.S. Department of Agriculture Miscellaneous Publication 1147, Washington, D.C., USA.

Lee, R. 1999. Arizona. Pages 72−73 and 160−163 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club, Missoula, Montana, and Foundation for North American Wild Sheep, Missoula, Montana, USA.

Lehnert, M. E., and J. A. Bissonette. 1997. Effectiveness of highway crosswalk structures at reducing deer-vehicle collisions. Wildlife Society Bulletin 25:809-818.

Leopold, A. 1941. Cheatgrass takes over. The Land 1:310-313.

Lindzey, F. G., W. G. Hepworth, T. A. Mattson, and A. F. Reese. 1997. Potential for competitive interactions between mule deer and elk in the western United States and Canada: a review. Wyoming Cooperative Fish and Wildlife Research Unit, Laramie, Wyoming, USA.

Loft, E. R., J. W. Menke, and J. G. Kie. 1991. Habitat shifts by mule deer − the influence of cattle grazing. Journal of Wildlife Management 55:16−26.

Loft, E. R., J. W. Menke, J. G. Kie, and R. C. Bertram. 1987. Influence of cattle stocking rate on the structural profile of deer hiding cover. Journal of Wildlife Management 51:655−664.

Longhurst, W. M., N. F. Baker, G. E. Connolly, and R. A. Fisk. 1970. Total body water and water turnover in sheep and deer. American Journal of Veterinary Research 31:673-677.

Lutz, D. W., M. Cox, B. F. Wakeling, D. McWhirter, L. H. Carpenter, S. Rosenstock, D. Stroud, L. C. Bender, and A. F. Reeve. 2003. Impacts and changes to mule deer habitat. Pages 13-61 in J. C. de Vos, Jr., M. R. Conover, and N. E. Headrick, editors. Mule deer conservation: issues and management strategies. Berryman Institute Press. Utah State University, Logan, USA.

Mack, R. N. 1981. Invasion of Bromus tectorum L. into western North America: an ecological chronicle. Agroecosystems 7:145-165.

Mackie, R. J. 1970. Range ecology and relations of mule deer, elk, and cattle in the Missouri River Breaks, Montana. Wildlife Monographs 20:1−79.

Mackie, R. J. 1981. Interspecific relationships. Pages 487−507 in O. C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Mackie, R. J., J. G. Kie, D. F. Pac, and K. L. Hamlin. 2003. Mule deer. Pages 889–905 in G. A. Feldhamer, B. C. Thompson, and J. A. Chapman, editors. Wild mammals of North America: biology, management, and conservation. Second edition. Johns Hopkins University Press, Baltimore, Maryland, USA.

Maestas, J. D., R. L. Knight, and W. C. Gilgert. 2002. Cows, condos, or neither: what's best for rangeland ecosystems? Rangelands 24:36 – 42.

Main, M. B., and B. E. Coblentz. 1996. Sexual segregation in Rocky Mountain mule deer. Journal of Wildlife Management 60:497-507.

Mason, R., L. H. Carpenter, M. Cox, J. C. deVos, J. Fairchild, D. J. Freddy, J. R. Heffelfinger, R. H. Kahn, S. M. McCorquodale, D. F. Pac, D. Summers, G. C. White, and B. K. Williams. 2006. A case for standardized ungulate surveys and data management in the western United States. Wildlife Society Bulletin 34:1238-1242.

McArthur, E. D., and S. B. Monsen. 2004a. Chenopod shrubs. Pages 467-492 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 2. U.S. Forest Service, Fort Collins, Colorado, USA.

McArthur, E. D., and S. B. Monsen. 2004b. Composite shrubs. Pages 493-538 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 2. U.S. Forest Service, Fort Collins, Colorado, USA.

McCullough, D. R. 1999. Density dependence and life-history strategies of ungulates. Journal of Mammalogy 80:1130 – 1146.

McNaughton, S. J., D. G. Milchunas, and D. A. Frank. 1996. How can net primary productivity be measured in grazing ecosystems? Ecology 77:974-977.

Meffe, G. K., and C. R. Carroll. 1997. Principles of conservation biology. Second edition. Sinauer and Associates, Sunderland, Massachusetts, USA.

Merrill, E. H., T. P. Hemker, K. P. Woodruff, and L. Kuck. 1994. Impacts of mining facilities on fall migration of mule deer. Wildlife Society Bulletin 22:68-73.

Milchunas, D. G. 2006. Responses of plant communities to grazing in the southwestern United States. General Technical Report RMRS-GTR-169. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado, USA.

Milchunas, D. G., M. I. Dyer, O. C. Wallmo, and D. E. Johnson. 1978. In-vivo/in-vitro relationships of Colorado mule deer forages. Special Report Number 43. Colorado Division of Wildlife, Fort Collins, Colorado, USA.

Miller, R. F., T. J. Svejcar, and N. E. West. 1994. Implications of livestock grazing in the Intermountain sagebrush region: plant composition. Pages 101-146 in M. Vavra, W. A. Laycock, and R. D. Pieper, editors. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.

Milner, J. M., D. A. Elston, and S. D. Albon. 1999. Estimating the contributions of population density and climatic fluctuations to interannual variation in survival of Soay sheep. Journal of Animal Ecology 68:1235 – 1247.

Monsen, S. B. 2004. Controlling plant competition. Pages 57-61 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Monsen, S. B., and R. Stevens. 2004. Seedbed preparation and seeding practices. Pages 121-154 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Monsen, S. B., R. Stevens, and N. Shaw. 2004a. Grasses. Pages 295-424 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 2. U.S. Forest Service, Fort Collins, Colorado, USA.

Monsen, S. B., R. Stevens, and N. Shaw. 2004b. Shrubs of other families. Pages 597-698 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 2. U.S. Forest Service, Fort Collins, Colorado, USA.

Mosley, J. C. 1996. Prescribed sheep grazing to suppress cheatgrass: a review. Sheep and Goat Research Journal 12:74-81.

Murphy, D. A., and J. A. Coates. 1966. Effects of dietary protein on deer. Transactions of the North American Wildlife and Natural Resources Conference 31:129 – 139.

Nagy, J. G., H. W. Steinhoff, and G. M. Ward. 1964. Effects of essential oils of sagebrush on deer rumen microbial function. Journal of Wildlife Management 28:785 – 790.

National Energy Policy. 2001. Reliable, affordable, and environmentally sound energy for America's future. Report of the national energy policy development group. U.S. Government Printing Office, Washington, D.C., USA.

National Research Council (NRC). 1981. Effect of environment on nutrient requirements of domestic animals. National Academy Press, Washington, D.C., USA.

Neal, L., T. Gilbert, T. Eason, L. Grant, and T. Roberts. 2003. Resolving landscape level highway impacts on the Florida black bear and other listed wildlife species. Pages 226-233 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Neff, D. J. 1974. Forage preferences of trained mule deer on the Beaver Creek watersheds. Arizona Game and Fish Department, Phoenix, Arizona, USA.

New Mexico Department of Game and Fish (NMDGF). 2004. Guidelines for oil and gas development and fish and wildlife resources. Santa Fe, New Mexico. <http://www.wildlife.state.nm.us/conservation/>. Accessed March 2006.

Ng, S. J., J. W. Dole, R. M. Sauvajot, S. P. D. Riley, and T. J. Valone. 2004. Use of highway underpasses by wildlife in southern California. Biological Conservation 115:499-507.

Noss, R. F., and A.Y. Cooperrider. 1994. Saving nature's legacy: protecting and restoring biodiversity. Defenders of Wildlife, Washington, D.C, and Island Press, Washington, D.C, USA.

Noy-Meir, I. 1993. Compensating growth of grazed plants and its relevance to the use of rangelands. Ecological Applications 3:32 – 34.

O'Gara, B. W., and R. G. Dundas. 2002. Distribution: past and present. Pages 67 – 119 in D. E. Toweill and J. W. Thomas, editors. North American elk: ecology and management. Wildlife Management Institute, Washington, D.C., and Smithsonian Institution Press, Washington, D.C., USA.

Paige, K. N. 1992. Overcompensation in response to mammalian herbivory – from mutualistic to antagonistic interactions. Ecology 73:2076 – 2085.

Painter, E. L., and A. J. Belsky. 1993. Application of herbivore optimization theory to rangelands of the western United States. Ecological Applications 3:2 – 9.

Pearson, H. A. 1973. Calculating grazing intensity for maximizing profit on Ponderosa pine range in northern Arizona. Journal of Range Management 26:277 – 278.

Peek, J. M., and P. R. Krausman. 1996. Grazing and mule deer. Pages 183 – 192 in P. R. Krausman, editor. Rangeland wildlife. Society for Range Management, Denver, Colorado, USA.

Peeples, S. S. 1991. Temporal and spatial patterns of foraging by cattle. Thesis, Colorado State University, Fort Collins, USA.

Perry, C., and R. Overly. 1977. Impact of roads on big game distributions in portions of the Blue Mountains of Washington, 1972-1973. Washington Department of Game Applied Research Bulletin 11, Olympia, USA.

Personius, T. L., C. L. Wambolt, J. R. Stephens, and R. G. Kelsey. 1987. Crude terpenoid influence on mule deer preference for sagebrush. Journal of Range Management 40:84 – 88.

Pieper, R. D. 1994. Ecological implications of livestock grazing. Pages 177 – 211 in M. Vavra, W. A. Laycock, and R. D. Pieper, editors. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.

Plummer, A. P., D. R. Christensen, and S. B. Monsen. 1968. Restoring big-game range in Utah. Publication Number 68 – 3, Utah Division of Fish and Game, Salt Lake City, Utah, USA.

Pojar, T. M., and D. C. Bowden. 2004. Neonatal mule deer fawn survival in west-central Colorado. Journal of Wildlife Management 68:550-660.

Provenza, F. D. 2003. Foraging behavior: managing to survive in a world of change. Behavioral principles for human, animal, vegetation, and ecosystem management. Utah State University, Logan, USA.

Provenza, F. D., and D. F. Balph. 1988. Development of dietary choice in livestock on rangelands and its implications for management. Journal of Animal Science 66:2356 – 2368.

Provenza, F. D., J. A. Pfister, and C. D. Cheney. 1992. Mechanisms of learning in diet selection with reference to phytotoxicosis in herbivores. Journal of Range Management 45:36 – 45.

Provenza, F. D., J. J. Villalba, C. D. Cheney, and S. J. Werner. 1998. Self-organization of foraging behavior: from simplicity to complexity without goals. Nutrition Research Reviews 11:199 – 222.

Puglisi, M. J., J. S. Lindzey, and E. D. Bellis. 1974. Factors associated with highway mortality of white-tailed deer. Journal of Wildlife Management 38:799-807.

Pyke, D. A. 1999. Invasive exotic plants in the sagebrush ecosystems of the Intermountain West. Pages 43-54 in Proceedings: sagebrush steppe ecosystems symposium, Boise, Idaho, USA.

Pyke, D. A., and S. J. Novak. 1994. Cheatgrass demography – establishment attributes, recruitment, ecotypes, and genetic variability. Pages 12-21 in Proceedings: symposium on ecology, management, and restoration of Intermountain annual rangelands. 18-22 May 1992, Boise, Idaho, USA.

Pyke, D. A., J. E. Herrick, P. Shaver, and M. Pellant. 2002. Rangeland health attributes and indicators for qualitative assessment. Journal of Range Management 55:584 – 597.

Quan, J., and E. Teachout. 2003. Balancing needs of transportation and the environment: successes and on-going challenges for the transportation liaison program at the USFWS in Washington state. Pages 570-572 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Ragotzkie, K. E., and J. A. Bailey. 1991. Desert mule deer use of grazed and ungrazed habitats. Journal of Range Management 44:487 – 490.

Rasmussen, D. I. 1941. Biotic communities of Kaibab Plateau, Arizona. Ecological Monographs 11:229 – 275.

Reed, D. F. 1981. Mule deer behavior at a highway underpass exit. Journal of Wildlife Management 45:542-543.

Reed, D. F., T. D. Beck, and T. N. Woodward. 1982. Methods of reducing deer-vehicle accidents: benefit-cost analysis. Wildlife Society Bulletin 10:349-354.

Reed, D. F., T. N. Woodward, and T. D. Beck. 1979. Regional deer-vehicle accident research. Report No. FHWA-CO-RD-79-11. Colorado Division of Highways, Denver, Colorado, USA.

Reed, D. F., T. N. Woodward, and T. M. Pojar. 1975. Behavioral response to mule deer to a highway underpass. Journal of Wildlife Management 39:361-367.

Reiner, R. J., and P. J. Urness. 1982. Effect of grazing horses managed as manipulators of big game winter range. Journal of Range Management 35:567 – 571.

Reudiger, B. 2001. High, wide, and handsome: designing more effective wildlife and fish crossing structures for roads and highways. Pages 509-516 in 2001 Proceedings of the International Conference on Ecology and Transportation. G. Evnik and K. P. McDermott, editors. Center for Transportation and the Environment, North Carolina State University, Raleigh, North Carolina, USA.

Reudiger, B., and J. Lloyd. 2003. A rapid assessment process for determining potential wildlife, fish and plant linkages for highways. Pages 205-225 in 2003 Proceedings of the International Conference on Ecology and Transportation. C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Center for Transportation and the Environment, North Carolina State University, Raleigh, North Carolina, USA.

Reynolds, H. G. 1966. Use of openings in spruce-fir forests of Arizona by elk, deer, and cattle. Research Note RM-66, U.S. Forest Service, Fort Collins, Colorado, USA.

Rice, W. E. 2004. Desert water harvesting to benefit wildlife: a simple, cheap, and durable sub-surface water harvester for remote locations. Environmental Monitoring and Assessment 99:251-257.

Riordan, L. E. 1956. Selectivity and utilization of three key species of range forage by cattle, sheep, and deer in western Colorado. Thesis, Colorado Agricultural and Mechanical College, Fort Collins, Colorado, USA.

Roath, L. R., E. M. Hardy, G. Wockner, S. Porter, N. T. Hobbs, and D. J. Freddy. 2006. The habitat assessment model: a tool to improve wildlife habitat management. Department of Forest, Range and Watershed Stewardship and the Natural Resource Ecology Laboratory, Colorado State University, and Colorado Division of Wildlife, Fort Collins, USA.

Roaza, R. 2003. Environmental planning in Florida: Florida's environmental screening tool: laying the technology foundation for efficient transportation decision making. Pages 520-524 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.
Robbins, C. T. 1983. Wildlife feeding and nutrition. Academic Press, New York, New York, USA.

Robertson, J. H. 1954. Half-century changes on northern Nevada ranges. Journal of Range Management 7:117-121.

Robinette, W. L., N. V. Hancock, and D. A. Jones. 1977. The Oak Creek mule deer herd in Utah. Publication Number 77 – 15, Utah Division of Wildlife Resources, Salt Lake City, Utah, USA.

Romin, L. A., and J. A. Bissonette. 1996. Deer-vehicle collisions: status of state monitoring activities and mitigation efforts. Wildlife Society Bulletin 24:276-283.

Rosenstock, S. S., W. B. Ballard, and J. C. deVos, Jr. 1999. Viewpoint: benefits and impacts of wildlife water developments. Journal of Range Management 52:302-311.

Rosenstock, S. S., M. J. Rabe, C. S. O'Brien, and R. B. Waddell. 2004. Studies of wildlife water developments in southwestern Arizona: wildlife use, water quality, wildlife diseases, wildlife mortalities, and influences on native pollinators. Arizona Game and Fish Department, Research Branch Technical Guidance Bulletin Number 8, Phoenix, Arizona, USA.

Rost, G. R., and J. A. Bailey. 1979. Distribution of mule deer and elk in relation to roads. Journal of Wildlife Management 47:634-641.

Russo, J. P. 1964. The Kaibab North deer herd: its history, problems and management. Wildlife Bulletin Number 7, Arizona Game and Fish Department, Phoenix, Arizona, USA.

Samuel, M. D., E. O. Garton, M. W. Schlegel, and R. G. Carson. 1987. Visibility bias during aerial surveys of elk in northcentral Idaho. Journal of Wildlife Management 51:622 – 630.

Sandoval, L., J. Holechek, J. Biggs, R. Valdez, and D. VanLeeuwen. 2005. Elk and mule deer diets in north-central New Mexico. Rangeland Ecology and Management 58:366 – 372.

Sawyer, H., F. Lindzey, D. McWhirter, and K. Andrews. 2002. Potential effects of oil and gas development on mule deer and pronghorn populations in western Wyoming. Transactions of the 67th North American Wildlife and Natural Resources Conference 67:350-365.

Schwabe, K. A., and P. W. Schuhmann. 2002. Deer-vehicle collisions and deer value: an analysis of competing literatures. Wildlife Society Bulletin 30:609-615.

Schwartz, C. C., J. G. Nagy, and W. L. Regelin. 1980a. Juniper oil yield, terpenoid concentration, and antimicrobial effects on deer. Journal of Wildlife Management 44:107 – 113.

Schwartz, C. C., W. L. Regelin, and J. G. Nagy. 1980b. Deer preference for juniper forage and volatile oil treated foods. Journal of Wildlife Management 44:114 – 120.

Schupp, E. W., J. C. Chamber, S. B. Vander Wall, J. M. Gómez, and M. Fuentes. 1999. Piñon and juniper seed dispersal and seedling recruitment at woodland ecotones. Pages 66-70 in McArthur, E. D., W. K. Ostler, and C. L. Wambolt, compilers. Proceedings: shrubland ecotones. RMRS-P-11, U.S. Forest Serivce, Ogden, Utah, USA.

Scott, C. B., F. D. Provenza, and R. E. Banner. 1995. Dietary habits and social interactions affect choice of feeding location by sheep. Applied Animal Behaviour Science 45:225 – 237.

Senft, R. L., M. B. Coughenour, D. W. Bailey, L. R. Rittenhouse, O. E. Sala, and D. M. Swift. 1987. Large herbivore foraging and ecological hierarchies. BioScience 37:789 – 799.

Serhveen, C., R. Shoemaker, and L. Lawrence. 2003. A sampling of wildlife use in relation to structure variable for bridges and culverts under I-90 between Alberton and St. Regis, Montana. Pages 331-341 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Severson, K. E., and A. L. Medina. 1983. Deer and elk habitat management in the Southwest. Journal of Range Management Monograph 2.

Severson, K. E., and P. J. Urness. 1994. Livestock grazing: a tool to improve wildlife habitat. Pages 232 – 249 in M. Vavra, W. A. Laycock, and R. D. Pieper, editors. Ecological implications of livestock herbivory in the West. Society for Range Management, Denver, Colorado, USA.

Shaw, N. L., S. B. Monsen, and R. Stevens. 2004. Rosaceous shrubs. Pages 539-596 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 2. U.S. Forest Service, Fort Collins, Colorado, USA.

Shepherd, H. R. 1971. Effects of clipping on key browse species in southwestern Colorado. Technical Publication Number 28, Colorado Game, Fish and Parks, Fort Collins, Colorado, USA.

Shields, W. 1999. Utah. Pages 108 – 111 in D. E. Toweill and V. Geist, editors. Return of royalty: wild sheep of North America. Boone and Crockett Club, Missoula, Montana, and Foundation for North American Wild Sheep, Missoula, Montana, USA.

Short, H. L. 1970. Deer in Arizona and New Mexico: their ecology and a theory explaining recent population decreases. General Technical Report RM-70. U.S. Forest Service, Fort Collins, Colorado, USA.

Short, H. L. 1981. Nutrition and metabolism. Pages 99 – 127 in O. C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Short, H. L., and J. C. Reagor. 1970. Cell wall digestibility affects forage value of woody twigs. Journal of Wildlife Management 34:964-967.

Singleton, P. H., W. L. Gaines, and J. F. Lehmkuhl. 2002. Landscape permeability for large carnivores in Washington: a geographic information system weighted-distance and least-cost corridor assessment. U.S. Forest Service Research Paper PNW-RP-549, Portland, Oregon, USA.

Singer, F. J., and J. L. Doherty. 1985. Managing mountain goats at a highway crossing. Wildlife Society Bulletin 13:469-477.

Smith, M. A., J. C. Malechek, and K. O. Fulgham. 1979. Forage selection by mule deer on winter range grazed by sheep in spring. Journal of Range Management 32:40 – 45.

Spalinger, D. E., C. T. Robbins, and T. A. Hanley. 1986. The assessment of handling time in ruminants: the effect of plant chemical and physical structure on the rate of breakdown of plant particles in the rumen of mule deer and elk. Canadian Journal of Zoology 64:312 – 321.

SRM Task Group (Society for Range Management Task Group on Unity in Concepts and Terminology Committee, Society for Range Management). 1995. New concepts for assessment of rangeland condition. Journal of Range Management 48:271 – 282.

Stalling, D. 2003. Gas and oil development on western public lands; impacts on fish, wildlife, hunting, and angling. Trout Unlimited-Public Lands Initiative. < http://www.tu.org/atf/cf/ > . Accessed March 2006.

Stevens, R. 2004a. Basic considerations for range and wildland revegetation and restoration. Pages 19-24 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Stevens, R. 2004b. Incorporating wildlife habitat needs into restoration and rehabilitation projects. Pages 155-173 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Stevens, R., and S. B. Monsen. 2004a. Mechanical plant control. Pages 65-88 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Stevens, R., and S. B. Monsen. 2004b. Guidelines for restoration and and rehabilitation of principal plant communities. Pages 199-294 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Stevens, R., and S. B. Monsen. 2004c. Forbs for seeding range and wildlife habitats. Pages 425-466 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Stewart, G., and A. E. Young. 1939. The hazard of basing permanent grazing on Bromus tectorum. Agronomy Journal 31:1003-1015.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: resource partitioning and competitive displacement. Journal of Mammalogy 83:229 – 244.

Stohlgren, T. J., K. A. Bull, and Y. Otsuki. 1998. Comparison of rangeland vegetation sampling techniques in the Central Grasslands. Journal of Range Management 51:164-172

Sullivan, T. L., A. F. Williams, T. A. Messmer, L. A. Hellinga, and S. Y. Kyrychenko. 2004. Effectiveness of temporary warning signs in reducing deer-vehicle collisions during mule deer migrations. Wildlife Society Bulletin 32:907-915.

Swift, D. M. 1983. A simulation-model of energy and nitrogen-balance for free-ranging ruminants. Journal of Wildlife Management 47:620 – 645.

Swift, P. K., J. D. Wehausen, H. B. Ernest, R. S. Singer, A. M. Pauli, H. Kinde, T. E. Rocke, and V. C. Bleich. 2000. Desert bighorn sheep mortality due to presumptive type C botulism in California. Journal of Wildlife Diseases 36:184-189.

SWReGAP (Southwest Regional GAP Analysis Project). 2004. Provisional digital landcover dataset for the southwestern United States. U.S. Geological Survey and Utah State University, Logan, USA. <http://earth.gis.usu.edu/swgap/>. Accessed 17 May 2006.

Tessmann, S., J. Bohne, B. Oakleaf, B. Rudd, S. Smith, V. Stetler, D. Stroud, S. Wolff. 2004. DRAFT: Minimum recommendations to sustain important wildlife habitats affected by oil and gas development: a strategy for managing energy development consistently with the FLPMA principles of multiple use and sustained yield. Wyoming Game and Fish Department, Cheyenne, Wyoming, USA.

Thomas, J. W., H. Black, Jr., R. J. Scherzinger, and R. J. Pedersen. 1979. Deer and elk. Pages 104-127 in Wildlife habitats in managed forests - the Blue Mountains of Oregon and Washington. Agricultural handbook Number 533, U.S. Forest Service, Portland, Oregon, USA.

Thurow, T. L., and C. A. Taylor, Jr. 1999. Viewpoint: The role of drought in range management. Journal of Range Management 52:413−419.

Torbit, S. C., L. H. Carpenter, D. M. Swift, and A. W. Alldredge. 1985. Differential loss of fat and protein by mule deer during winter. Journal of Wildlife Management 49:80−85.

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

Torstenson, W. L. F., J. C. Mosley, T. K. Brewer, M. W. Tess, and J. E. Knight. 2006. Elk, mule deer, and cattle foraging relationships on foothill and mountain rangeland. Rangeland Ecology and Management 59:80−87.

Trombulak, S. C., and C. A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14:18-30.

Trout, L. E., and J. L. Thiessen. 1973. Physical condition and range relationships of the Owyhee deer herd. Federal Aid in Wildlife Restoration, Job Completion Report, Project W−141−R−2. Idaho Department of Fish and Game, Boise, Idaho, USA.

Tucker, D. G., E. S. Gardner, and B. F. Wakeling. 2004. Elk habitat use in relation to residential development in the Hualapai Mountains, Arizona. Pages 89-96 in C. van Riper III and K. L. Cole, editors. The Colorado Plateau: cultural, biological, and physical research. University of Arizona Press, Tucson, Arizona, USA.

Unsworth, J. W., L. Kuck, and E. O. Garton. 1990. Elk sightability model validation at the National Bison Range, Montana. Wildlife Society Bulletin 18:113−115.

Unsworth, J. W., D. F. Pac, G. C. White, and R. M. Bartmann. 1999. Mule deer survival in Colorado, Idaho, and Montana. Journal of Wildlife Management 63:315−326.

Urness, P. J. 1976. Mule deer habitat changes resulting from livestock practices. Pages 21−35 in G. W. Workman and J. B. Low, editors. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

United States Department of the Interior (USDI), Bureau of Land Management. 1997. White River resource area; resource management plan/environmental impact statement. <http://www.co.blm.gov/nepa/rmpdocs/wrfodocs/wrformp/>. Accessed March 2006.

United States Department of the Interior (USDI), Bureau of Land Management. 1999. Glenwood Springs resource area; oil and gas leasing and development: final supplemental environmental impact statement. <http://www.co.blm.gov/nepa/rmpdocs/glenwood_springs/>. Accessed March 2006

United States Department of the Interior (USDI), Bureau of Land Management. 2000. Public land statistics. Washington, D.C., USA. <http://www.blm.gov/natacq/pls00/>. Accessed 25 June 2006.

United States Department of the Interior (USDI), Bureau of Land Management. 2003. Farmington proposed resource management plan and final environmental impact statement. <http://www.nm.blm.gov/ffo/ffo_p_rmp_feis/ffo_p_rmp_index/>. Accessed March 2006

United States Department of the Interior (USDI), Bureau of Land Management. 2004. Public Land Statistics. Washington, D.C., USA. <http://www.blm.gov/natacq/pls04/>. Accessed 25 June 2006.

United States Department of Interior (USDI), Bureau of Land Management. 2005a. Programmatic Environmental Assessment: Large capacity and small capacity wildlife water developments. NV-020-05-08. Winnemucca Field Office, Nevada, USA.

United States Department of the Interior (USDI), United States Geological Survey. 2005b. Coal-bed gas resources of the Rocky Mountain region. U.S. Geological Survey Fact Sheet 158-02. 2pp. <http://pubs.usgs.gov/fs/fs-158-02/>. Accessed March 2006.

Vallentine, J. F. 2004. Herbicides for plant control. Pages 89-100 in S. B. Monsen, R. Stevens, and N. L. Shaw, compilers. Restoring western ranges and wildlands. General Technical Report RMRS-GTR-136, volume 1. U.S. Forest Service, Fort Collins, Colorado, USA.

Van Soest, P. J. 1982. Nutritional ecology of the ruminant. O & B Books, Corvallis, Oregon, USA.

Vavra, M. 2005. Livestock grazing and wildlife: developing compatibilities. Rangeland Ecology and Management 58:128−134.

Vavra, M., and D. Ganskopp. 1998. Grazing behavior in ungulates: current concepts and future challenges. Annals of Arid Zone 37:319−335.

Verme, L. J., and D. E. Ullrey. 1972. Feeding and nutrition of deer. Pages 275–291 in D. C. Church, editor. Digestive physiology and nutrition of ruminants. Volume 3, Practical nutrition. D. C. Church, Corvallis, Oregon, USA.

Verme, L. J., and D. E. Ullrey. 1984. Physiology and nutrition. Pages 91-118 in L. K. Halls, editor. White-tailed deer: ecology and management. Wildlife Management Institute, Washington, D.C., and Stackpole, Harrisburg, Pennsylvania, USA.

Vogel, W. O. 1989. Responses of deer to density and distribution of housing in Montana. Wildlife Society Bulletin 17:406-413.

Wallace, M. C., and P. R. Krausman. 1987. Elk, mule deer, and cattle habitats in central Arizona. Journal of Range Management 40:80–83.

Wallmo, O. C. 1978. Mule and black-tailed deer. Pages 31-41 in J. L. Schmidt and D. L. Gilbert, editors. Big game of North America: ecology and management. Stackpole Books, Harrisburg, Pennsylvania, USA.

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

Wallmo, O. C., and W. L. Regelin. 1981. Rocky Mountain and Intermountain habitats. Part 1: Food habits and nutrition. Pages 387–398 in O. C. Wallmo, editor. Mule and black-tailed deer of North America. Wildlife Management Institute, Washington, D.C., and University of Nebraska Press, Lincoln, Nebraska, USA.

Wambolt, C. L. 1996. Mule deer and elk foraging preference for 4 sagebrush taxa. Journal of Range Management 49:499–503.

Wang, G., N. T. Hobbs, R. B. Boone, A. W. Illius, I. J. Gordon, J. E. Gross, and K. L. Hamlin. 2006. Spatial and temporal variability modify density dependence in populations of large herbivores. Ecology 87:95–102.

Ward, A. L. 1971. In vitro digestibility of elk winter forage in southern Wyoming. Journal of Wildlife Management 35:681–688.

Warg, S. A. 1938. Life history and economic studies on Bromus tectorum. Thesis. Montana State University, Missoula, USA.

Weisberg, P. J., N. T. Hobbs, J. E. Ellis, and M. B. Coughenour. 2002. An ecosystem approach to population management of ungulates. Journal of Environmental Management 65:181–197.

Welch, B. L. 1989. Nutritive value of shrubs. Pages 405–424 in C. M. McKell, editor. The biology and utilization of shrubs. Academic Press, San Diego, California, USA.

Welch, B. L. 1997. Comment: big sagebrush pro versus con. Journal of Range Management 50:322–323.

Welch, B. L., and E. D. McArthur. 1981. Variation of monoterpenoid content among subspecies and accessions of Artemisia tridentata grown in a uniform garden. Journal of Range Management 34:380–384.

Welch, B. L., E. D. McArthur, and J. N. Davis. 1983. Mule deer preference and monoterpenoids (essential oils). Journal of Range Management 36:485–487.

Welch, B. L., and J. C. Pederson. 1981. In vitro digestibility among accessions of big sagebrush by wild mule deer and its relationship to monoterpenoid content. Journal of Range Management 34:497–500.

Whisenant, S. G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. Pages 4-10 in S. B. Monsen and S. G. Kitchen, compilers. Proceedings – ecology and management of annual rangelands. U.S. Forest Service General Technical Report INT-GTR-313, Ogden, Utah, USA.

White, G. C., and R. M. Bartmann. 1998a. Effect of density reduction on overwinter survival of free-ranging mule deer fawns. Journal of Wildlife Management 62:214–225.

White, G. C., and R. M. Bartmann. 1998b. Mule deer management – what should be monitored? Pages 104–118 in J. C. deVos, Jr., editor. Proceedings of the 1997 Deer/Elk workshop, Rio Rico, Arizona. Arizona Game and Fish Department, Phoenix, Arizona, USA.

White, G. C., and B. C. Lubow. 2002. Fitting population models to multiple sources of observed data. Journal of Wildlife Management 66:300–309.

White, P. A., and M. Ernst. 2003. Second nature: improving transportation without putting nature second. Defenders of Wildlife, Washington, D.C., USA.

Wildeman, G., and J. H. Brock. 2000. Grazing in the southwest: history of land use and grazing since 1540. Pages 1–25 in R. Jemison and C. Raish, editors. Livestock management in the American Southwest: ecology, society, and economics. Elsevier Science B.V., Amsterdam, The Netherlands.

Winder, J. A., C. C. Bailey, M. G. Thomas, and J. L. Holechek. 2000. Breed and stocking rate effects on Chihuahuan Desert cattle production. Journal of Range Management 53:32–38.

Wisdom, M. J., A. A. Ager, H. K. Preisler, N. J. Cimon, and B. K. Johnson. 2005. Effects of Off-Road Recreation on Mule Deer and Elk. Pages 67-80 in Wisdom, M. J., technical editor, The Starkey Project: a synthesis of long-term studies of elk and mule deer. Reprinted from the 2004 Transactions of the North American Wildlife and Natural Resources Conference, Alliance Communications Group, Lawrence, Kansas, USA.

Wisdom, M. J., M. M. Rowland, B. K. Johnson, and B. L. Dick. 2004. Overview of the Starkey Project: mule deer and elk research for management benefits. Transactions of the North American Wildlife and Natural Resources Conference 69:455–474.

Wolfe, M. L. 1976. Reliability of mule deer population measurements. Pages 93–98 in G. W. Workman and J. B. Low, editors. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

Wood, M. K., B. A. Buchanan, and W. Skeet. 1995. Shrub preference and utilization by big game on New Mexico reclaimed mine land. Journal of Range Management 48:431 – 437.

Workman, G. W., and J. B. Low, editors. 1976. Mule deer decline in the west: a symposium. Utah State University, Logan, USA.

Wostl, R. 2003. A programmatic Section 7 consultation to restore habitat connectivity and achieve recovery for a federally threatened species: Preble's meadow jumping mouse. Pages 608-612 in C. L. Irwin, P. Garrett, and K. P. McDermott, editors. Proceedings of the International Conference on Ecology and Transportation. Center for Transportation and the Environment, North Carolina State University, Raleigh, USA.

Yensen, D. L. 1981. The 1900 invasion of alien plants into southern Idaho. Great Basin Naturalist 41:176-183.

Yoakum, J. D. 2004a. Distribution and abundance. Pages 75 – 105 in B. W. O'Gara and J. D. Yoakum, editors. Pronghorn ecology and management. Wildlife Management Institute, Washington, D.C., and University Press of Colorado, Boulder, Colorado, USA.

Yoakum, J. D. 2004b. Relationships with other herbivores. Pages 503 – 538 in B. W. O'Gara and J. D. Yoakum, editors. Pronghorn ecology and management. Wildlife Management Institute, Washington, D.C., and University Press of Colorado, Boulder, Colorado, USA.

Yoakum, J., W. P. Dasmann, H. R. Sanderson, C. M. Nixon, and H. S. Crawford. 1980. Habitat improvement techniques. Pages 329-403 in S. D. Schemnitz, editor. Wildlife management techniques manual. Fourth edition. The Wildlife Society, Washington, D.C., USA.

Young, J. A., and F. L. Allen. 1997. Cheatgrass and range science: 1930-1950. Journal of Range Management 50:530-535.

Young, J. A., and R. A. Evans. 1985. Demography of Bromus tectorum in Artemesia communities. Pages 489-502 in J. White, editor. The population structure of vegetation. Dr. W. Junk, Dordrecht, The Netherlands.

Young, J. A , R. A. Evans, and P. T. Tueller. 1976. Great Basin plant communities – pristine and ungrazed. Pages 187-215 in R. Elston and P. Headrick, editors. Holocene environmental change in the Great Basin. Nevada Archaeology Survey, Research Paper Number 6, University of Nevada, Reno, Nevada, USA.

Young, J. A , R. A. Evans, R. E. Eckert, Jr., and B. L. Kay. 1987. Cheatgrass. Rangelands 9:266-270.

APPENDIX A.

Alphabetical listing by category of common names (with scientific names) of species cited in the text.

TREES AND SHRUBS

Aspen, Trembling (*Populus tremuloides*)
Bitterbrush, Antelope (*Purshia tridentata.*)
Blackbrush (*Coleogyne ramosissima*)
Chokecherry (*Prunus virginiana*)
Cliffrose (*Cowania stansburiana*)
Eriogonum, Sulfur (*Eriogonum umbellatum*)
Figwort (*Scrophularia californica*)
Fir, Douglas (*Pseudotsuga menziesii*)
Fir, Subalpine (*Abies lasiocarpa*)
Fir, White (*Abies concolor*)
Juniper, Alligator (*Juniperus deppeana*)
Juniper, Utah (*Juniperus osteosperma*)
Juniper, Western (*Juniperus occidentalis*)
Mountain Mahogany, True (*Cercocarpus montanus*)
Oak, Gambel (*Quercus gambelii*)
Pine, Lodgepole (*Pinus contorta*)
Pine, Pinyon (*Pinus edulis*)
Pine, Ponderosa (*Pinus ponderosa*)
Rabbitbrush (*Chrysothamnus* spp.)
Rose, Woods (*Rosa woodsii*)
Sagebrush, Big (*Artemisia tridentata*)
Saltbush, Four-wing (*Atriplex canescens*)
Serviceberry (*Amelanchier* spp.)
Serviceberry, Utah (*Amelanchier utahensis*)
Snakeweed (*Gutierrezia* spp.)
Snowberry (*Symphoricarpos* spp.)
Spruce, Blue (*Picea pungens*)
Spruce, Engelmann (*Picea engelmannii*)

FORBS AND GRASSES

Alfalfa (*Medicago sativa*)
Aster (*Aster* spp.)
Balsamroot, Arrowleaf (*Balsamorhiza sagittata*)
Bluegrass (*Poa* spp.)
Bluegrass, Mutton (*Poa fendleriana*)
Bluegrass, Sanderg (*Poa secunda*)
Brome, Smooth (*Bromus inermis*)
Buckwheat (*Eriogonum* spp.)
Burnet, Small (*Sanguisorba minor*)
Cheatgrass (*Bromus tectorum*)
Cryptantha (*Cryptantha sericea*)
Fescue (*Festuca* spp.)
Fescue, Idaho (*Festuca idahoensis*)
Globemallow (*Sphaeralcea* spp.)
Goldenweed (*Haplopappus* spp.)
Grama, Blue (*Bouteloua gracilis*)
Hoarycress (*Cardaria draba*)
Hound's-tongue (*Cynoglossum officinale*)
Junegrass, Prairie (*Koeleria macrantha*)
Knapweed, Diffuse (*Centaurea diffusa*)
Knapweed, Russian (*Centaurea repens*)
Knapweed, Spotted (*Centaurea maculosa*)
Leafy Spurge (*Euphorbia esula*)
Lupine (*Lupinus* spp.)
Needle and Thread Grass (*Stipa comata*)
Needlegrass (*Stipa* spp.)

Orchardgrass (*Dactylis glomerata*)
Penstemon, Mat (*Penstemon* spp.)
Phlox (*Phlox* spp.)
Ricegrass, Indian (*Oryzopsis hymenoides*)
Sage, Fringed (*Artemisia frigida*)
Sagewort (*Artemsia ludoviciana*)
Sainfoin (*Onobrychis viciaefolia*)
Squirreltail, Bottlebrush (*Sitanion hystrix*)
Tamarisk (*Tamarix ramosissima*)
Thistle, Canada (*Cirsium arvense*)
Thistle, Musk (*Cardus nutans*)
Thistle, Russian (*Salsola tragus*)
Thistle, Scotch (*Onopordum acanthium*)
Toadflax, Dalmatian (*Linaria dalmatica*)
Toadflax, Yellow (*Linaria vulgaris*)
Tumblemustard (*Sisymbrium altissimum*)
Wheatgrass, Bluebunch (*Pseudoroegneria spicata*)
Wheatgrass (*Agropyron* spp.)
Wheatgrass, Fairway Crested (*Agropyron cristatum*)
Wheatgrass, Western (*Agropyron smithii*)
Wildrye, Great Basin (*Elymus cinereus*)

ANIMALS

Bison (*Bison bison*)
Cattle, Domestic (*Bos taurus*)
Coyote (*Canis latrans*)
Deer, Black-tailed (*Odocoileus hemionus columbianus*)
Deer, Mule (*Odocoileus hemionus hemionus*)
Deer, White-tailed (*Odocoileus virginianus*)
Elk, Rocky Mountain (*Cervus elaphus*)
Goat, Domestic (*Capra hircus*)
Horse, Domestic or Feral (*Equus caballus*)
Midge (*Culicoides* spp.)
Pronghorn (*Antilocapra americana*)
Sheep, Desert Bighorn (*Ovis canadensis nelsoni*)
Sheep, Domestic (*Ovis aries*)
Sheep, Rocky Mountain Bighorn (*Ovis canadensis canadensis*)

APPENDIX B.

List of important forage plants [Common name (Scientific name)] for mule deer in the Colorado Plateau Ecoregion. Adapted from Short (1970), Kufeld et al. (1973), Bartmann (1983), McArthur and Monsen (2004a, b), Monsen et al. (2004a, b), Shaw et al. (2004), and Stevens and Monsen (2004c).* Generally preferred species; ** Preferred during certain seasons or growth stages; *** Species that are commonly eaten by mule deer but would seldom be expected to meet maintenance energy and/or nitrogen requirements.

NATIVE TREES AND SHRUBS

Apache Plume (*Fallugia paradoxa*)**
Aspen, Trembling (*Populus tremuloides*)*
Birch, Bog (*Betula glandulosa*)**
Bitterbrush, Antelope (*Purshia tridentata*)*
Blackbrush (*Coleogyne ramosissima*)***
Blueberry (*Vaccinium* spp.)*
Budsage (*Artemisia spinescens*)**

Native Trees and Shrubs cont'd

Ceanothus, Desert (*Ceanothus greggii*)**
Ceanothus, Fendler (*Ceanothus fendleri*)**
Ceanothus, Martin (*Ceanothus martini*)**
Chokecherry (*Prunus virginiana*)*
Cliffrose, Stansbury (*Cowania stansburiana*)*
Cottonwood, Narrowleaf (*Populus angustifolia*)**
Currant, Golden (*Ribes aureum*)*
Currant, Wax (*Ribes cereum*)**
Dogwood, Redosier (*Cornus stolonifera*)*
Elderberry, Blue (*Sambucus cerulean*)**
Ephedra, Green or Mormon Tea (*Ephedra viridis*)***
Eriogonum, Sulfur (*Eriogonum umbellatum*)*
Fir, Douglas (*Pseudotsuga menziesii*)***
Fir, White (*Abies concolor*)***
Grape, Oregon (*Mahonia repens*)**
Greasewood, Black (*Sarcobatus vermiculatus*)***
Hopsage, Spiny (*Grayia spinosa*)**
Horsebrush, Gray (*Tetradymia canescens*)***
Juniper, Creeping (*Juniperous horizontalis*)***
Juniper, Rocky Mountain (*Juniperus scopulorum*)***
Juniper, Utah (*Juniperus osteosperma*)***
Juniper, Western (*Juniperus occidentalis*)***
Mahogany, Curlleaf Mountain (*Cercocarpus ledifolius*)*
Mahogany, True Mountain (*Cercocarpus montanus*)*
Maple, Rocky Mountain (*Acer glabrum*)**
Manzanita, Greenleaf (*Arctostaphylos patula*)***
Ninebark, Mallowleaf (*Physocarpus malvaceous*)**
Oak, Gambel (*Quercus gambelii*)**
Pine, Pinyon (*Pinus edulis*)***
Pine, Ponderosa (*Pinus ponderosa*)***
Rabbitbrush, Dwarf (*Chrysothamnus depressus*)**
Rabbitbrush, Low (*Chrysothamnus viscidiflorus*)**
Rabbitbrush, Rubber (*Chrysothamnus nauseosus*)**
Rose, Woods (*Rosa woodsii*)**
Sagebrush, Big (*Artemisia tridentata*)**
Sagebrush, Bigelow (*Artemsia bigelovii*)**
Sagebrush, Black (*Artemisia nova*)**
Sagebrush, Low (*Artemisia arbuscula*)**
Sagebrush, Silver (*Artemsia cana*)**
Saltbush, Four-wing (*Atriplex canescens*)**
Serviceberry, Saskatoon (*Amelanchier alnifolia*)*
Serviceberry, Utah (*Amelanchier utahensis*)*
Shadscale (*Atriplex confertifolia*)**
Snowberry (*Symphoricarpos* spp.)*
Sumac, Rocky Mountain (*Rhus glabra*)**
Sumac, Skunkbush (*Rhus aromatica*)**
Willow (*Salix* spp.)*
Winterfat (*Ceratoides lanata*)**
Yucca, Soapweed (*Yucca glauca*)**

Native Forbs, Grasses and Sedges

Aster (*Aster* spp.)**
Balsamroot, Arrowleaf (*Balsamorhiza sagittata*)**
Bluebell, Tall (*Mertensia arizonica*)**
Bluegrass, Mutton (*Poa fendleriana*)**
Bluegrass, Sanderg (*Poa secunda*)**
Brome, Nodding (*Bromus frondosus*)**
Buckwheat (*Eriogonum* spp.)*
Cinquefoil (*Potentilla* spp.)*
Cryptantha (*Cryptantha sericea*)**
Dropseed, Sand (*Sporobolus cryptandrus*)**

Eriogonum, Spearleaf (*Eriogunum lonchophyllum*)
Fescue (*Festuca* spp.)**
Fescue, Idaho (*Festuca idahoensis*)**
Galleta Grass (*Hilaria sericea*)**
Geranium (*Geranium* spp.)**
Globemallow (*Sphaeralcea* spp.)**
Goldenweed, Nuttall (*Haplopappus nuttallii*)**
Grama, Black (*Bouteloua eriopoda*)**
Grama, Blue (*Bouteloua gracilis*)**
Grama, Sideoats (*Bouteloua curtipendula*)**
Groundsel, Butterweed (*Senecio serra*)**
Helianthella, Oneflower (*Helianthella uniflora*)**
Hymenopappus, Fineleaf (*Hymenopappus filifolius*)**
Junegrass (*Koeleria* spp.)**
Lomatium (*Lomatium* spp.)**
Lupine, Silky (*Lupinus sericeus*)**
Lupine, Tailcup (*Lupinus caudatus*)**
Needle and Thread Grass (*Stipa comata*)**
Parsnip, Cow (*Heracleum lanatum*)**
Penstemon (*Penstemon* spp.)**
Phlox (*Phlox* spp.)**
Ricegrass, Indian (*Oryzopsis hymenoides*)**
Sage, Fringed (*Artemisia frigida*)**
Sagewort (*Artemisia ludoviciana*)**
Squirreltail, Bottlebrush (*Elymus elymoides*)**
Sedge (*Carex* spp.)**
Sweetvetch, Utah (*Hedysarum boreale*)*
Wheatgrass, Bluebunch (*Pseudoroegneria spicata*)**
Wheatgrass, Slender (*Agropyron trachycaulum*)**
Wheatgrass, Thickspike (*Agropyron dasystachyum*)**
Wheatgrass, Western (*Agropyron smithii*)**
Wildrye, Great Basin (*Elymus cinereus*)**
Yarrow, Western (*Achillea millefolium*)**

Non-native Forbs, Grasses and Sedges

Alfalfa (Medicago sativa)*
Bluegrass, Kentucky (*Poa pratensis*)**
Brome, Smooth (*Bromus inermis*)**
Burnet, Small (*Sanguisorba minor*)*
Cheatgrass (*Bromus tectorum*)**
Clover, Strawberry (*Trifolium fragiferum*)*
Crownvetch (*Coronilla varia*)*
Foxtail, Creeping (*Alopecurus arundinaceus*)**
Kochia, Forage (*Kochia prostrata*)**
Orchardgrass (*Dactylus glomerata*)**
Sainfoin (*Onobrychis viciaefolia*)*
Sweetclover, Yellow (*Melilotus officinalis*)**
Timothy (*Phleum pretense*)**
Wheatgrass, Standard Crested (*Agropyron desertorum*)**
Wheatgrass, Fairway Crested (*Agropyron cristatum*)**
Wheatgrass, Intermediate (*Agropyron intermedium*)**

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