HABITAT GUIDELINES FOR MULE DEER

INTERMOUNTAIN WEST 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 (USFWS) 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. With this wide latitudinal and geographic range comes a great diversity of different climatic regimes and vegetation associations. With this range of habitats comes an incredibly diverse array of behavioral and ecological adaptations that have allowed this species to succeed amid such diversity.

These diverse environmental and climatic conditions result in a myriad of dynamic relationships between mule deer and their habitats. Within the geographic distribution of mule deer, however, areas can be grouped together into "ecoregions" within which deer populations share certain similarities regarding the issues and challenges that land managers must face. Within 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.

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 the frequency of fire and resulted in a more open landscape. In contrast, the California Woodland Chaparral and many forested areas in the Intermountain West are lacking the natural fire regime that once opened canopies 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 the 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. Over time, we have learned much about mule deer foods and cover, but more remains to be learned. For example, we have learned that cover is not a simple matter; the amelioration that vegetation and topography provide under highly variable weather conditions is a key aspect of mule deer well-being. Mule deer have basic life history requirements that weave a common thread throughout many issues facing mule deer.

Mule deer are primarily browsers, with a majority of their diet comprised of forbs (weeds) and browse (leaves and twigs of woody shrubs). Deer digestive tracts differ from cattle (*Bos taurus*) and elk (*Cervus elaphus*) 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 than 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 the biological carrying capacity is reached. If deer populations remain at or beyond carrying capacity, they begin to affect their habitats in a negative manner. The manager must also be aware that long-term impacts like drought conditions and vegetation succession can significantly lower the carrying capacity for deer and even if a droughty period ends, the overall capacity may be lower than it might have been 20 years earlier. This may well be the situation in many mule deer habitats in the west and the manager must be cognizant of this factor.

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 in the name 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 thus 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. An obvious question to habitat managers will be—at what scale do I apply my treatments? This is a legitimate question and obviously a difficult question to answer. Treated areas must be sufficiently large to produce a "treatment" effect. There is no 1 "cookbook" rule for scale of treatment. However, the manager should realize the effect of a properly applied treatment is larger than the actual number of acres treated because deer will move in and out of the treatments and thus a larger area of habitat will benefit. 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 the benefit will be greatest and then subsequent habitat improvement activities can be linked to this core area.

These habitat management guidelines are intended to be used by a broad spectrum of people involved and interested in mule deer habitat management and stewardship on public and private lands. These guidelines are tiered from the North American Mule Deer Conservation Plan. The photographs and specific guidelines therein are intended to communicate important components of mule deer habitats across the range of the species and suggest management strategies. The authors do not take credit for some of the guidelines presented. Those guidelines developed elsewhere are simply reiterated in this document to emphasize, and perhaps validate, their importance to mule deer and their habitats. Further, it is recognized many land managers have multiple-use mandates or other primary objectives other than mule deer.

THE INTERMOUNTAIN WEST ECOREGION

Description

The various mountain ranges and valleys west of the Rocky Mountains, east of the Sierra Nevada, and south of the Canadian border comprise the Intermountain West (IMW) (Fig. 1). The IMW includes portions of California, Oregon, Washington, Idaho, Wyoming, Colorado, Utah, and most of Nevada. Mule deer in this ecoregion inhabit areas primarily classified as sagebrush (*Artemisia* spp.)-steppe. However, the lower latitudes of this ecoregion include the Great Basin, which is considered a cold desert, whereas forests occupy many of the upper elevations in the ecoregion. The prevailing climate in the IMW is semi-arid. The northern regions receive most of the annual precipitation in the form of snow at higher elevations. Although annual precipitation in the IMW is highly variable (5 - 30 in.), most annual precipitation values are in the range of 10-20 inches. Winter snow accumulation in the high country can be significant and is essential to assure perennial spring and stream flows. Winter snow pack is also critical in providing soil moisture necessary for production and maintenance of high quality mule deer forage. Drought and overgrazing can substantially limit summer forage production. Much of the lower elevations are characterized by low precipitation. Soils are variable and often consist of basalt and other volcanic derivatives in the valleys and lowlands, whereas many of the higher elevations and mountain ranges contain granitic-based soils. Most of the soils throughout this ecoregion are nitrogen limited.

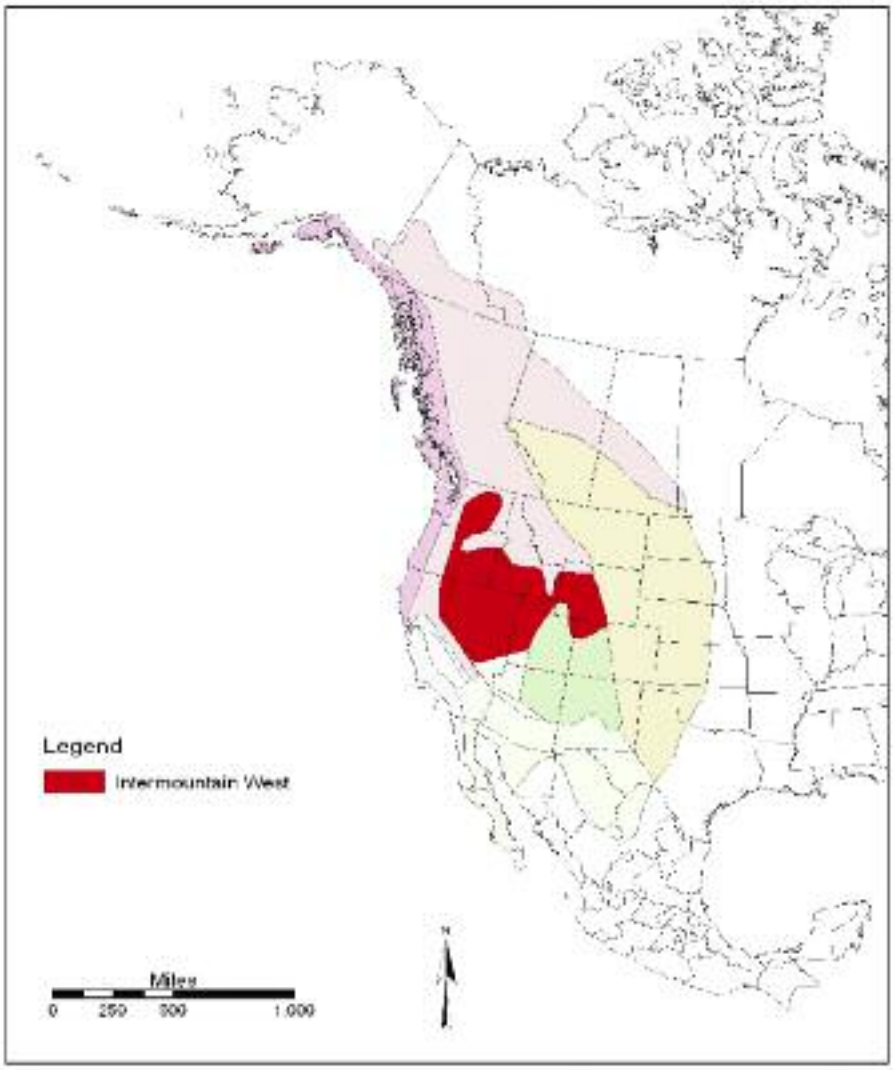

Figure 1. The Intermountain West Ecoregion (Sue Boe/Arizona Game and Fish Department [AGFD]).

Ecoregion-specific deer ecology

Historically, this ecoregion was the epicenter of mule deer distribution and many of the classic mule deer studies occurred in this region. Seasonal migrations are common, with deer moving great distances from higher elevation summer ranges to lower elevation winter ranges. Deep snows in winter can be a problem. Some areas, however, support large mule deer populations year-round. Diversity of vegetation and topography usually characterize areas with higher populations. Humans, primarily to improve forage for livestock, have manipulated many historic transitional and winter ranges. Agricultural and urban conversions are common in this region.

Key management issues include loss of shrubland (sagebrush and mountain brush species) integrity, conversion of native vegetation to agriculture lands and residential developments, and cumulative habitat degradation from overgrazing. Loss of lands and fragmentation of habitats caused by urbanization and recreation use are major threats. Pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) encroachment is also a major problem because thousands of acres of valuable mule deer range, primarily shrublands, are being taken over by pinyon-juniper (P-J) each year (Miller et al. 2008).

Fire patterns are a concern at lower elevations in this region, (Clements and Young 1997). The proliferation of cheatgrass has shortened the fire frequency from a historic 30- to 100-year cycle to a 5- to 10-year cycle in portions of the region. The result is conversion of thousands of acres of woody vegetation to cheatgrass and other invasive (or

undesirable) species. Livestock grazing systems that not only degrade native herbaceous understory, but are inadequate in controlling the spread of invasive plants or rejuvenating decadent shrubs further complicate matters. Range and wildlife managers must seek creative solutions to these issues or valuable shrublands will be lost.

Winter maintenance habitat appears to exert less influence than total amount and quality of reproductive and summer maintenance habitat on population size or overall density of deer (Mackie et al. 1998). However, during extreme winter events or significant habitat losses, as has occurred with extensive winter range fires or urban development, winter range can exert significant influence on population size (Pac et al. 1991). Mule deer in mountain-foothill environments contend with winter energy deficits that are of longer duration than experienced in areas of the Great Plains. In these environments, recruitment averages 30 fawns:100 adults on winter ranges with severe environments and 40 fawns:100 adults on milder sites. Following droughts and severe winters, recruitment can reach lows of 5-20 fawns:100 adults, while natural mortality of does may exceed 15%. Environmental extremes, forage quality, and the resulting condition of animals are key factors in mule deer population dynamics in the IMW.

Climate change may increase environmental extremes and influence habitat changes for mule deer in the IMW. The rate of global warming has increased 30-fold in the last 10,000 – 20,000 years (deVos and McKinney 2007). Changes in vegetative communities have been observed as a result of increased greenhouse gases including CO2, changes in precipitation and snowfall patterns, and increased temperatures (deVos and McKinney 2007). Since about 1950, global climate change corresponds with widespread changes in distribution and trends of biotic communities. Included in these changes is a shift towards the poles of species of about 4 miles per decade, a retreat upward on mountains of about 20 feet per decade, and earlier onset of spring activities by many species of plants and animals (deVos and McKinney 2007). Within the IMW, responses to climate change may include expanded distribution of woody species, reduced nutritional quality of forages, increased frequency of stand-converting wildfires, and spread of invasive plants and insects. These changes and trends have increased in the past 150 years, resulting in different biotic communities and interactions between species. As global warming progresses, the extent of these changes and altered biological interactions will increase. Although the causes for mule deer population declines over the past century are varied, some of the decline can be attributed to weather extremes including large-scale droughts and severe winters. Predicted climate changes are likely to exert a strong influence on biodiversity of vegetative communities in western North America. Many of these changes are likely to challenge the adaptability of mule deer and may alter abundance and distribution of mule deer in the IMW.

MAJOR IMPACTS TO MULE DEER HABITAT IN THE INTERMOUNTAIN WEST

Vegetative species composition has been modified.
In some cases, noxious or invasive species have proliferated in native plant communities, replacing native shrub communities with a perennial herbaceous understory to nonnative grasslands dominated by invasive plants. More subtly, some less desirable species have become more abundant at the expense of more desirable species (e.g., rabbitbrush [*Ericameria* spp., *Chysothamnus* spp.] replacing higher quality antelope bitterbrush [*Purshia tridentata*] or cheatgrass replacing perennial grasses).

Vegetative structure has been modified.
Expansion and maturation of pinyon-juniper woodlands in the absence of disturbance has decreased understory diversity and productivity, resulting in less forage for deer. Increasing woody cover in some cases decreases the amount and diversity of herbaceous species. Often, mule deer browse species are decreased as a result of encroachment by woody species. Concurrently, the expansion of non-native invasive species such as cheatgrass, have dramatically altered vegetative structure across entire landscapes.

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.

Usable habitat has been lost and fragmented due to human encroachment and associated activities.
The human population of the IMW is increasing rapidly as many people move to the area because of the natural beauty, desirable climate, job opportunities, 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 IMW, resulting in extensive development for energy extraction on public and private lands. Lower elevation winter range areas are being most impacted by development. In addition, an ever increasing number of people are recreating on public lands in the IMW and use of motorized transportation in the backcountry is becoming more popular every year.

Excessive Herbivory

Background

By most accounts from early explorers, trappers, and settlers to the Intermountain West, mule deer were not overly abundant (Gruell 1986). In the late 1800s, as human settlement progressed in the IMW, the numbers of domestic sheep (*Ovis aries*), goats (*Capra hircus*), cattle, and horses (*Equus caballus*) increased dramatically on most rangelands. Millions of nomadic sheep and cattle roamed unregulated through much of this ecoregion (Fig. 2). Shortly after the turn of the century, the U. S. Forest Reserves (now the U.S. Forest Service [USFS]) and in 1934, the U.S. Grazing Service (now the Bureau of Land Management [BLM]) were formed, in part to administer grazing on public lands. Gradually, grazing regulations were implemented. Along with regulation came fences, grazing seasons, forage allocation, and other infrastructure.

Figure 2. Historic land use practices such as this early 1900s nomadic sheep grazing in Nevada significantly altered and ultimately improved mule deer habitat by providing shrubs a competitive advantage over severely depleted herbaceous vegetation. (Photo courtesy of Nevada Historical Society).

From the late 1800s through the early 1900s the destructive and exhaustive overgrazing by livestock and feral horses contributed to a landscape-wide stand renewal process (Clements and Young 1997). Healthy perennial bunchgrass/shrub-steppe communities were turned to a landscape with severely depleted herbaceous understories. This in turn gave way to shrub seedling establishment and in mid- to upper-elevations, resulted in early-seral mountain brush dominated shrublands. These shrublands literally fueled a dramatic increase in mule deer populations range-wide. Although overgrazing and associated disturbances were instrumental in creating and maintaining productive mule deer habitat prior to the mid 20th century (Gruell 1986), over the long-term, improper grazing has reduced the quality and capacity of mule deer habitats (Pickford 1932, Cottam and Evans 1945, Reynolds and Trost 1980, Martin and Klein 1984).

Figure 3. Visible effects of excessive herbivory in year-round mule deer habitat in western Nevada. (Photo by Mike Cox/NDOW).

There is much confusion about the interchangeability of terms such as grazing, over-grazing, and overuse. A discussion of the effects of livestock on vegetation must be based on a consistent use of terminology. "Grazing" is neither good nor bad, it is simply consumption of available forage by an herbivore. Grazing the annual production of herbage at inappropriately high intensities is termed "overuse." "Overgrazing" describes a condition where the range is chronically overused for a multi-year period resulting in degeneration in plant species composition and soil quality (Severson and Urness 1994). There are different levels of overgrazing; range can be slightly overgrazed or severely overgrazed (Severson and Medina 1983).

Issues and Concerns

Grazing and Mule Deer Habitat

Livestock grazing has the potential to change both food and cover available to deer. Although precipitation and environmental extremes are the most important factors affecting deer nutrition and fawn survival in the IMW, habitat conditions impacted by ungulate density determine how much of that nutrition and cover remains available to deer. Livestock grazing can cause both short- and long-term changes to mule deer habitat (Peek and Krausman 1996, Bleich et al. 2005). Grazing at light to moderate levels has

Figure 4. Healthy mule deer habitat contains a diversity of trees, browse, forbs, and grasses to provide for the nutrition and cover requirements of both fawns and adults. (Photo by Mike Cox/NDOW).

Figure 5. During drought or when herbaceous material is overused, cattle feed more heavily on browse (like this stunted bitterbrush on outside of exclosure), which can decrease important nutritional resources for deer and seed production for plants. (Photo by Mike Cox/NDOW).

little impact on deer, but overuse in arid environments removes much of the herbaceous cover that is crucial for doe nutrition and fawning cover (Bowyer and Bleich 1984, Loft et al. 1987, Galindo-Leal et al. 1994). Long-term changes resulting from overgrazing include undesirable changes in plant communities, decreased mulch cover, decreased water infiltration, compacted soil, increased water runoff, decreased plant vigor and production, and a drier microclimate at ground level (Fig. 3, Severson and Medina 1983).

Overgrazing also removes browse leaves and twigs important to mule deer, further exacerbating poor nutritional conditions created by removal of forbs (Hanson and McCulloch 1955). Mule deer benefit from consumption of forbs throughout the year. Consumption of grasses and forbs in spring and summer are especially important to mule deer (Austin and Urness 1985). However, heavy livestock use can result in significant reductions in species richness, primarily by decreasing amounts of grasses and forbs (Cottam and Evans 1945, Austin et al. 1986). Additionally, livestock sometimes browse important deer shrubs excessively (Swank 1958, Knipe 1977). Heavy utilization of bitterbrush can be especially harmful to mule deer. Dasmann and Blaisdell (1954) found steep declines in fawn survival when bitterbrush utilization exceeded 34%. Jones (2000) reviewed literature from arid rangelands in western North America and found overuse and overgrazing had significant detrimental effects on 11 of 16 variables measured (mostly soil and vegetation characteristics).

Reducing the intensity of grazing generally results in improvements in range condition, but there is a misconception that removing cattle will always result in the range recovering to a climax state or pristine condition (Pieper 1994:202, Briske et al. 2003). Long-term deferments from grazing in arid and semi-arid regions may not result in any significant improvement in range condition (Laycock 1991, Holechek et al. 1998:191), or improvements may take 40-50 years (Valone et al. 2002, Guo 2004). Although overgrazing has impacted the IMW, grazing is sustainable in this ecoregion if stocking rates are at appropriate levels and season of use is given consideration (Fig. 4, Holechek et al. 1999).

Mechanisms of Competition

Competition between 2 species can occur for any resource that is in short supply and used by both. Concerns of ungulate competition are usually focused on forage resources. The degree of forage competition between 2 species depends primarily on the amount of dietary overlap (similarity in diet) and whether the plants used by both are in short supply (Holechek et al. 1998:385). A high degree of dietary overlap alone does not infer competition; it only indicates the potential exists.

Competition for resources can occur between native ungulates in some cases, but generally competition is

greater between 2 species that have not evolved separate niches. White-tailed deer (*Odocoileus virginianus*) and mule deer have very similar diets in the Southwest (Anthony and Smith 1977), but generally stay separated spatially by occupying different elevation zones. Deer carrying capacity fluctuates slightly in the IMW resulting in varying potential for competition. Periods of high deer densities and excessive browsing can lower the quality and condition of deer browse in some areas.

Elk and bison (*Bison bison*) occur(red) intermittently throughout much of the IMW ecoregion. Both bison and elk are primarily grazers; however elk are more flexible in both habitat and forage use and can impact forbs and browse, while very little spatial overlap exists between mule deer and bison. The ecological relationship between elk and mule deer has been studied and although there exists a possibility of population level competition, results have been inconsistent.

Domestic sheep and goats have diets very similar to deer (forbs and browse) and as such have the potential to seriously reduce forage available to deer (Smith and Julander 1953). Increasing demand for goat meat has resulted in renewed interest in raising goats on public land. However, cattle are by far the most important class of livestock to consider here because of their abundance and widespread distribution across the IMW.

Dietary overlap is an important consideration, but if shared forage plants are not used heavily there may be no competition for food. Proper levels of grazing allow different types of ungulates to assume their natural dietary niche. Under appropriate grazing regimes, cattle primarily eat grass (if available) and have a lesser impact on forbs and browse. However, many forbs are highly palatable to cattle and, given their larger size, cattle can remove a large volume of forbs (Lyons and Wright 2003). During drought or when the annual growth of herbaceous material is overused, cattle and elk can switch more heavily to browse and competition with deer increases (Severson and Medina 1983). Hot season grazing by cattle can often lead to overutilization of browse (Fig. 5).

Ungulates are not the only class of animals that can affect vegetation and potentially compete with mule deer for forage. In some cases, cyclic lagomorph populations common in the IMW can reach levels that significantly affect the herbaceous understory as well as low growing leaders of browse species. Additionally, rodents can impact grass and forb density through seed predation and herbivory (Brown and Heske 1990, Howe and Brown 1999). As a result, it is important for managers to consider all grazers in the area and how they are using vegetation.

Deer avoid areas occupied by large numbers of cattle, and they are more abundant in areas ungrazed by cattle (McIntosh and Krausman 1982, Wallace and Krausman 1987). This may be related to nutritional resources, lack of cover, or behavioral avoidance. Overuse and, ultimately, overgrazing can reduce the amount of cover to an extent that fewer deer can occupy an area regardless of forage availability. This is especially important during parturition and early fawn rearing, when cover for fawns is vital to their survival (Loft et al. 1987). Horejsi (1982) reported that grazing negatively impacted fawn survival only during drought years. In late-seral stage shrublands, ungrazed areas provide better habitat for mule deer than grazed sites. It is recognized in early to mid-seral stage mountain brush communities with adequate moisture; livestock use may be much less competitive and at times can stimulate succulent vegetative growth. Because of the widespread presence of cattle throughout the IMW, using appropriate grazing practices may be one of the best possibilities for improving mule deer nutrition on a landscape scale (Fig. 6, Longhurst et al. 1976).

Figure 6. With appropriate grazing practices mule deer benefit from adequate cover and herbaceous understory forage especially during spring and early summer. Both photos have similar site potential in low elevation, valley bottoms, but the lower site has been overgrazed by livestock and lacks necessary resources to support a healthy deer herd. (Photos by Mike Cox/NDOW).

Figure 7. A mountain big sagebrush community in southern Idaho suffering from overgrazing; with appropriate stocking rates, more herbaceous understory would remain during the early summer months when forbs and even grasses can be an important nutritional component to mule deer diets (Photo by Gene Gray/IDFG).

Stocking Rate

Selecting the appropriate stocking rate is the most important consideration in range management decisions from the standpoint of vegetation, livestock, wildlife, and economic return (Lyons and Wright 2003). Stocking rate has more influence on vegetation productivity than any other grazing factor (Holechek 1994, 1996; Holechek et al. 1998, 2000). Overstocking can prevent range improvement in an otherwise appropriate grazing system (Fig. 7, Eckert and Spencer 1987); therefore, a good grazing system alone will not result in range improvement if the stocking level exceeds sustainability. Timing and intensity of grazing are important considerations, but more than any other parameter, stocking rate determines whether an area is properly grazed or overused. Therefore, stocking rate is the key to maintaining nutritional and cover requirements of mule deer in the IMW.

As important as stocking rate is, at times there are other considerations that are nearly as important to maintaining high quality mule deer habitat. The timing of grazing, for example, can be important when the goal is providing fawning cover or retaining an herbaceous layer of forbs. In some cases, even grazing at a low or moderate stocking rate during spring forb production may negatively affect the amount of nutrition available to mule deer in semi-arid regions.

Rotational Grazing

Savory (1988) advocated grazing pastures intensively and moving livestock frequently to improve range conditions while simultaneously increasing the stocking rate. It was claimed that range managers could commonly double the stocking rate and see improvements in range and livestock productivity (Holechek et al. 2000). During the last few decades research has failed to confirm these claims. A synthesis of grazing studies worldwide found that short-duration grazing was not superior to continuous grazing when stocking rates were the same (Briske et al. 2008). The increased "hoof action" of a large number of cattle did not increase water infiltration in the soil as claimed by Savory (1988). In arid ecosystems, there was no advantage to various rotation grazing systems over continuous grazing when considering range condition, grazing efficiency, livestock productivity, or financial returns (Holechek 1994, 1996; Holechek et al. 1999). Despite this, some range managers continue to allow or even promote inappropriately high stocking rates with short-duration grazing. One concept of short-duration grazing that can have a positive benefit on mule deer habitat even under a continuous grazing strategy is periodic redistribution of livestock. This is especially true in the IMW where terrain is often rugged, steep, with limited water distribution, and sensitive riparian areas used as mule deer fawning habitats can receive excessive use.

Riparian

Riparian vegetation occupies a small proportion of the land area in the IMW but has an extremely important function in providing for the year-round habitat requirements of mule deer. These linear habitat features provide mature trees for thermal and screening cover and drainage patterns promote pooling of water, growth of forbs, and a greater diversity of important shrubs (Fig. 8). Unfortunately, these elements also attract livestock for the same reasons (Fig. 9). Belsky et al. (1999) summarized research documenting negative effects of livestock overgrazing on riparian ecosystems in the West. Riparian habitats must be carefully considered in overall grazing strategies.

Improving Habitat with Livestock

Some work has been done to investigate the use of livestock as a mule deer habitat improvement tool (Severson 1990). Improving habitat with livestock grazing does not include simply relaxing grazing pressure to improve conditions, but rather actually altering the condition or structure of forage to increase deer carrying capacity above that in the absence of livestock. Livestock grazing has resulted in improvements to mule deer habitat in the past, but these improvements have not always been planned actions (Connolly and Wallmo 1981). Managers must be wary of blanket claims that heavy grazing improves mule deer habitat and guard against this being used as an excuse for overgrazing. In reality, improvements can only be made through strictly manipulated timing of grazing specifically for this purpose (Severson and Medina 1983), based upon a carefully crafted management plan.

Timing and location of a treatment needed to improve mule deer habitat may not be in the best interest of the livestock operator from a financial standpoint (Longhurst et al. 1976). Severson and DeBano (1991) showed that goats could be

Figure 8. Riparian corridors, similar to this one in central Nevada, make up a small proportion of the land area, but are vitally important to wildlife for the resources they provide and to facilitate landscape connectivity. (Photo by Mike Cox/NDOW).

Figure 9. Riparian corridors are extremely important habitat features for mule deer so grazing plans must provide for their protection. (Photo by Mike Cox/NDOW).

used to reduce shrub cover in central Arizona, but the shrub species reduced were the ones favored by deer. This emphasizes the need to be extremely careful when planning efforts to improve deer habitat using livestock as tools.

Implementing multi-species grazing systems to benefit mule deer is challenging, but with collaboration and cooperation between land and wildlife managers success can be achieved. Increasing threats to rangelands and mule deer winter range are rapidly occurring from urban and exurban development (Maestas et al. 2002). Mule deer and their habitats will fare much better in landscapes dominated by traditional agricultural ranching operations and the open spaces they maintain. Therefore, it is critical that natural resource and ranching interests to work together in conserving rangelands and wildlife habitat.

Guidelines

It is recognized that public land managers follow various rangeland assessment and management protocols under federal policy. While most of the guidelines should be consistent with these protocols, others, because they specifically focus on optimizing mule deer habitats, may be beyond the scope of federal policies.

A. Grazing Plan

Grazing should always be done under the direction of a grazing management plan that provides for adaptive management and considers provisions outlined in The Wildlife Society's (2003) position statement regarding livestock grazing on federal rangelands. The overall goal of a grazing plan should be based upon maintaining appropriate ecosystem functions. Healthy rangelands benefit both wildlife and livestock.

1. In the IMW, identified goals and management actions need to
   - Maintain or increase density, vigor, cover, and diversity of vegetation species, particularly native perennial grasses and forbs.
   - Consider mule deer browse density, vigor, and productivity.
   - Decrease exotic (e.g., cheatgrass, tumble mustard [*Sisymbrium* spp.]) and increaser species (e.g., rabbitbrush), while increasing native palatable species.
   - Increase in health of riparian areas (see below).
2. Managers should develop grazing plans in cooperation

with rangeland management specialists familiar with local vegetation associations. Guidelines developed in one habitat type may not be completely applicable in another.

3. If the plan covers a ranch that includes several administrative agencies, include the entire ranch in a coordinated ranch management plan. A coordinated plan might allow greater flexibility to rotate seasonally between pastures and to rotate season of use of pastures annually.
4. The plan and any associated rotational system should be flexible enough for the landowner, permittee, or land management agency to adapt to changing environmental conditions.
5. Develop a contingency plan for reaching maximum utilization level, particularly in drought conditions. Drought is defined as "prolonged dry weather, generally when precipitation is less than 75% of average annual amount" (Society for Range Management 1989). Using this criterion for the city of Elko, NV, over the 118-year period of 1888-2005, drought occurred in 25% of the years (Western Regional Climate Center website 2008).
6. Management of riparian areas must be carefully planned (Elmore and Kauffman 1994). In these environments, timing of grazing may be more important than overall stocking rate.
7. Use classes of livestock that are least apt to impact preferred deer dietary items

B. Stocking Rate

1. Maintain stocking rates in IMW at levels below the long-term capacity of the land. Because of dramatic environmental fluctuations, stocking at full capacity results in overuse in approximately ½ the years and may necessitate supplemental feeding or liquidation of livestock. Martin (1975) concluded the best approach would be stocking at ≤90% of average proper stocking, but with some reductions during prolonged severe drought.
2. Steep slopes, areas of extremely dense shrubs, and lands distant from water sources should not be considered when calculating grazable land area (Fulbright and Ortega 2006). Holechek et al. (1998) recommend that land with slopes between 11% and 30% be reduced in grazing capacity by 30%, slopes between 31% and 60% - reduced by 60%, and slopes >60% be deleted from the grazable land area. Also, they suggested areas 1-2 miles from water be reduced in grazing capacity by 50% and lands >2 miles from water be deleted from the grazable land area.
3. To facilitate comparison of stocking levels between ranches in similar areas, stocking levels should be clearly stated in uniform terms. Stocking levels should be given in terms of "head per square mile yearlong," using only capable and suitable acres for the calculation of area in the allotment.

C. Utilization Rates and Stubble Heights

1. Utilization rate is closely related to stocking rate. Reduction of utilization can usually be accomplished by simply reducing the stocking rate accordingly.
2. Consider timing of grazing; even light stocking rates in some vegetation associations (e.g., riparian) can be detrimental if grazing occurs at the wrong time of year.
3. Annual monitoring of plant production and grazing intensity is essential for proper management of rangeland resources. Some monitoring programs are labor intensive, but rangeland can be evaluated with more qualitative guidelines such as those outlined by Holechek and Galt (2000, Table 1).
4. Manage for utilization rates of 25-35% of annual forage production in low sage and 30-40% use in pinyon-juniper, mahogany, mountain brush, and mixed conifer stands (Table 2).
   These utilization rates were developed for optimal livestock management; cattle utilization rates to optimize mule deer habitat quality would be at the lower end of these ranges (Lyons and Wright 2003).
5. Avoid heavy grazing (>50% averaged over the whole area) (Table 1). Depending on topography, there might be some tolerance of heavy use on up to 30% of the grazable land, but immediate reduction in livestock numbers is needed anytime use on >33% of the area is classified as severe (Holechek and Galt 2000).
6. Avoid heavy use of the same areas year after year (Table 1, Holechek and Galt 2000).
7. Consider residual vegetation height when evaluating intensity of grazing, rather than simply the percentage of annual herbage removed (Hanselka et al. 2001).
8. Livestock should not be allowed to browse >50% of the annual leader growth (by weight) of woody shrubs, which equates to approximately 50% of the leaders browsed (Holechek and Galt 2000, Table 3).

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 chapter).
2. Livestock and elk herds are attracted to newly treated areas, 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:
   - Where the threat of invasive annual plants is not an issue, pastures should be rested 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.

Table 1. Qualitative characteristics of grazing intensity categories (from Holechek and Galt 2000).

Qualitative Grazing Intensity Category	Use of Forage (% by weight)	Qualitative Indicators of Grazing Intensity
Light to non-use	0-30	Only choice plants and areas show use; there is no use of poor forage plants.
Conservative	31-40	Choice forage plants have abundant seed stalks; areas > 1 mi. from water show little use; approx. ⅓ to ½ of primary forage plants show grazing on key areas.
Moderate	41-50	Most accessible range shows use; key areas show patchy appearance with ½ to ⅔ of primary forage plants showing use; grazing is noticeable in zone 1-1.5 mi. from water.
Heavy	51-60	Nearly all primary forage plants show grazing on key areas; palatable shrubs show hedging; key areas show lack of seed stalks; grazing is noticeable in areas > 1.5 mi. from water.
Severe	> 60	Key areas show clipped or mowed appearance (no stubble height); shrubs are severely hedged; there is evidence of livestock trailing to forage; areas > 1.5 mi. from water lack stubble height.

Table 2. Recommended grazing utilization standards for IMW ecosystems (based on Holechek et al. 1998:207).

Representative Vegetation Types	Annual Precipitation (in.)	Utilization Maximum on Poor Ranges or Ranges Grazed in Growing Season (%)*	Utilization Maximum on Good Ranges Grazed in Dormant Season (%)
Low Sage	< 12	25	35
Pinyon-Juniper, Mahogany	10-21	30	40
Bitterbrush, Snowbrush, Snowberry, Mountain Big Sage, Mixed Conifer	16-50	30	40

** If a pasture is used during the growing season, no use is allowed during other times of that year (i.e., livestock cannot be returned the pasture later that same year).*

Table 3. Grazing intensity guide for key shrubs (common winterfat [Krascheninnikovia spp.], fourwing saltbush [Atriplex canescens], and mountain mahogany [Cercocarpus spp.]) (from Holechek and Galt 2000).

Qualitative Grazing Intensity Category	Use of Current Year Browse Production (% by weight)	Leaders Browsed (%)
Light to non-use	< 30	< 15
Conservative	31-50	16-50
Moderate	51-75	51-80
Heavy	76-90	81-100
Severe	> 90	100, plus old growth used

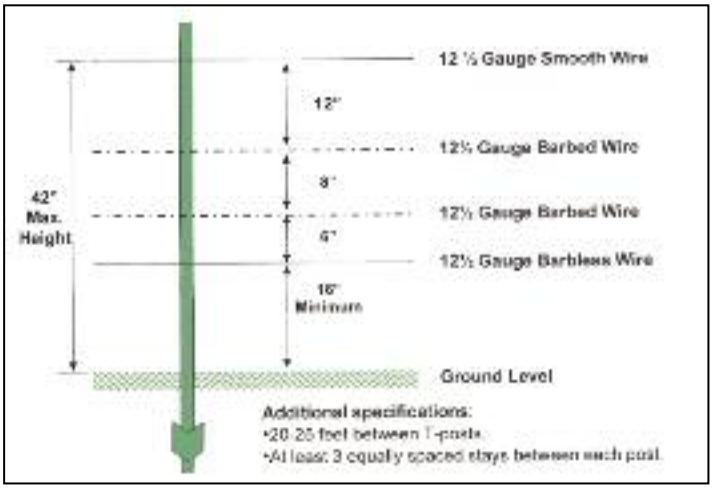

Figure 10. Specifications for a 4-strand, wildlife-friendly fence. Modification to existing fences can be accomplished by either removal of the bottom wire of an existing 4-strand fence or replacement of the bottom wire with a smooth wire that is ≥16 inches off the ground allowing for deer fawn (and pronghorn) movement.

- In areas with high deer and elk densities, shrub establishment may require planting seedlings using nursery stock. In extreme cases, temporary high fence may be required to exclude wild ungulates (in addition to livestock) until shrubs are successfully established

E. Fencing

1. Construct fences to allow wildlife passage (Fig. 10). Wildlife-friendly fencing will save livestock operators money by reducing fence repairs, particularly in areas with elk.
2. Remove or replace fences that are not wildlife-friendly.
3. 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.
4. 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.

Non-native Invasive Species

Background

Invasions of non-native plant species have caused widespread damage to natural systems throughout the IMW ecoregion (Beck 1993). Several negative ecological impacts are associated with infestations: displacement of native plants; reduction in biodiversity; alteration of normal ecological processes such as nutrient and water cycling; increased soil erosion; increased stream sedimentation, and alteration of fire regimes. In addition to negative impacts on natural systems, invasive, non-native plants cause substantial economic losses to agricultural interests (Pimentel et al. 2005) and reduce recreational values. Because of the insidious nature of non-native plant invasions, negative effects often go unnoticed until damage is severe, sometimes entailing nearly complete conversion of native habitats.

Although many invasive, non-native species are present in the IMW, some have proven more problematic because of the extent of infestations across large-scale landscapes, their ability to invade diverse native plant communities, and their impacts to quality and quantity of more desirable native plant species. Seven species of invasive, non-native plants are widespread in the IMW and significantly impact mule deer habitat: cheatgrass (Figs. 11-14), diffuse knapweed (*Centaurea diffusa*), spotted knapweed (*Centaurea biebersteinii*), yellow star-thistle (*Centaurea solstitialis*), leafy spurge (*Euphorbia esula*), rush skeletonweed (*Chondrilla juncea*), medusahead (*Taeniatherum caput-medusae*), and salt cedar (*Tamarix pentandra*).

Issues and Concerns

Via direct or indirect impacts, infestations of invasive, non-native plants can have significant impacts to native plant communities, wildlife habitat, and wildlife species supported by those communities. Although environmental damage caused by invasive plants is well-recognized, explicit knowledge regarding the full impacts of invasive plants on mule deer is lacking. As Beck (1993) stated, "The weed science community has spent a lot of time learning how to control weeds v. understanding their biology, ecology, and impacts." Invasion by non-native plant species in native plant communities results in changes in structure, species composition, and functional dynamics of those communities. These changes can reduce quantity and quality of mule deer forage, alter thermal and escape cover, reduce water availability, alter distribution of mule deer on the landscape, and concentrate mule deer on remaining non-infested areas, resulting in over-utilization of critical habitats such as winter range. For example, since the 1960s, Nevada has experienced extensive cheatgrass invasions that have resulted from and fueled wildfires unprecedented in size and intensity that in turn, have caused widespread loss of sagebrush-dominated habitats (Fig. 12). Between 1999 and 2001, a deer herd area in northeastern Nevada lost >660,000 acres to fires. In 2006, >610,000 acres burned in this same area, including one fire that exceeded 245,000 acres. Crucial mule deer winter ranges in the southern portion of this herd area were reduced from 184,320 acres in the early 1960s to <20,000 acres in 2007. Commensurate with the habitat loss, mule deer numbers in

this herd area declined due to substantial increases in both fawn and adult winter mortality. The population estimate for this deer herd in 2008 was approximately 1/5 of the 30,000 mule deer present in the 1960s and data strongly suggest habitat loss related to fire and cheatgrass invasion was the primary cause (Cox 2008).

Cheatgrass is particularly problematic and has had substantial impacts on rangelands and associated wildlife habitats in the IMW (Fig. 13). Cheatgrass, named for its ability to "cheat" other plants of water and nutrients, increases fire frequency (Whisenant 1990) and out-competes seedlings of native perennial plants (Reichenberger and Pyke 1990). Cheatgrass, native to Asia, has an entirely different phenology than most native plant species: it germinates much earlier and matures and cures earlier. Mack (1981) estimated cheatgrass occupied ≥41 million hectares in the western United States and considered it the dominant herbaceous plant in the IMW. In the shrub-steppe habitat of the Great Basin in Idaho and Utah, near monocultures of primarily cheatgrass along with other invasive plants exist on ≥5 million hectares (Whisenant 1990).

Two main impacts to mule deer habitats occur as a result of cheatgrass invasion. First, cheatgrass eliminates native perennial grasses and forbs that are more palatable and nutritious. Second, cheatgrass increases frequency and intensity of wildfires that destroy native shrublands, which are critical to mule deer diets and cover needs (deVos et al. 2003). Prior to invasion by cheatgrass in low-elevation sagebrush-bunchgrass coummunities, Billings (1994) states wildfires were rare and Young and Evans (1981) reported a 90-year fire interval , which allowed time for shrubs to reestablish. Currently, cheatgrass infested areas may burn as frequently as every 6–10 years. Also, because wildfire intensity is much greater, root systems and seed banks are "sterilized" so native plant recovery is more difficult. This new fire regime significantly impacts diversity and composition of native plant communities and associated wildlife habitat values (Fig. 14). Shrubs and other plants critical to mule deer populations have been reduced or removed altogether. This accelerated wildfire cycle has eliminated extensive stands of antelope bitterbrush, a preferred mule deer forage, in the northwestern Great Basin in Idaho, Oregon, Nevada, and California (Monsen and Shaw 1994).

Although not typically implicated in changes in fire regimes, other invasive plants have significant impacts on mule deer and their habitats. Stalling (1998) reported spotted knapweed invasion reduced deer and elk forage by 70% on parts of Theodore Roosevelt National Park in North Dakota. Native ungulates generally do not consume spotted knapweed or use it only rarely; Guenther (1989) did not detect knapweed in diets of mule deer, even though it was common on mule

Figure 11. Moderate-level invasion by non-native plant species into mule deer winter range in Twin Falls County, ID. Cheatgrass infestations are not dense; however, cheatgrass is distributed throughout understory. (Photo by Mark Fleming/IDFG).

Figure 12. Wildfire-killed sagebrush plants (skeletons) surrounded by invading cheatgrass prevents understory plants and sagebrush seedlings from establishing. (Photo by Mike Cox/NDOW).

Figure 13. High-level invasion by non-native plant species into mule deer winter range in Twin Falls County, ID. Cheatgrass dominates the understory; native grasses are present, but uncommon. (Photo by Mark Fleming/IDFG).

Figure 14. Severe-level invasion by non-native plant species into former sagebrush-dominated rangeland in Jerome County, ID. Area has burned multiple times and is dominated by cheatgrass and other invasive non-native plant species. (Photo by Mark Fleming/IDFG).

deer range in Montana. Further system degradation and destabilization result from infestations. Spotted knapweed infestations on hillsides increased runoff by 56% and sediment yield by 192% as compared to adjacent hillsides covered with native bunch grass (Lacey et al. 1989). Thus, invasive plants reduce potential nutrition and habitat value for mule deer through several avenues.

Human-caused disturbances such as fire and improper livestock grazing management practices have contributed to an accelerated spread of invasive, non-native plant species in the IMW. However, absence of fire or livestock grazing does not assure protection from invasion by non-native plant species (Frost and Launchbaugh 2003). Non-native plants are capable of invading plant communities without human assistance. For example, diffuse knapweed invaded a bluebunch wheatgrass (*Pseudoroegnaria spicata*) community in western Montana in the absence of grazing (Lacey et al. 1990). Likewise, spotted knapweed has invaded plant communities that had not been defoliated, and moderate defoliation did not accelerate the invasion process (Sheley and Jacobs 1997).

Magnitude of impact of invasive, non-native plant species on mule deer depends on the ecological significance of impacted areas to mule deer and the extent of infestations. Determining ecological significance of any given habitat requires site-specific knowledge about mule deer populations and habitat use. Ecologically significant habitats will include, but are not limited to, important fawning habitats and winter ranges.

Guidelines

A. The Management Plan

An initial inventory of habitat condition to determine presence and abundance of invasive plant species must be made. Invasive plant species of concern should be identified and prioritized according to their perceived threats to mule deer habitat. Throughout the IMW, 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.

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. 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 and land managers 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 and systems, appropriate stocking rates, and altering season of use.
2. Feed livestock only certified weed-seed-free hay or forage prior to entering and while within an area of concern.
3. Require motorized vehicles be cleaned prior to entry into areas of likely non-native species invasion.
4. Limit or prohibit activities that result in soil disturbance.
5. Evaluate road and trail systems. Close non-essential roads and trails.
6. Use a variety of mechanical, cultural, chemical, and biological (i.e., insects or fungi) control methods to reduce threats of invasive plant species and improve habitat for mule deer.
7. 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.

8. To specifically control cheatgrass and not harm preferred plants and seedlings, utilize approved and effective herbicides at the appropriate time, rate, and distribution in relation to local site conditions and management goals (Fig. 15,Vollmer et al. 2007).
9. Quickly rehabilitate rangelands impacted by wildfire or other disturbance during the first fall or winter post-disturbance. The key is to successfully establish perennial seedlings that will compete with invasive annual species before they dominate the site during the first year post-disturbance. (Fig. 16).
10. Use native and non-native seeded species that will reduce dominance of cheatgrass (e.g., crested wheatgrass [*Agropyron cristatum*] will compete well with cheatgrass and forage kochia [*Kochia prostrata*] will establish in the presence of cheatgrass and provide forage for mule deer). When seeding crested wheatgrass, be careful to use an appropriate seeding rate because under specific site conditions it can dominate and retard native shrub germination. Ideally, managers should proactively develop native seed sources that can compete with cheatgrass and other invasive plants.
11. 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.
12. Consider the potential for non-native plant invasions before new disturbances such as road construction, mineral development, prescribed fire, and recreational activities.
13. Support and implement new research and methods to reduce prevalence of cheatgrass in critical mule deer habitat.
14. Support efforts by public land managers that require certified weed free hay for feeding livestock on public lands.
15. 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.

Successional Changes

Background

The impact of plant succession on mule deer habitat in the IMW varies with a number of correlated factors including elevation, climate, soils, and ultimately, vegetation type. Higher elevation habitat types in the IMW are primarily composed of deciduous and coniferous forests. Non-riparian deciduous forests are typically a monoculture of quaking aspen (*Populus tremuloides*), whereas coniferous forests are composed of ponderosa (*Pinus ponderosa*)-Jeffrey pine (*P.jeffreyi*), Douglas-fir (*Pseudotsuga menziesii*)-white fir (*Abies concolor*), spruce (*Picea* spp.)-fir, lodgepole pine (*Pinus contorta*), or mixed conifer stands. Descending in

Figure 15. Top photo depicts a mid-May post-treatment site following a fall application of Plateau® herbicide the previous fall on a dense stand of cheatgrass. Bottom photo is the same treated area in late June of the same year with productive "release" of perennial grasses (primarily Stipa comata). (Photos by Keith Schoup/WGFD).

Figure 16. Successful establishment of native and nonnative plants from a post-wildfire seeding treatment; Two-year old seedlings from left to right: bitterbrush, sagebrush, and forage kochia (photo by Mike Cox/NDOW)

elevation, the primary vegetation types shift to other mountain shrub species (i.e., mountain mahogany (true [*Cercocarpus montanus*] and curl-leaf [*C. ledifolius*]), bitterbrush, serviceberry [*Amelanchier* spp.], and snowberry [*Symphoricarpos* spp.]). The lowest vegetation communities, which typically serve as mule deer winter range, are primarily composed of P-J woodlands, sagebrush, bitterbrush, or salt-desert (i.e., saltbush and cliffrose [*Purshia mexicana*]) shrublands.

Many of the deer in this ecoregion migrate between relatively moist higher elevation, summer range habitats and lower, drier, foothill or basin wintering areas (Carpenter and Wallmo 1981, Kie and Czech 2000). In most of the IMW, 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 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 noted by Carpenter and Wallmo (1981), throughout much of the IMW, 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 desert portions of the IMW where aspen and mountain shrub communities are limited. While there is less evidence indicating transitional ranges play a limiting role for mule deer in the IMW, 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, they have inherently established themselves and have thus out-competed other plants for resources. However, even when dominant plant types are highly useful to mule deer, overall contribution to their body condition may not be positive (Fig. 17). 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 2004).

Both vegetation and deer can respond positively to disturbance. Shepherd (1971), concluded at moderate removal rates (20-30%) of current annual growth, browsing was invigorating and decreased leader die-off. He also found serviceberry, antelope bitterbrush, big sagebrush (*Artemisia tridentata*), and mountain mahogany could sustain even higher removal rates. But depending on the amount of timely moisture, plant age, and cumulative years of browsing, consistent removal rates >40% greatly diminish the plant's ability to set seed and restrict recruitment of young plants.

Figure 17. Though mule deer use this bitterbrush/sagebrush stand (upper photo) which provides good cover and snow intercept during the winter and spring months, the majority of it (lower photo) is in late-seral stage, overgrown, and exhibits severely reduced forage quality and available leader growth, along with reduced understory productivity. (Photos by Mike Cox/NDOW).

Figure 18. Understory productivity typically diminishes over time in Pinyon-Juniper woodlands such as this area near Lander, WY. (Photo by Carrie Dobey/WGFD).

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 by shading. As P-J stands reach late seral stages, their value as cover increases, but understory vegetation is drastically reduced by shading effects and reduced water availability (Fig. 18). Late seral Gambel oak (*Quercus gambelii*) and mountain shrub communities can become so dense that deer movement is restricted and forage production and available leader growth are reduced (Fig. 17). However, older and taller sagebrush plants can also function as wind and snow breaks, thus providing refuge from harsh winter conditions and breaking up snow pack, which enhances foraging efficiency (Fig. 17). Late seral-stage sagebrush habitat can also out-compete surrounding vegetation, resulting in little or no understory growth.

As plants mature, their quality as forage for mule deer generally declines (Wallmo 1978, Stevens 2004, Wasley 2004). During early, pre-senescent stages, the majority of current annual growth occurs as leaders (Short and Reagor 1970). For mule deer, leaders are more digestible because they have thinner cell walls and less cellulose and therefore, are highly preferred over other plant parts (Wasley 2004). As plants age, 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, and 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 IMW pertains to encroachment of P-J forests into surrounding areas (Fig. 19). Juniper and pinyon occupy >30 million hectares in the IMW (West 1999), growing in a broad array of environments. Western juniper (*Juniperus occidentalis*), the northern variant of the P-J cover type, occupies 3.2 million hectares in eastern Oregon, northeast California, southwest Idaho, and northwest Nevada. Post-settlement juniper woodland expansion in the West has been most frequently attributed to introduction and overstocking of livestock, reduced role of fire, and optimal climatic conditions during the late 1800s and early 1900s (Burkhardt and Tisdale 1976, Heyerdahl et al. 2006, Tausch 1999).

Figure 19. Three comparative photos of the same site in Stansbury Valley, UT depicting the P-J encroachment into sagebrush shrublands. Top photo taken in 1901, middle photo in 1976, and bottom photo taken in 2008. (Photo by Tom Becker/UDWR).

Many juniper and pinyon species in the IMW are relatively long lived (>1,000 years and >600 years, respectively). However, depending on location, 66-90% of these communities are <130 years old (Miller and Rose 1999).

Although mature P-J forests provide high quality cover for mule deer, expansion of these forests into surrounding grass and sagebrush communities leads to further reduction of browse. As P-J 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 P-J 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 P-J expansion and thinning follow an ebb and flow pattern that is strongly related to drought and pluvial periods.

Similar to P-J forests, mature sagebrush can also greatly reduce understory vegetation. Encompassing a large proportion of deer winter range in the IMW, 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 (Smith 1950, Leach 1956, Welch and Andrus 1977, Young and Clements 2002). 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 sagebrush, important shrub species on winter range in the IMW include serviceberry, bitterbrush, mountain mahogany (both true and curl-leaf), cliffrose, four-wing saltbush, and winter fat. Important forbs or forb-like plants include buckwheat (*Eriogonum* spp.), fringed sagebrush (*Artemisia frigida*), and phlox (*Phlox* spp.). Useful grasses include blue grama (*Bouteloua gracilis*), native wheatgrass, fescue (*Festuca* spp.), and bluegrass (*Poa* spp., Table 4).

Habitat treatment efforts typically focus on increasing abundance of desirable plants or reducing abundance of undesirable plants. Dependent upon the primary objective, different habitat improvement techniques should be used accordingly (Monsen 2004) and include fire, harvest treatment, chemical treatment, and mechanical disturbance. Not all treatment methods are useful in all habitat types.

Fire was a natural occurrence across the landscape prior to Euro-American settlement, however its current presence (whether natural or artificial) is seldom tolerated. Nevertheless, fire still has a role primarily at higher elevations with little or no human development. Prescribed burning, where feasible, is usually the method of choice. When properly implemented, prescribed burns mimic natural disturbances and enhance natural processes

Table 4. Common, native winter and transition range shrubs, forbs, and grasses used by mule deer in the IMW. List compiled from Kufeld et al. (1973), Carpenter et al. (1979), Milchunas et al. (1978), and Bartmann (1983). Scientific names are provided in Appendix B.

Shrubs	Forbs or Forb-Like	Graminoids
Big sagebrush	Aster	Indian ricegrass
Serviceberry	Sagewort	Needle and thread
Mountain mahogany	Phlox	Basin wildrye
Snowberry	Snakeweed	Sandberg bluegrass
Rabbitbrush	Cryptantha	Blue grama
Bitterbrush	Globemallow	Bottlebrush squirreltail
Gambel oak	Buckwheat	Junegrass
Rose	Penstemon	Needlegrass
Chokecherry	Fringed sagebrush	Idaho fescue
Aspen	Goldenweed	Bluebunch wheatgrass
Ponderosa pine	Arrowleaf balsamroot	Western wheatgrass
Cliffrose	Lupine	Mutton grass

such as nitrogen and carbon cycling. An alternative to controlled burning in forested habitats is application of timber 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.

Figure 20. Various mechanical treatments such as the Lawson aerator (upper photo) and Dixie harrow (lower photo) can help restore shrub vigor and/or reset late-seral stage shrub and P/J communities to early seral communities with young shrubs and increased understory of grasses and forbs. (Photos courtesy of Kevin Hurley/WGFD and Kreig Rasmussen/USFS-Fishlake)

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 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 (Fig. 20). 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.

To implement a successful treatment one needs to consider the following: introduction of undesirable plants, site potential, moisture regime, treatment scale, design, and juxtaposition. A major concern is invasion of undesirable plant species following treatment. In the IMW, 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. 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 cover.

Figure 21. Example of appropriately scaled treatment area recommended for mule deer with high amount of edge adjacent to remaining P-J stands. Both a big Dixie harrow and bobcat brush saw were used to open dense P-J canopy. (Photo by Kreig Rasmussen/USFS-Fishlake).

Guidelines

To positively influence and change 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 IMW, 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 piecemeal 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 across a broad landscape 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 goal 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 (approx. 5 acres) would receive more use than larger areas (≥25 acres) (Fig. 21).
4. Consideration of competition Treatments should not be considered in areas where they are likely to receive detrimental ungulate use during the initial revegetation phase. Although some grazing can be beneficial (e.g., salting oak brush so cattle will break it down; using domestic sheep or goats to help control noxious species), the unintended grazing and browsing of desirable seedling plants before they become established and vigorous can reduce deer use to less than pre-treatment levels.

B. Treatment Delivery

Regardless of primary treatment type there are several key aspects of implementation 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 inches of annual precipitation, reseeding may not be imperative, but might improve the treatment effect. In a best-case scenario, 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 and beneficial non-native species, preferably seed from sites with similar conditions, should be used when reseeding. Use of a seed mix increases community structure and function, initiates natural succession processes, increases probability of success, improves ground cover and watershed stability, and increases habitat diversity (Stevens 2004). Non-invasive, non-native forbs (particularly nitrogen-fixing legumes) with high palatability (e.g., alfalfa [*Medicago sativa*], small burnet [*Sanguisorba minor*], and sainfoin [*Onobrychis viciifolia*]) can also be used along with native species. Non-native grasses (e.g., crested

wheatgrass, smooth brome [*Bromus inermis*], orchardgrass [*Dactylis glomerata*]) should only be used 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. Prior to treatment, a seed mixture of pure live seed (PLS) should be in hand and tested for quality. Seeds of some common native grass and forb species are commercially available (Jorgensen and Stevens 2004). Date, method, depth of seeding, germination rates, and compatibility of different species should also be considered (Monsen and Stevens 2004, Stevens and Monsen 2004). 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 therefore need to be distributed at different rates (Stevens 2004).

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, fourwing saltbush, and bitterbrush, browse seed is often not as readily available as seed for some grasses and forbs.
4. Where commercial seed collection operations occur on public lands, permits should require that an adequate amount of seed is left for shrub seedling recruitment within the harvested stand. Also, "nursery plots" of shrub species whose seed is difficult to acquire are recommended.
5. Road avoidance - Treatment areas should be well screened from roads whenever possible by leaving trees and shrubs along travel corridors. Roads into treatment areas should be blocked whenever possible.

C. Post-Treatment Assessment

1. 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 or herbicide application to control undesirable species.

Shrubland Integrity

Background

Fire historically played a primary role as a disturbance factor in shrub ecosystems (Daubenmire 1968, Burkhardt and Tisdale 1976, Gruell 1985, Eddleman and Doescher 1999, Miller and Eddleman 2000). Nevertheless, shrub-steppe habitats can be affected by a variety of other factors including insects, rodents, climatic changes, grazing, and disease (Champlin and Winward 1982, Wright and Bailey 1982, Hironaka et al. 1983, Crane and Fisher 1986, Kauffman 1990, Young 1990, Peterson 1995, Tart 1996, Miller and Eddleman 2000, Paysen et al. 2000, Ryan 2000). Mule deer have evolved with fire that has impacted sagebrush stands with variable return intervals, depending on moisture regimes, topography, soils, and plant communities (Bunting et al. 1987). The natural disturbance elements are varied in these environments. Fires are inevitable wherever sufficient fuels accumulate. Ignitions and conditions suitable for ignition may or may not be limiting factors.

There are competing theories on how often fire historically burned these ecosystems (Winward 1991, Welch and Criddle 2003, Baker 2006). Some scientists believe pre-settlement fires may have occurred every 100 to 200 years in low sagebrush (*Artemisia arbuscula*) community types (Young and Evans 1981, Miller and Rose 1999) and 30 to 110 years in Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) community types (Young and Evans 1981, Winward 1991, Wright and Bailey 1982). In more mesic sagebrush types characterized by mountain big sagebrush (*Artemisia tridentata vaseyana*), fire return intervals have been reported to occur between 12 and 25 years (Houston 1973, Burkhardt and Tisdale 1976, Miller and Rose 1999).

Other scientists believe wildfires have been a relatively uncommon event in many sagebrush environments, including most Wyoming big sagebrush communities (Connelly et al. 2000, Nelle et al. 2000, Baker 2006). Baker (2006) provides an analysis of fire frequency in sagebrush communities, which suggests fire rotation may be much longer than previously reported. He indicates that fire rotation in low sagebrush may be a minimum of 325-450 years, 100-240 years in Wyoming big sagebrush and 70-200 years or more in mountain big sagebrush. Some of these plant communities can maintain themselves over time in the absence of disturbances such as fire (Lommasson 1948, Anderson and Inouye 2001, Welch and Criddle 2003).

Most sagebrush species have features that are poorly adapted to fire. Exposure to fire generally results in the death of the plants and these shrubs have poor seed dispersing mechanisms, which limits reestablishment of seedlings following large fires (Welch and Criddle 2003,

Cooper et al. 2007). Silver sagebrush (*Artemisia cana*), one of several exceptions, readily resprouts from the roots when the crown is killed by fire and, unlike big sagebrush, apparently evolved in areas where frequent fires shaped ecological processes (Adams et al. 2004).

Although competing theories exist as to how often sagebrush communities burned historically, it appears there is little question that the frequency and size of wildfires have increased dramatically in many parts of the IMW over the last 20 years and that these trends appear to be accelerating (Suring et al. 2005). The end result has often been a loss of many sagebrush dominated habitats (Connelly et al. 2004).

Issues and Concerns

In the late 1800s and early 1900s there were beneficial disturbances or events in IMW shrublands that contributed to mule deer irruptions. These disturbances were summarized by Gruell (1986): 1) succession of rangelands from dominance by grasses to dominance by woody plants that constitute superior mule deer habitat (Julander 1962, Leopold 1950, Longhurst et al. 1976); 2) conversion of forests to shrub fields by wildfire and logging, which generally resulted in improved deer habitat (Lyon 1969); 3) conservation and predator control dramatically reduced deer mortality (Leopold et al. 1947); and 4) reduction in numbers of livestock on the open range increased the amount of forage available to mule deer (Rasmussen and Gaufin 1949).

Various changes have occurred in shrub communities over the past century that have negatively affected mule deer and their habitats. These changes have taken on many forms which include: 1) invasion and dispersal of non-native plants; 2) removal of habitat due to the construction of housing developments, mining, oil-mineral development, and road building; 3) cumulative effects of livestock and feral horse grazing; 4) manipulation of these communities for agriculture and other forms of production; 5) pinyon-juniper encroachment, forest maturation, and fire suppression, 6) sagebrush removal activities, and 7) increased low-elevation wildland fires.

Livestock use and fire suppression have led to less productive shrub community conditions (Anderson 1958, Bennett 1999). Anderson (1958:26-27) reported browsing of shrub communities in Wyoming was heavy, and stated: "It is very alarming … to note that each of the areas examined and reported on here exhibit definite signs of range deterioration. This vegetation deterioration is in the incipient stages in some areas; in other areas is much more serious and has progressed to the point where recovery will be a long, slow process." He continued, "It is suggested the ultimate goal be to maintain game herds at a level where average winter mortality is kept at a minimum, average annual forage utilization falls within the proper limits, and vegetative trends are stabilized or are upward." Nonetheless, based on others' examination of these sites at later dates (1960s and 1970s), even further declines in shrub conditions had taken place. Three of the more commonly recognized changes in shrublands of the IMW include loss of herbaceous understory species (grasses and forbs), conversion to invasive species dominated habitats, and decadent browse resulting from a lack of disturbance.

Mule deer are a highly selective browser, very dependent on rumen microbes to derive energy from plant matter. As forage plants mature, their cell walls thicken. Parts contained within cells are up to 98% digestible (Short and Reagor 1970). Some of the cell wall constituents can be broken down by microbes in the rumen, while others cannot. Lignin, a non-carbohydrate polymer that binds the cell together, is indigestible. The older a plant becomes, the more cell wall material it contains, hence, the older a plant, typically, the less digestible. Additionally, older age plants typically possess greater amounts of chemical constituents that make the plant taste bad or smell bad in order to protect itself from herbivory. Finally, many of the preferred browse species lose vigor with age. Bitterbrush, in many places one of the most important browse species for mule deer (Hormay 1943, Nord 1965), exhibits decreased leader and seed production as it ages. At 60 years old, seed production and leader growth begin to decline (McConnell and Smith 1977). Not only does the lack of leaders present obvious problems for foraging mule deer, but the lack of seed production significantly reduces a plant's ability to replace itself or recruit new plants.

General Guidelines

A mix of seral stages should be maintained in a temporal and spatial mosaic. Vertical structure of the shrub community, regardless of seral stage, should be considered for those wildlife species of importance in the project area. Size, design, and positioning of treatments, as well as the analysis area itself, should be derived by consensus of local resource experts. Consideration should be given to species of special interest and management needs. Shrub ecosystems are inherently variable and recommendations may need to be adjusted for local conditions, considering differences in precipitation, soil types, and current community health and condition. General management guidelines and specific species guidelines include:

1. Promote a healthy, productive mosaic of shrub age classes and canopy covers with a diversity of plant species in sustainable sagebrush communities.
2. Maintain or restore important shrub communities.
3. Evaluate rehabilitation or restoration work following

disturbances focusing on immediate reestablishment of native vegetation species suited to local range sites.
4. Mitigate shrub ecosystem loss, fragmentation, or degradation.
5. Promote communication and cooperation between all entities involved in the management of sagebrush and other shrubland ecosystems.

Species Specific Guidelines

Antelope Bitterbrush and Desert Bitterbrush (*Purshia glandulosa*)

Bitterbrush occurs mostly on well-drained sites varying from sandy to rocky soils at elevations as low as 200 feet in Oregon and up to 9,000 feet in mountain ranges throughout the IMW. Cliffrose is a closely related shrub with evergreen, hairy leaves that grows in drier climates of the extreme southern portion of the IMW.

Annual reproductive and vegetative growth of bitterbrush at moderate elevations starts with leaf development in late March through late April, followed by flowering in late April to mid-May; leader growth after flowering through September, seed ripening and fall in mid-July, and leaf drop in the fall (Young and Clements 2002). Based on the timing of leaf drop and development and their added nutritional value relative to stems, it is surmised that mule deer use antelope bitterbrush primarily in the fall and again in the spring when leaves appear (Young and Clements 2002).

Antelope bitterbrush flowers on second year leaders or twigs. The amount of flowering is dependent on the previous spring's growing conditions and the past fall and winter browsing pressure. Seed mortality caused by insects during the spring flowering period can reach nearly 50% (Clements and Young 2007). Bitterbrush seeds are too heavy for wind dispersal and therefore, natural dispersal of seeds and regeneration of plants is closely related to seed caching or "scatter hoarding" of granivorous rodents (Clements and Young 1996, Hormay 1943, Nord 1965).

Bitterbrush seeds actually germinate in the winter under snow after undergoing a critical moist-prechilling treatment for ≥3-4 weeks at or just above freezing. Germination and root growth during winter, followed by early spring emergence, is an adaptation to elongate roots for moisture extraction, while at the same time avoiding rodent predation (Young and Clements 2002).

Restoration efforts have included raising bitterbrush seed in a nursery to develop 2-year old seedling plants for planting. Clements and Young (2000) found this method was expensive and only yielded 5% survival of planted seedlings. Extensive browsing by animals, competition for soil moisture with other plants, and lack of soil microorganisms contributed to poor success.

Useful mechanical treatment methods with the primary goal of increasing seedling establishment and not twig production on old plants include: roller-choppers, hydro-axes, flails, anchor chains, Dixie harrows, or brush beaters. Controlled burns conducted during early spring or fall to maintain moderate heat intensity allow for possible re-sprouting. Young and Clements (2002) observed burns conducted in May or June produced more re-sprouting, but also had higher plant mortality due to summer desiccation.

Sprouting of bitterbrush after fire can be anywhere from rare to abundant (≥25%) and is influenced by genetics,

Figure 22. The upper photo is senescent bitterbrush plant that lacks leader growth; the lower photo is a vigorous young bitterbrush plant < 20 years old that has extensive leader growth and provides palatable and nutritious browse for mule deer. (Photos by Mike Cox/NDOW).

physiological status, fire intensity, and soil moisture after the fire. Cliffrose and desert bitterbrush consistently sprout once the aerial portion of the plant is removed (based on clipping studies) and appear to be more adapted to sprouting after wildfires.

Bitterbrush provides between 8% and 14% crude protein to browsing mule deer depending on the season (Clements and Young 2007). Foliage production of bitterbrush peaks at approximately 60 years of age under early season livestock grazing (McConnell and Smith 1977). Stands >80 years old lack adequate seedling recruitment to recruit new, vigorous plants. Excessive "hot season" livestock grazing can greatly contribute to a lack of seed production and mortality of seedlings, perpetuating even-aged, decadent bitterbrush communities (Clements and Young 2001).

Guidelines

1. Assess bitterbrush community age structure or senescence at a geographic scale based on known areas of seasonal mule deer use to ensure a mixture of age and size classes; make observations of flowering and seedling establishment or lack of recruitment; identify percent of community >60 years old or plants showing signs of senescence; identify future timelines for treatment based on this assessment (Fig.22).
2. Only treat those bitterbrush stands where >40% of plants are >60 years of age or are decadent.
3. Reseed (with seed collected from the most recent summer seed production) treated stands and after wildfires and prescribed burns during the first fall. Seed in microsites or patches that have either enhanced soil moisture or lack plant competition at a depth of 1-2 inches (Young and Clements 2002). Seeding mechanically, where possible, or by hand on steep slopes.
4. Avoid "hot season" livestock grazing in bitterbrush communities where inadequate grass and forb understory densities and biomass exist, causing livestock to "switch" to browsing bitterbrush leader growth in July and August.

Curl-leaf Mountain Mahogany

Throughout the IMW, curl-leaf mountain mahogany stands are most abundant above 6,000 feet elevation, and occur as high as nearly 10,000 feet in central Nevada (Schultz et al. 1990). Regardless of elevation, sufficient winter and spring precipitation to support this evergreen shrub or small tree is essential. Curl-leaf mountain mahogany can grow on relatively shallow or deep loamy soil, (Scheldt and Tisdale 1970, Blackburn et al. 1969). In the Greater Yellowstone Ecosystem, curl-leaf mountain-mahogany grows most commonly on limestone soils (Marston and Anderson 1991), typically on south-facing slopes. Individual stands of this plant range in size from <1 acre to several hundred acres or more (Schultz 1987).

Curl-leaf mountain mahogany stands are quite variable (Schultz 1987, Schultz et al. 1990, Schultz et al. 1991). Some stands are near-monocultures, while others occupy open savannas associated with mountain big sagebrush (*Artemisia tridentata vaseyana*), other shrubs, and numerous species of perennial grasses and forbs (Fig 23).

Figure 23. Young curl-leaf mountain mahogany plants (in foreground) are found in association with mountain big sagebrush. (Photo by Brad Schultz/University of Nevada, Reno).

Figure 24. A decadent and high canopy curl-leaf mountain mahogany stand with very little seedling production due to seedling shade intolerance in central Nevada. (Photo by Brad Schultz/University of Nevada, Reno).

At low elevations, relatively fast growing pinyon pine can establish in mahogany stands and eventually crowd out this short-statured, shade intolerant species.

Curl-leaf mountain mahogany plants are long-lived and can reach >1,300 years of age (Schultz 1987, Schultz et al. 1990). Individual stands can have a mean plant age of >700 years (Schultz et al. 1991). Stands with a closed, or nearly closed, canopy often have few or no young plants in the understory (Fig. 24, Schultz et al. 1990, 1991), despite high seed density (Russell and Schupp 1998, Ibanez and Schupp 2002). Young plants are more common in stands with a savanna structure. Curl-leaf mountain mahogany is self-compatible for pollination, exhibiting high seed maturity and viability (Russell et al. 1998).

Deep litter throughout stands with high canopy cover appears to facilitate seed germination, but retards seedling survival due to poor contact between the root and the soil (Schultz et al. 1996, Ibanez and Schupp 2001). Reproduction in large stands with high canopy cover occurs most often in either canopy gaps with increased exposure of bare ground, or around the stand perimeter under sagebrush plants (Schultz 1987, Schultz et al. 1991).

Curl-leaf mountain mahogany can burn quite readily (Gruell et al. 1985, Ross 1999). Fire can easily remove all curl-leaf mountain mahogany from a stand. At best, it is a very weak sprouter. Not all mahogany stands, however, are susceptible to fire. Some are encircled by low sagebrush plant communities and have low or discontinuous fuels (Schultz 1987). Some stands appear to have burned numerous times, but with little or no mortality to existing trees (Arno and Wilson 1986, Schultz 1987). The influence and effect of fire is complex and appears to depend on landscape, stand, and environmental characteristics. Additional disturbance mechanisms causing substantial mortality are leaf defoliators affecting entire watersheds (Furniss et al. 1988), or sapsuckers (*Sphyrapicus* spp.) affecting stands near riparian areas (Ross 1999).

Curl-leaf mountain mahogany is very palatable and highly nutritious for mule deer. It is an especially important forage species for wintering mule deer. Domestic livestock, including sheep and cattle, will also occasionally use it (Mitchell 1951, Smith and Hubbard 1954). Once curl-leaf mountain mahogany is several years old, it appears to be very browse tolerant. Yearling plants have only 4-8 leaves and are often completely consumed in 1 bite (Scheldt and Tisdale 1970, Schultz 1987).

Guidelines

1. Thin stands to create canopy gaps to promote seedling survival and increase plant recruitment. Consider aerial drip torch ignitions just prior to a late-fall snow event to create a mosaic of openings favorable for seedling establishment.
2. Treated stands need to be large enough and spatially distributed so browsers do not concentrate on relatively few plants.
3. Maintain other shrub species to inhibit litter accumulation and protect mahogany seedlings.
4. Consider soil type and site potential when vegetation management treatments are planned and implemented to establish reasonable project expectations.
5. Remove conifers and consider soil disturbance/site preparation to promote seedlings and maintain curl-leaf mountain mahogany stands.
6. Conduct seedling plantings post-wildfire where entire stands are lost with little or no seed source remains to naturally restore the stand. Encourage research to identify effective methods in restoring curl-leaf mountain mahogany stands post-wildfire.

Serviceberry

Serviceberry in the IMW occurs on mountain slopes, hillsides, and riparian zones in well-drained and typically mesic soils, although local moisture regimes vary from moist to seasonally dry (Hemmer 1975) (Fig. 25). Within the IMW, serviceberry is usually found between 4,000 and 9,000 feet in elevation. A plant can reach a height of 15 feet in a form of a small tree. Serviceberry is slow growing, but once mature plants are established they are tolerant of short-term dry periods, but prolonged drought cycles may cause stress and even plant mortality. In Montana it does not occur on sites with less than 14 inches of annual precipitation (Hemmer 1975).

Carpenter et al. (1979) found that besides big sagebrush, the next most abundant shrub stems consumed by mule deer after grasses and forbs dried out, were serviceberry and snowberry. Serviceberry is often a primary component of winter mule deer diets (Martin et al. 1951, Martinka 1968, Plummer et al. 1968, Kufeld et al. 1973). A diet consisting solely of serviceberry can be fatal due to presence of cyanogenic glycosides (highly concentrated in young twigs and least concentrated in older leaves) (Quinton 1985). Serviceberry is deciduous with leaf drop typically in October, leaves formed in April, flowering in May and fruit ripening in July.

Serviceberry reproduces from seed, by sprouting from the root crown, rhizomes, and by layering (Bradley 1984). Serviceberry's primary response after a fire is to sprout from the root crown and/or rhizomes (Hemmer 1976, Bradley 1984). Seedling establishment is not an important post-fire regeneration strategy. Stickney (1986) found on 21 plots in a wildfire, 100% of serviceberry regeneration resulted from sprouting of burned plants. When the root crown is killed by fire, serviceberry sprouts from rhizomes further beneath

Figure 25. A serviceberry thicket in a big sagebrush/rabbitbrush community used by mule deer year-round (Photo by Mike Cox/NDOW).

Figure 26. Large (10 – 15 feet tall) decadent tree-form serviceberry plants lacking succulent leader growth and containing a large amount of woody stems. (Photo by Mike Cox/NDOW).

Figure 27. A healthy and productive snowberry plant in association with mountain big sagebrush prior to fruit development. (Photo by Mike Cox/NDOW).

the soil surface. Sprouting response has been greater after a spring burn vs. a fall burn (Noste 1982). Serviceberry in forested habitat types is fire-dependent and declines with fire exclusion and resultant canopy closure (Gruell 1983).

Guidelines

1. Prevent browsing by livestock and/or wild ungulates in excess of 50 percent of annual serviceberry growth; this level of browsing can be detrimental to the vigor and survival of the serviceberry plants.
2. Consider mechanical treatments or spring burns when serviceberry plants have matured to small trees in order to introduce more early-seral plants to the serviceberry thicket (Fig 26).

Snowberry

Snowberry is a deciduous, montane shrub that occurs on a wide variety of sites and aspects ranging from moist to fairly dry and in both acidic and basic soils (Fig. 27). It usually occurs in sandy loam to clay loam. Elevational range for snowberry in the IMW is from 5,000 to 10,000 feet. Although averaging 2 to 4 feet in height, plants on good sites can grow up to 5 feet, while those on poor sites are barely a foot tall. Snowberry phenological events above 7,500 feet are as follows: mid-June - full leaf out; end of June -full bloom; mid- to late-August – fruit ripe; mid-October – leave drop (Costello and Price 1939). Snowberry basal shoots have been observed growing in the soil before total snowmelt (Willard 1971).

Snowberry is an important forage species for mule deer on high elevation summer ranges (Plummer et al. 1968, Carpenter et al. 1979, Collins and Urness 1983, Beck and Peek 2005). The carbohydrate reserves for snowberry have been found to peak at full bloom in late June and continue high as plants become dormant in the fall (Donart 1969). Snowberry plants withstand browsing well and produce numerous basal sprouts following browsing (Willard 1971).

Snowberry will be top-killed by most fires of medium or high severity (Fischer and Clayton 1983). Snowberry will sprout from basal buds at the root crown following a fire from its root crown (Zschaechner 1985). Recovery from sprouting after severe fires may be variable. Even after severe fire, pre-fire plant frequency and canopy cover have been reestablished within 15 years (Blaisdell 1953, Zschaechner 1985). It is unclear which burn season is most advantageous to sprouting and restoring early-seral stages of snowberry.

Guidelines

1. Monitor early season livestock and/or wild ungulate browsing of snowberry to ensure adequate plant material exists for mule deer through the late summer period when most other plants have dried and are less palatable.

2. Conduct mechanical treatments, prescribed burns, or managed wildfires when a large portion of a snowberry community is decadent to introduce more early-seral plants (Fig. 28).

Snowbrush (*Ceonathus velutinus*)

Snowbrush ceanothus, the most common *Ceanothus* spp. in the IMW, is an evergreen shrub that grows 2 to 9 feet tall, and occurs from 3,500 to 10,000 feet (Fig. 29). Although snowbrush ceanothus grows in almost any soil, it grows best in medium- to coarse-textured, well-drained soils 20 to 60+ inches deep (Sutton and Johnson 1974). Snowbrush ceanothus has a single large taproot and a deep, spreading root system. The roots extend to depths of 6 to 8 feet and extend laterally past the crown of the plant (Curtis 1952) with nitrogen-fixing root nodules (Mozingo 1987). Viable snowbrush ceanothus seed can be stored in the soil for up to 200 years (Lackschewitz 1991). Snowbrush ceanothus seeds germinate in the spring. New leaf buds break as early as mid-April and leaves continue growth until early July. Flowering begins in May or June and fruit ripens from late June to early August (Schmidt and Lotan 1980).

Snowbrush ceanothus is a valuable year-round browse species for mule deer (Leach 1956, Kufeld et al. 1973, Lackschewitz 1991). Snowbrush ceanothus can be described as a seral dominant, becoming common after major disturbances, especially fire by regenerating from seed stimulated by “heat treatment” (Ruha et al. 1996). Where its seeds are present in the soil, snowbrush ceanothus may dominate early seral growth following a "medium or hot" fire. It also sprouts vigorously from the root crown after fire (Ruha et al. 1996). Fire creates conditions more favorable for snowbrush ceanothus growth by removing the overstory. Dry weather patterns following canopy removal and repeated severe fires are likely to produce persistent seral snowbrush shrubfields (Smith and Fischer 1997). In the IMW, pure stands of snowbrush ceanothus may form on south-facing slopes. A high-severity fall burn is more likely to produce a dense stand of snowbrush ceanothus than a "cooler" spring burn because spring burns produce fewer sprouts (Young 1981). Prescribed fires that do not burn hot may not stimulate seed germination, and therefore may not increase snowbrush ceanothus (Thompson 1990).

Guidelines

1. To moderately increase snowbrush density through sprouting, conduct a spring burn to minimize fire intensity.
2. Conduct fall burns in decadent snowbrush stands to reestablish early-seral plants through sprouting and seed germination.
3. To control greenleaf manzanita (Arctostaphylos patula) in snowbrush communities, repeated burning (2 burns within 5-years) has been shown to cause near complete mortality of both mature and seedling manzanita plants. Mechanical treatment could be used on remaining snowbrush plants to stimulate sprouting in conjunction with seeding or planting of preferred shrubs and herbaceous species adapted to site conditions.

Figure 28. A decadent snowberry plant; late-seral stage stands dominated by this growth form should be considered for treatment (Photo by Mike Cox/NDOW).

Figure 29. A mountain slope with snowbrush used by mule deer primarily as summer and transition range in the Sierra Nevada Mountains, Nevada. (Photo by Mike Cox/NDOW).

Wyoming Big Sagebrush (*Artemisia tridentata wyomingensis*)

Wyoming big sagebrush community types occupy relatively arid sites in the western United States and account for the largest area of the big sagebrush cover types (Tisdale 1994). This species commonly occurs from foothills to basins and valley bottoms (Dorn 1988). Wyoming big sagebrush tends to grow on shallower, well-drained, and xeric soils when compared to mountain and basin big sagebrush (*Artemisia tridentata tridentata*) (Barker and McKell 1983). Most of the

Figure 30. A vigorous Wyoming big sagebrush community exhibiting a diversity of shrub age-classes dominated by mid and late seral plants and a healthy understory of grasses and forbs on Chalk Mountain, Wyoming. (Photo by Justin Binfet/WGFD).

Figure 31. Wyoming big sagebrush community with a cheatgrass understory in Bates Hole, Wyoming where, perhaps, the only appropriate treatment would be an herbicide application to control cheatgrass to restore a native herbaceous understory. (Photo by Justin Binfet/WGFD).

Figure 32. A productive mountain big sagebrush stand dominated by mid- to late-seral stage sagebrush plants with a healthy herbaceous understory used by summering mule deer in the Trout Creek Mountains, southern Oregon (Photo by Mike Cox/NDOW).

Wyoming big sagebrush occurs in the 5-9 inch and 10-14 inch precipitation zones. Wyoming big sagebrush is a long-lived species, exceeding 150 years in undisturbed settings (Ferguson 1964). Plants averaged 42 years (range 26-57) at an undisturbed site in south-central Wyoming (Sturges 1977).

Increased fire frequency in lower elevation sagebrush habitats, often closely tied to invasion of annual grasses such as cheatgrass (*Bromus tectorum*), has resulted in losses of sagebrush over large expanses in the Intermountain West and Great Basin (Mack 1981, Miller et al. 1994, Crawford et al. 2004). In addition, decreased fire frequency in higher elevation sagebrush habitats and impacts from inappropriate livestock grazing and other factors have resulted in conifer encroachment, and subsequent reduction of the herbaceous understory and sagebrush canopy cover over large areas (Miller and Rose 1995, Miller and Eddleman 2001, Crawford et al. 2004).

Sagebrush overstory is both spatially and temporally diverse due to the extensive geographic region occupied by the sagebrush ecosystem in North America (Schroeder et al. 1999, Miller and Eddleman 2001). One reason for variation in sagebrush canopy cover among seasons is likely due to the dynamic nature of sagebrush stands. Sagebrush canopy cover is not static but changes both before and after the stand matures. In southwestern Montana canopy cover of Wyoming big sagebrush varied from 10.6 to 16.1% over an 18-year period (Wambolt and Payne 1986). Post-fire canopy and density recovery under optimal conditions may take 30-40 years (Young and Evans 1981, Winward 1991, Bunting et al. 1987), or may take well over 100 years (Baker 2006, Cooper et al. 2007).

Guidelines

1. On the landscape of interest, maintain at least 70% of sagebrush-dominated plant communities with a diversity of age classes emphasizing mid- to late-seral stages, and a healthy understory of native grasses and forbs (Fig. 30).
2. Use extreme caution or do not treat stands where cheatgrass or other invasive species are present (Fig. 31).
3. Do not treat those areas with thin topsoil and limited productivity.
4. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008).
5. Aggressively suppress stand-replacing wildfires where >30% of the landscape of interest may burn to protect intact Wyoming big sagebrush communities.

Mountain Big Sagebrush
(*Artemisia tridentata vaseyana*)

Mountain big sagebrush generally occurs on foothills, ridges, slopes, and valleys in the upper elevational range

of big sagebrush. Moderately deep and well-drained soils are typical of occupied sites (Beetle 1961). This subspecies grows well in full sunlight, but also tolerates shade and often occurs in association with conifers and aspen (Noste and Bushey 1987, Tart 1996).

Normally, mountain big sagebrush stands recover much more quickly following fire than do Wyoming big sagebrush stands (Baker 2006). The number of years to return to pre-burn density and canopy cover may vary: 15-20 years (Bunting et al. 1987); 15-30 years (Champlin and Winward 1982, Hironaka et al. 1983); and slightly more than 30 years (Cooper et al. 2007, Lesica et al. 2007). Beetle and Johnson (1982) indicated mountain big sagebrush self-replaces post burn. Rapid growing seedlings reach reproductive maturity at 3-5 years (Bunting et al. 1987).

From a landscape perspective, a portion of the terrain historically did not carry fire well. Examples of areas that did not burn are windswept ridge tops and sites with shallow soils where fine fuel production is limited.

Guidelines

1. On the landscape of interest, maintain at least 70% of sagebrush-dominated plant communities with a diversity of age classes emphasizing mid- to late-seral stages, and a healthy understory of native grasses and forbs (Figs. 32, 33, and 34).
2. Use extreme caution or do not treat stands where cheatgrass or other invasive species are present
3. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008) (Fig. 35).
4. Protect intact stands of mountain big sagebrush from stand replacing wildfires where >30% of the landscape of interest may burn. Take reasonable precautions and follow strict burn plan guidelines when conducting prescribed burns to introduce early-seral stage sagebrush (Figs. 36 and 37).
5. Do not treat those areas with thin topsoil and limited productivity.

Figure 33. When big sagebrush stands become unproductive with a large percentage of dead or decadent plants (upper photo), mechanical treatment, such as a Dixie Harrow with simultaneous overseeding of sagebrush (lower photo) can greatly benefit the stand to introduce sagebrush seedlings. (Photos by Kreig Rasmussen/USFS-Fishlake).

Figure 35. A mountain big sagebrush community in northern Nevada with a variety of herbaceous plant species that are an important nutritional component to mule deer summer diets (Photo by MikeCox/NDOW).

Figure 34. Series of photos on the same big sagebrush site in the Dixie National Forest, Utah, showing, results of a Dixie Harrow treatment to introduce more early seral sagebrush and healthier and diverse herbaceous understory. Left photo with no treatment; middle photo 6 years after a once- over treatment; right photo 6 years after a twice-over treatment. (Photos by Kreig Rasmussen/USFS-Fishlake).

Figure 36. A control burn applied in a sagebrush stand with >30% cover in Wyoming. Prescribed fire can be an effective tool in mountain big sagebrush communities at the appropriate scale to introduce younger-aged sagebrush plants and reintroduce herbaceous understory as long as the site has deep soils, adequate moisture regime, and threats of invasive non-native plants are properly managed. (Photo by Kevin Hurley/WGFD)

Figure 37. Photos of a big sagebrush (both Wyoming and mountain subspecies) site 4 years post-wildfire in Sierra Nevada Mountains, Nevada used as transition range by mule deer. Upper photo on a north-facing slope showing natural sagebrush and bitterbrush recovery with interspace perennial bunchgrass without cheatgrass; lower photo taken on same site but on a south-facing slope with natural recovery of sagebrush and Ceanothus with a dense understory of cheatgrass (Photos by Mike Cox/NDOW).

Basin Big Sagebrush

Basin big sagebrush tends to grow in deep, fertile soils, and is an indicator of productive sites. Many sites once dominated by basin big sagebrush are now farmlands, where it is restricted to field edges, swales, and along drainage ways (Collins 1984). Basin big sagebrush commonly grows in association with cheatgrass, bluebunch wheatgrass, Thurber's needlegrass (*Achnatherum thurberianum*), needle and thread (*Hesperostipa comata*), Idaho fescue (*Festuca idahoensis*), and Sandberg bluegrass (*Poa secunda*) (Hodgkinson 1989).

Sapsis (1990) suggested fire return intervals in basin big sagebrush are intermediate between mountain big sagebrush and Wyoming big sagebrush (15–70 yrs). Repeat fires within short intervals have removed this species from extensive areas (Bunting 1990).

Fires in basin big sagebrush communities, although variable in severity, are typically stand replacing with most plants killed, and resprouting does not occur (Sapsis and Kauffman 1991). Scattered, unburned basin big sagebrush may survive, particularly where the soil is thin and rocky and where low herbaceous biomass limits the fire's spread (Bushey 1987). Basin big sagebrush reinvades a site primarily by off-site seed or seed from plants that survive in unburned patches.

Rate of stand recovery depends on season of burn, which affects availability of seed, postfire precipitation patterns, and amount of interference offered by other regenerating plant species (Daubenmire 1975). Seedling establishment may begin immediately following a disturbance, but it usually takes a decade or more before basin big sagebrush dominates the site. In Wyoming, where basin big sagebrush has been removed by chemical means, it regained its pretreatment cover in 17 years in stands where grazing was not controlled (Johnson 1969).

Mycorrhizal associations may also affect stand recovery. The presence of *Glomus* spp. fungi may be required for the successful establishment of seedlings. Areas where basin big sagebrush cover has been eliminated due to frequent fire and subsequently dominated by nonmycorrhizal cheatgrass may no longer have the fungi in the soil. Basin big sagebrush reestablishment may be inhibited on these sites (Rosentreter and Jorgenson 1986).

Guidelines

1. Do not treat basin big sagebrush where the potential is high for cheatgrass or other invasive species to dominate

the understory and restrict sagebrush seedling establishment.
2. Treat no more than 15% of the existing community where late-seral stage sagebrush plants dominate and the understory production or diversity appears limited (Fig.38).
3. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008).
4. Aggressively suppress stand-replacing wildfires where >30% of the landscape of interest may burn to protect intact basin big sagebrush communities.

Silver Sagebrush (*Artemisia cana*)
Silver sagebrush typically grows in basins and along drainages where it represents a potential natural community. Upland sites usually have a sandy soil component whereas coarse, alluvial deposits comprise bottomland sites. Many of the lowland sites are also subjected to periodic flooding, erosion, and deposition. Site preference includes locations influenced by high water tables, especially where roots can intersect the water table for at least part of the growing season (Johnson 1979). Silver sagebrush has high forage value and palatability for wintering wildlife, including mule deer (Beetle and Johnson 1982).

Unlike other sagebrush species, silver sagebrush is fire-adapted and reestablishes primarily through root sprouting and rhizomes following burning (Beetle 1960). Prescribed burning can create a wide range of plant responses and densities (White and Currie 1983). Pre-burn densities are quickly restored following most spring burning.

Mortality is directly related to fire intensity, fire severity, and season of burning. White and Currie (1983) conducted spring and fall burns under comparable site conditions on a mixed-grass prairie in eastern Montana. Fall burning produced 75% mortality of totally consumed plants, whereas spring burning resulted in 33% mortality of totally burned plants. Fall fire severity was greater as a result of reduced soil moisture conditions.

Guidelines
1. Consider treatment in dense silver sagebrush stands where understory species have been depleted.
2. Use caution or do not treat stands where cheatgrass or other invasive species are present.
3. Use spring burns to increase plant coverage, rejuvenate sagebrush plants, and enhance understory vegetation.
4. Use fall burns for shifting the competitive advantage to herbaceous species.
5. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008).

Figure 38. A late-seral basin big sagebrush stand in western Nevada with typical plants exceeding 8 feet in height with poor sagebrush age class diversity and limited understory diversity (Photo by Mike Cox/NDOW).

Tall Threetip Sagebrush (*Artemisia tripartita tripartita*)
Tall threetip sagebrush is generally found on flat to relatively steep, moderate to deep, well-drained, loamy to sandy loam soils and is especially common along river drainages up to 9,000 feet (Beetle and Johnson 1982). This species is also tolerant of dry soil conditions and found from 6,000 feet to 7,000 feet in Wyoming (Beetle 1960). Stands of tall threetip sagebrush often occur adjacent to mountain big sagebrush stands, but usually on moister soils at higher elevations (Blaisdell et al. 1982).

Care must be exercised when treating mixed stands of tall threetip and vasey big sagebrush because tall threetip sagebrush is capable of vigorous vegetative regeneration and site domination. Thus, mixed stands can be converted entirely to tall threetip sagebrush with reduced species diversity (Passey and Hugie 1962). This is of more concern if fire intervals are shortened. However, quick recovery results in short-term establishment of ground cover, as well as structure and species diversity in mixed stands.

There are few landscape management objectives for this subspecies, however, a hot fall fire can be used when tall threetip dominates the site to thin threetip and increase the herbaceous component. Grazing management systems should be considered, as it becomes more dominant on overgrazed ranges (Hironaka et al. 1983).

Guidelines
1. Use caution when treating mixed stands of three-tip and big sagebrush to avoid site domination by three-tip following disturbance.
2. Use caution or do not treat stands where cheatgrass or other invasive species are present.
3. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008).

Black Sagebrush (*Artemisia nova*)
and Little Sagebrush (*Artemisia arbuscula*)
Black sagebrush and little sagebrush are usually associated with areas with little soil profile development on lower slopes of high-desert foothills. Typical sites consist of dry, shallow, gravelly, well-drained soils of alluvial fans, sills, mountain slopes, and wind-blown ridges. Black sagebrush communities located on impermeable layers (clay or bedrock) at approximately 1-foot depth and within higher precipitation zones (12-14 in.) are quite capable of producing adequate fuels for fire spread. Where fire does occur, plants are easily killed by fire and recovery is very slow (West and Hassan 1985).

Guidelines
1. Do not use fire to treat black or little sagebrush and aggressively suppress stand-replacing wildfires where black or little sagebrush is an important forage plant.
2. Use extreme caution or do not treat stands where invasive species are present.
3. Maintain a herbaceous species composition consistent with the ecological capability of the site (USDA-NRCS 2008).

Oil and Gas Development

Background

Energy consumption and production continue 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 >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 (e.g., Arctic National Wildlife Refuge, Raton Basin, San Juan Basin, Uinta-Piceance Basin, Green River 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.

Because much of the IMW Ecoregion 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 (Fig. 39).

Coincidentally, much of the IMW contains significant accumulations of natural gas and coal deposits. Coal Bed Methane (CBM) and natural gas are becoming a predominant energy alternative within the IMW. Natural gas and CBM reserves can be found throughout much of the Rocky Mountains and the IMW (Fig. 40). Unfortunately, development and extraction activities associated with CBM and natural gas tend to be aggressive and therefore have the 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 amount and intensity of the disturbance, specific locations and arrangements of disturbance, and ecological importance of 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, have 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, coal bed methane, and oil shale. Many industries depend upon other materials (e.g., copper, uranium, vanadium, etc.) for their products or services and extracting these raw materials can have the very same effect on wildlife and the environment as oil and gas development.

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, disrupt access to habitat, or cause avoidance and stress. For this discussion, impact thresholds are based upon 2 quantitative measures: density of well locations (pads) and cumulative disturbance per section (a legal section of 640 acres or an area equivalent to 640 acres). Density of well locations has

Figure 39. Big Piney-LaBarge area adjacent to the Wyoming Range, illustrating effects of development on crucial mule deer winter range. (Photo by Dan Stroud/WGFD).

Figure 40. Coal Bed Methane locations within the Rocky Mountain region (Map courtesy of U.S. Geological Survey).

bearing on intensity of disturbances associated with oil and gas field operations, while 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. 41 and 42). 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/section or when road density exceeds 3 miles/section (USDI 1999). The following describe and define relative degrees of impact (Table 5).

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; 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 the impacts will be much more difficult to mitigate. It may not be possible to fully mitigate impacts caused by higher well densities, 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
The function and effectiveness of habitat would be severely compromised (Fig. 43). 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 2005a). 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,

Table 5. 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 and 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/section	5-16 wells and 20-80 acres disturbance/section	> 16 wells or > 80 acres disturbance/section

Figure 41. A typical gas well; 3-4 acre footprint. (Photo courtesy of BLM, Pinedale Field Office).

Figure 42. This storage area is an example of other facilities that directly remove habitat. (Photo courtesy of BLM, Pinedale Field Office).

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 (Figs. 41 and 42) 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 (Fig. 44), 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 miles of a road, and 160 square mile (97% of landscape) is within 0.25 miles 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 take ≥20 years, assuming that reclamation was done properly. 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, as success is highly dependent upon amount and timing of moisture; reseeding is usually required if reclamation is conducted > 1 year post-disturbance (Fig. 45).

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 (traffic, noise, pets, etc.) (Fig. 46).

During winter months, this stress could be particularly important because animals are typically operating at a negative energy balance. In addition, the diversion of energy reserves can be detrimental for other critical periods during the life cycle, such as gestation and lactation. Based on a simulated mine disturbance experiment, Kuck et al. (1985) suggested increased energy costs of movement, escape, and stress caused by frequent and unpredictable

disturbance may have been detrimental to elk calf growth. An Environmental Impact Statement on oil and gas development in the Glenwood Springs (NM) Resource Area 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, etc. are primary factors leading to avoidance of developed areas 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 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. This contributes to noise and increased human presence within the development area. Rost and Bailey (1979) found an inverse relationship between habitat use by deer and elk and distance to roads. This 'displacement' can result in under use of habitat near disturbances, whereas over use may occur in nearby 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 from a variety of increasing recreational activities.

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. The 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."

Figure 43. The Jonah gas field in western WY--extreme impact. (Photo courtesy of BLM, Pinedale Field Office).

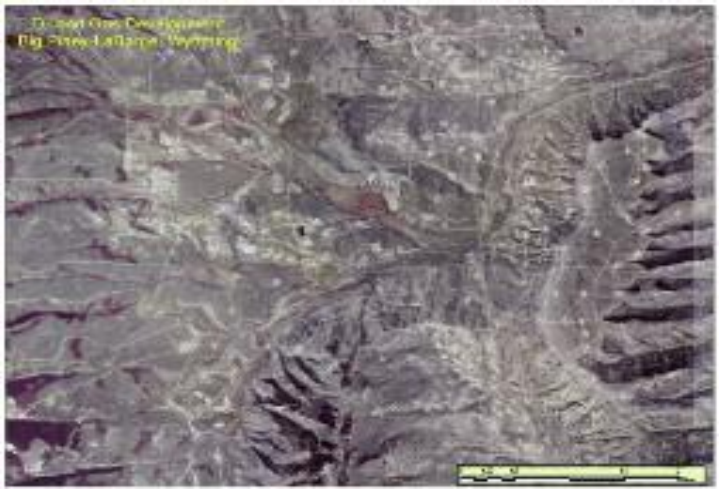

Figure 44. Color infrared photo of Big Piney-LaBarge area development, most of which occurs on mule deer crucial winter range. (Photo courtesy of BLM, Pinedale Field Office).

Figure 45. Early stages of reclamation in the Pinedale, WY area illustrating establishment of grasses and some shrubs. (Photo courtesy of Jonah Interagency Office, Pinedale, WY).

Figure 46. Mule deer on winter range with oil and gas development. (Photo by Mark Gocke/WGFD).

Studies by Sawyer et al. (2006) on the Pinedale Anticline (Mesa) near Pinedale led them to state: "Model coefficients and predictive maps suggested mule deer were less likely to occupy areas in close proximity to well pads than those farther away (Figs. 47-50). Changes in habitat selection appeared to be immediate (i.e., year 1 of development), and no evidence of well-pad acclimation occurred through the course of the study; rather, mule deer selected areas farther from well pads as development progressed."

Should development occur within or proximate to migration corridors (Fig. 51), 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, etc.), 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 (Fig. 44, 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 'edge effect' between developed and undeveloped areas often results in reduced forage quality and security cover, potentially increasing deer susceptibility to predation.

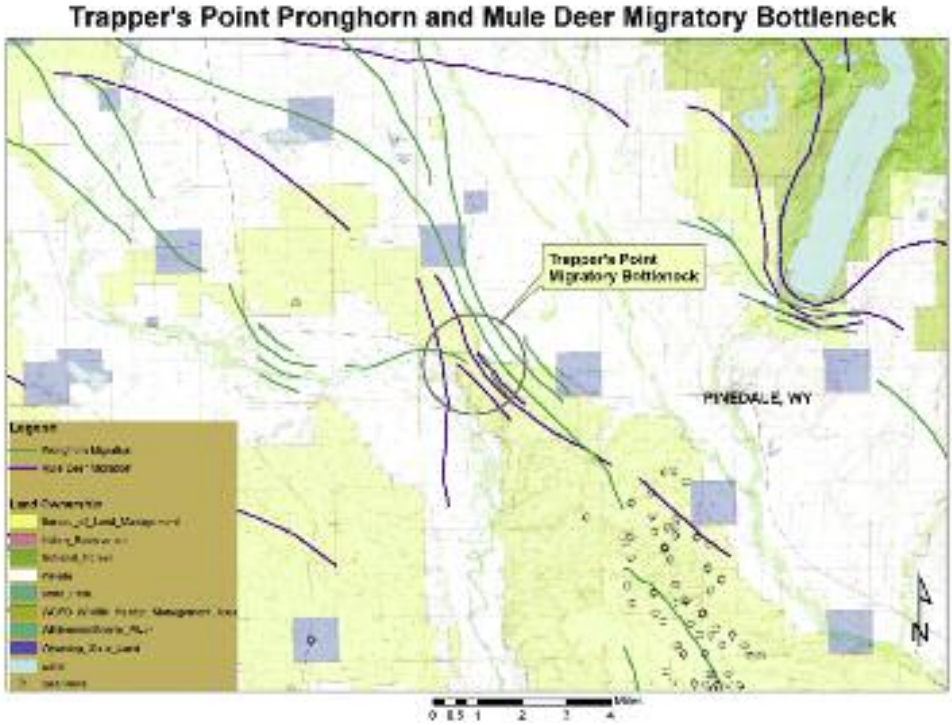

Figure 51. Map of pronghorn and mule deer migratory bottleneck in the Pinedale, WY area. (Courtesy of Dan Stroud/WGFD).

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 survival and productivity of the 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 support or service industries linked to development can aggravate adverse impacts. These impacts can, and are, similar to those that occur during construction and operations, only intensified. Vehicular traffic to support operations would likely increase

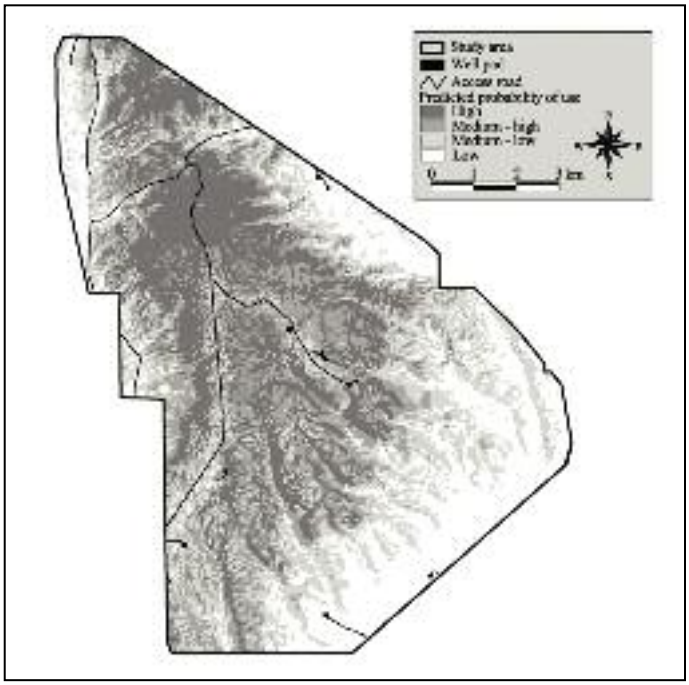

Figure 47. Predicted probabilities and associated categories of mule deer habitat use during 1998-1999 and 1999-2000 winters, before natural gas field development in western WY, USA. (Sawyer et al. 2006).

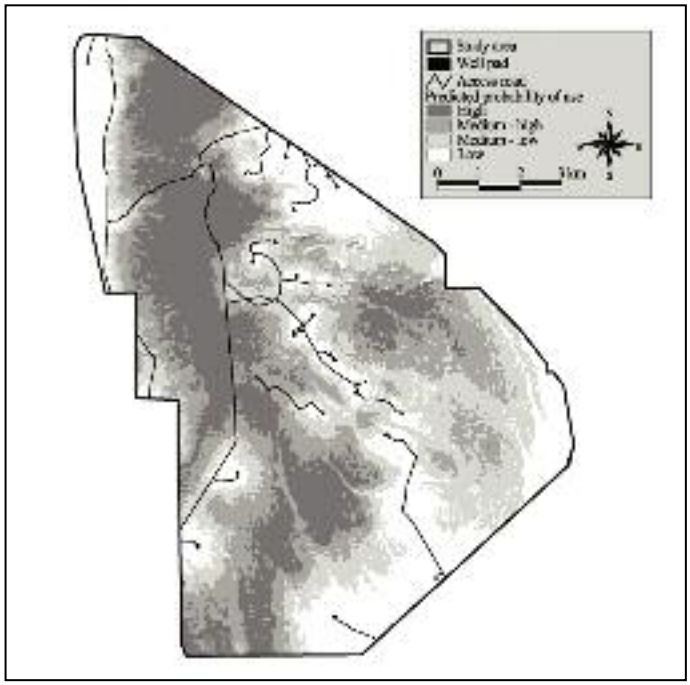

Figure 48. Predicted probabilities and associated categories of mule deer habitat use during year 1 (winter of 2000-2001) of natural gas field development in western WY, USA. (Sawyer et al. 2006).

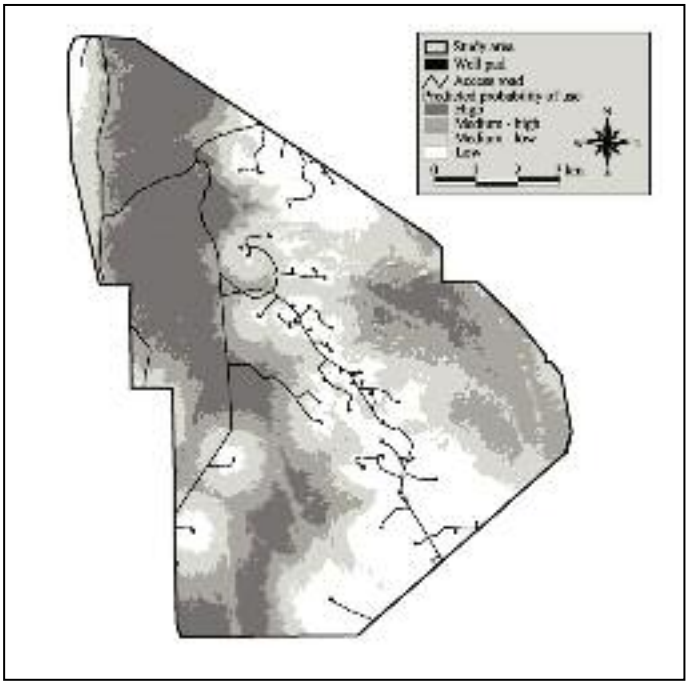

Figure 49. Predicted probabilities and associated categories of mule deer habitat use during year 2 (winter of 2001-2002) of natural gas field development in western WY, USA. (Sawyer et al. 2006).

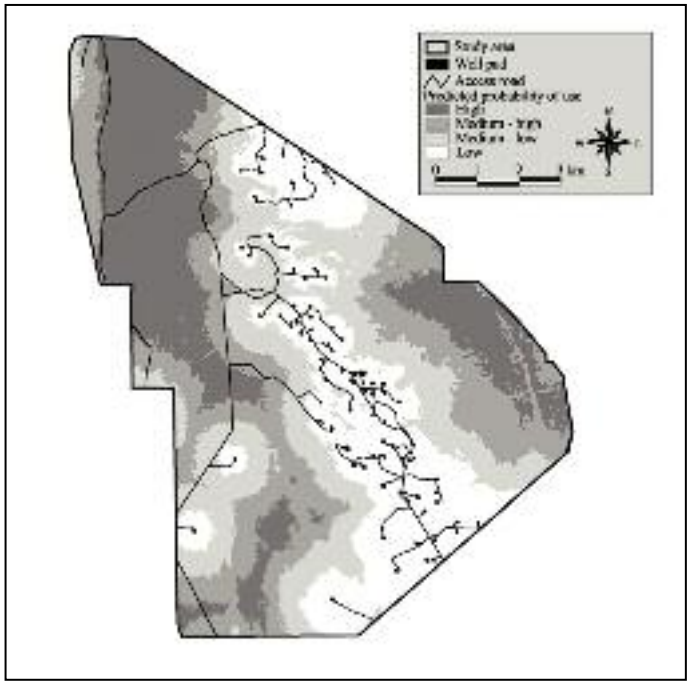

Figure 50. Predicted probabilities and associated categories of mule deer habitat use during year 3 (winter of 2002-2003) of natural gas field development in western WY, USA. (Sawyer et al. 2006).

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, or chemical defense compounds with toxic or allergenic properties (NMDGF 2007).

Activities occurring at the well site (drilling, pumping, etc.) or associated with product transportation to other destinations via pipeline or vehicle may lead to the release of a variety of toxic compounds. These compounds are common by-products and pose serious health risks to not only employees, but also the environment and the wildlife inhabiting the locality. All these events can decrease the amount of area available to mule deer and other wildlife. Finally, 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.

Figure 52. Well pads located on prominent sites contribute to increased visual and auditory impacts. (Photo courtesy of BLM, Pinedale Field Office).

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 (when deer are on winter range, fawning periods, etc.).
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 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.
8. Locate drill pads, roads, and

Figure 53. Close roads to reduce unnecessary access. (Photo courtesy of BLM, Pinedale Field Office).

facilities below ridgelines or behind topographic features (Fig. 52), 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 (reproduction, winter, parturition, and rearing).

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. 53).
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 roads.

C. Wells

1. Drill multiple wells from the same pad using directional (horizontal) drilling technologies (Fig. 54).
2. Disturb the minimum area (footprint) necessary to efficiently drill and operate a well.
3. Utilize "mat" drilling to eliminate top-soil removal (Fig. 55).

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.

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.

Figure 54. Well site with 3 wells. (Photo courtesy of BLM, Pinedale Field Office).

Figure 55. Mat drilling pad in the Jonah Field south of Pinedale, WY. (Photo courtesy of BLM, Pinedale Field Office).

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 view of 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

Figure 56. Reclamation of roads is necessary to eliminate permanent access and disturbance. (Photo courtesy of NMDGF).

Figure 57. Failed reclamation of a gas well. (Photo courtesy of BLM, Pinedale Field Office).

Figure 58. Successful final reclamation of a gas well. (Photo courtesy of BLM, Pinedale Field Office).

and kept in serviceable condition.

2. Produced water should not be pumped onto the surface except when beneficial for wildlife, provided 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, wells, etc.). 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 USFWS 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

Where questions or uncertainties exist about the degree of impact to specific resources, or effectiveness of mitigation, industries and companies should fund special studies to collect data for evaluation and documentation.

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 and mud clinging to equipment.
3. Request employees to clean mud from footwear 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 (dry, sandy, steep slopes).
4. A variety of native grasses, shrubs, 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.

M. Final Reclamation

1. Salvage topsoil during decommissioning operations and reapply to reclaimed surfaces.
2. Replant a mixture of forbs, grasses, and shrubs that are native to the area and suitable for the specific ecological site.
3. Restore vegetation cover, composition, and diversity to achieve numeric standards commensurate with the ecological site (Figs. 56 - 58).
4. Do not allow grazing on re-vegetated sites until the plants are established and can withstand herbivory.
5. Continue to monitor and treat reclaimed areas until plant cover, composition, and diversity standards have been met.
6. 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.

Open Pit and Hard Rock Mining

Background

Open pit mining, particularly throughout the Great Basin in the past 20 years, has become a significant part of the economy and environment. Advances in technology, which have allowed the mining industry to create large open pit mines, have included cyanide chemistry advances, ore deposit identification, and exploration technology. Many of these operations have or continue to utilize cyanide leaching techniques. Cyanide can create many harmful situations for mule deer, either directly (consumption of cyanide laden materials or solutions) or indirectly (ingestion of low pH solutions as a result of

Figure 59. An open-pit gold mine in northern Nevada, located on crucial mule deer transition and winter range (Photo by Ken Gray/NDOW).

Figure 60. Critical migration route for mule deer in northeastern Nevada is threatened with blockage by extensive mining activity and tailings impoundments. (pushpins and dots are locations of a mule deer doe with a GPS collar, created by Cody Schroeder/NDOW).

Figure 61. Cyanide-related mule deer mortality adjacent to an open pit mine leach pad. (Photo courtesy of NDOW).

chemical reactions.) Open pits, waste rock dumps, heap leach pads, tailings impoundments, and exploration activities can also contribute to loss of habitat for mule deer (Fig 59).

Figure 62. Process pond with deer exclusionary fencing and bird balls, Humboldt County, NV. (Photo courtesy of NDOW).

Figure 63. Operational tailings impoundment in Humboldt County, Nevada that is over 300 acres (upper photo) and reclaimed tailings impoundment in Humboldt County, NV that is >200 acres with restored 4-foot high basin big sagebrush (lower photo). (Photos courtesy of NDOW).

In most cases with mining-disturbed land, restoration is not practical, therefore reclamation of those disturbed lands is the preferred closure method. Reclamation of mining-disturbed land is crucial to long-term productivity. Proper topography, seed mix, soil cover type and depth, and precipitation are important factors to successful reclamation. If reclamation is successful, highly productive habitat can be established in the long term.

Issues and Concerns

Either directly or indirectly, open pit mining can affect mule deer and their habitat. Chemicals or solutions at mine sites can cause problems for mule deer through ingestion of acute or chronic sub-lethal levels of hazardous constituents. In the most extreme cases, direct and unmitigated habitat loss can occur. Migratory corridors can be restricted or eliminated by mine components such as the pit, waste rock dumps, heap leach pads, tailings impoundments, and haul roads (Fig. 60).

Chemical constituents not only cause acute toxicity, but also chronic problems and, in the case of radiological exposure, degenerative effects. Cyanide is most often the immediate constituent of potential concern (Fig. 61). Cyanide is most often applied to gold- or silver-bearing ore piles, “leaching out” the precious metals. Areas of concern for mule deer during this process are in the application phase when the cyanide solution levels are at their highest, collection ditches around the base of leach pads, artificial ponds where the solution is stored, and large tailings impoundments where solution is evaporated. During the application phase, small ponds can occur on the top and sides of leach pads where cyanide becomes accessible to mule deer. Collection ditches and ponds can also act as traps because they are usually lined with slippery plastic. Some heavy metals can accumulate in solutions, plants, and soils, and if accessible to wildlife, these constituents are toxic. Mercury, which is often associated with gold mineralization, is problematic due to bioaccumulation through trophic levels.

Permitted disturbance can reach 15 square miles for a large, open-pit mine. Habitat type conversion, temporary, or permanent loss can occur. Often, disturbed lands, where vegetation type conversions have occurred, are invaded and dominated by non-native, invasive plants such as cheatgrass. The same types of problems occur when disturbed land is not reclaimed and noxious or invasive species are allowed to dominate plant communities. Open pits not reclaimed will provide little or no value to mule deer and other wildlife because lack of topsoil seriously disrupts or completely inhibits natural plant succession.

Waste rock dumps, heap leach pads and associated haul roads present migratory pattern disruption. Mule deer can

be forced to use less desirable alternative migratory routes. If those alternative migratory routes do not exist, deer are forced to travel through the mine site exposing them to additional hazards such as haul trucks, cyanide, and fencing.

Guidelines

Active participation and input from the exploration scoping phase through the post-closure monitoring phase is essential to ensure the proper steps are taken to minimize impacts to mule deer. Wildlife agencies need to establish "cooperating agency" status with federal partners during National Environmental Policy Act assessment phase of projects. This status provides the opportunity for early disclosure of issues and concerns and provides wildlife expertise early in the Environmental Impact Statement or Environmental Assessment process. If the proper steps are taken and reclamation of the site is successful, post-mining habitat loss can be minimized.

A. Exploration

1. Limit exploration activities to appropriate seasons. If the project area is in crucial deer winter range, activities should be limited to summer months, and vice versa. If the project area is within a migratory corridor, activities should be limited to times when the corridor is not utilized by mule deer.
2. Minimize ground disturbance of all activities, including road construction, pad construction, and sumps. Apply stipulations in the plan of operations for all exploration activities.
3. Re-grade and reseed to reproduce pre-disturbance conditions and species composition. This may require activities for more than 1 growing season in areas affected by invasive plants such as cheatgrass. Native or non-native grasses and forbs can be used during the first year to compete with invasive species. Second or third year reseeding activities should focus on reestablishment of native shrubs, including sagebrush and bitterbrush.

B. Operation

1. Use exclusionary measures to keep mule deer and other wildlife out of operational areas with cyanide. These areas include heap leach pads with active cyanide application and associated collection systems (ditches and perforated pipe) and process ponds (barren and pregnant). Exclusionary measures include ≥8-foot high fencing. Gates that are used on a daily basis should be closed unless there is immediate traffic. "Bird balls," which are intended to exclude avian fauna, also can help minimize deer use by limiting the visibility of open water (Fig. 62).
2. Protect migration corridors. Avoid operations and construction in known migration corridors. If it is not feasible to manipulate placement of operations to allow deer to migrate, utilize a phased approach to allow for passable areas as a concurrent reclamation (reclamation of disturbed areas during operations) phase system is put into place.
3. Use concurrent reclamation at all possible locations on the mine site to ensure that, at any one time, the least amount of unreclaimed, non-productive habitat is exposed.

C. Closure and Final Reclamation

1. Maintain all preclusive fencing until hazards are removed in ponds, on heap leach pads, etc. Also, roadside fencing should be maintained until large truck and haul traffic ends.
2. Re-grade as soon as operations are completed. Topography of final slopes should be dictated by pre-disturbance conditions, not the standard 3:1 or 2.5:1. Simple, uniform slopes often erode and do not provide any topographic cover for deer.
3. Minimize uptake of heavy metals by vegetation and then mule deer. Reclaim using cover soil depths adequate to allow vegetation root systems to establish without reaching mined materials.
4. Reseed to establish pre-disturbance vegetation communities (if those original plant communities were natural with no non-native, invasive species)
5. Utilize plantings to close heap leach pads, waste rock dumps, and tailings impoundments to reduce infiltration from precipitation. Even though a heap leach pad has been closed, the chemical make-up of the soils has been changed and simply allowing water to flow into them can be harmful. Extremely low pH solutions (< 0.5) can drain out and carry heavy metals to areas where the solution is accessed by deer. Successful reclamation provides good cover and forage for deer (Fig. 63).

Human Encroachment

Background

Human activity can impact habitat suitability in 3 ways: displacing wildlife through habitat loss (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 agriculture conversion), or displacing wildlife by altering wildlife perception of habitat suitability through other than physical alteration (e.g., noise, activity).

Current levels of human influence and ever-increasing human populations clearly limit the potential for ever restoring mule deer populations to levels observed in the mid-20th century. Nevertheless, there exist opportunities for conservation and management actions that can reduce

Figure 64. Housing development and Interstate 80 highway in crucial mule deer winter range at the base of a south-facing mountain near Reno, NV typifies urban encroachment into mule deer habitat and blockage of migration corridors throughout much of the IMW. (Photo by Mike Cox/NDOW).

Figure 65. High-fenced facilities for privately owned ungulates eliminate access to habitat previously available to wild ungulates. (Photo courtesy of IFGD).

Figure 66. Deer attracted by feeding by some individuals leads to nuisance behavior on surrounding property in Salmon, ID. (Photo courtesy of IDFG).

impacts of human encroachment or restore habitat values and thereby maintain or increase mule deer numbers and associated public and ecological benefits.

Issues and Concerns

Habitat Loss

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. In other cases, development in wildlife habitat is simply a response to exploding human populations in western states and socioeconomic trends. Occupation of these habitats brings with it construction of homes, fencing, roadways, and other supporting infrastructure, such as stores, health facilities, and other buildings (Fig. 64).

These homes and communities are often located in important mule deer migration corridors or crucial winter range. Impacts of development often reach well beyond actual acreage covered by buildings, roads, and other infrastructure. In many cases, fences around these structures further exclude deer from usable resources. The resultant development destroys many of the features that initially drew people to these habitats. During the mid-1990s alone, human development occupied 5.4 million acres of open space in the West (Lutz et al. 2003).

Amount of habitat lost through road and railroad construction varies based on size and type of construction. Reed (1981a) estimated interstate, rural, and county highways usurp 45, 12, and 7 acres of land per mile of road. Ubiquitous travel networks through mule deer habitat on public forest and rangelands result in further loss of thousands of acres of habitat. Similarly, development of water impoundments and distribution systems eliminate habitat once available to deer. More recently, several western states have witnessed construction of "high-fenced" facilities designed to contain privately owned ungulates. These facilities can effectively eliminate thousands of acres of mule deer habitat (Fig. 65). Since 2001, >7,000 acres of occupied mule deer habitat were usurped by high-fenced facilities in east- and south-central Idaho alone.

Where development is unavoidable, "mitigation," through acquisition or management of land elsewhere is sometimes employed to offset habitat loss. It is important to recognize that mitigation projects, while better than doing nothing, often do not replace the lost habitat's effectiveness and suitability. Perhaps Reed (1981a:522-523) says it best, *"Hence the concept of compensation or mitigation becomes an absurdity as wildlife habitat continues to be whittled away."*

Reduced Habitat Suitability

Human activity has the ability to alter habitat suitability,

Figure 67. Indirect effects of wildlife unfriendly fences such as impeding migration, separation of fawn and doe or limiting access to important habitats will increase energy demands and place animals at higher risk to injury or predation, ultimately leading to lower survival. (Photo by Stan Harter/WGFD).

thereby influencing habitat quality. Although some human activity and man-made structures may seem innocuous, most reduce capability of the land to support deer, often through cumulative effects.

Altered fire regimes due to societal influences/misconceptions and human inhabitation frequently have led to long-term conversion from productive habitats to conditions much less suitable for mule deer (e.g., diverse shrub-steppe conversion to near monocultures of cheatgrass, or vigorous early seral mountain brush habitats converted to dense forests or decadent shrub communities providing little deer forage).

Conversion of natural habitats to agricultural lands can have mixed impacts on mule deer populations depending on extent of conversion, crops produced, and landowner tolerance. Extensive conversion of large areas to crops that provide little forage or cover will likely reduce deer numbers significantly or displace deer completely (e.g., expansive potato farming in the Snake River Plain). Conversely, crops that produce usable forage interspersed with adequate cover and native habitat can support high density deer populations provided landowners are amenable. However, differences in landowner tolerance within a local area or changes in ownership can lead to substantial conflicts and a need for intensive management actions. These situations likely result in increased cost:benefit ratios relative to management of intact systems.

In addition to directly usurping habitat, development of human communities often alter adjacent habitats as well. Shrub habitats providing food and cover may change to pasture or manicured lawns. Ornamental plants may replace native shrubs and forbs. People frequently bring domestic dogs and livestock that may compete with wildlife or jeopardize wildlife through disease transmission. Improper use of off highway vehicles (OHVs) can alter habitat characteristics through destruction of vegetation, compacting soil, and increasing erosion.

However, human occupation may provide some advantages to local wildlife populations (Tucker et al. 2004). Wildlife in some developed areas may acquire more water from artificial sites (e.g., pools, ponds) and enhanced forage (e.g., lawns, plantings, golf courses, agricultural fields) than in surrounding areas. There are typically fewer natural predators in urban areas that reduce wildlife mortality.

Though these advantages may occur, there are just as many times where these same habitat alterations have negative consequences. Enhanced forage conditions and decreased mortality ultimately lead to unhealthy animal densities, that may increase disease outbreaks and attract predators that in turn prey on domestic pets, as well as humans. Inevitably, some individuals will feed deer that can lead to aggressive behavior toward humans. Other people in the same area will suffer unacceptable damage to ornamental plants, gardens, and other property, at times leading to widespread unrest in a community. An insidious side-effect of such situations is creation of opinions that deer are nuisance wildlife (Fig. 66), similar to Canada geese (*Branta canadensis*) in many developed areas across the U.S. This devaluation of deer in the public eye will only increase difficulty in developing public support for mule deer and management of natural habitats (Lutz et al. 2003).

Although we often observe mule deer negotiating fences with apparent ease, fencing can create significant barriers or impediments to normal deer movement and increase energy demands. Fence permeability obviously varies with fence design, but all fences affect deer to some extent. Fences along major highways are often designed to completely exclude ungulates and therefore block

movements and eliminate migration corridors, effectively isolating some populations. Adult deer may be able to jump over net-wire or 5-6 strand, barbed-wire fences, but fawns are generally unable to negotiate such structures until several months old (Fig. 67).

Negative impacts of low permeability fence types are readily discernible, but even more permeable fences create problems for deer. In some cases, deer may spend several minutes walking back and forth along a fence to find a potential crossing point. Fences on slopes exacerbate problems because functional fence height increases significantly for deer on the downhill side (Wasley 2004) and deep snow can make an otherwise permeable fence impassable. Crossing fences also carries risks of injury that might later compromise an animal's ability to avoid predators or function normally. Because of ecoregional climate patterns and topography in the IMW, many deer populations display lengthy migrations (Heffelfinger et al. 2003) along which individual animals may encounter dozens of fences. The cumulative impact of repeated fence crossings can only increase energy costs and risk of injury, and potentially increase predation risk, particularly for fawns.

Road and railway development and upgrades can eliminate linkages and fragment important habitats (Noss and Cooperrider 1994, Forman and Alexander 1998, Forman 2000, Forman et al. 2003). Highway-associated impacts have been characterized as one of 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). Roadways, railways, and associated fences fragment habitat and impede movements for migratory herds (Lutz et al. 2003). These impacts are especially severe in western states where rapid human population growth and development have resulted in increased traffic volume and subsequent construction of new highways. Further, mule deer have demonstrated limited ability to alter migration to avoid impediments (Wasley 2004). Construction of a 4-lane, divided highway in southeastern Idaho was implicated in isolation and reduction of a previously migratory deer herd (Hanna 1982). Long-term fragmentation and isolation render populations more vulnerable to influences of stochastic events, and may lead to local mule deer extirpations.

Other human activity impacts directly tied to increased roadways include increased poaching of mule deer, unregulated off-highway travel, and ignition of wildfires. Roads also serve as corridors for dispersal of invasive plants that degrade habitats (White and Ernst 2003).

Past efforts to address highway impacts 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 in the U.S. have made tremendous commitments to early multi-disciplinary connectivity planning, including Washington (Quan and Teachout 2003), Colorado (Wostl 2003), and southern California (Ng et al. 2004); 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 use more credible information and conduct scientific analyses to support recommendations like design and location of passage structures. Information and analyses include deer-vehicle collision databases, simple GIS mapping of linkage needs (Ruediger and Lloyd 2003), or more complex modeling of wildlife permeability (Singleton et al. 2002).

Figure 68. Highway underpass wildlife crossing on 4-lane stretch of U.S. Highway 395 in northeastern California, fenced to direct migrating deer safely under the roadway. (Photo by Mike Cox/NDOW).

Structures designed to promote wildlife permeability across highways are increasingly being implemented throughout North America, especially large, bridged structures (e.g., underpasses or overpasses) designed specifically for ungulate and large predator passage (Fig. 68) (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). It is important to conduct scientifically sound monitoring and evaluation of wildlife response to improve future passage structure effectiveness (Clevenger and Waltho 2003, Hardy et al. 2003).

Increased Vulnerability

Many factors influence vulnerability of ungulates during hunting seasons, including timing and length of the season, hunter numbers, and security cover (Moroz 1991). Increased roads and motorized trails, combined with increasing use of off-road vehicles (e.g. ATVs) have reduced security cover available to mule deer. Impacts of reduced security cover include lowered survival, reduced age structure, lower male to female ratios, and more restrictive hunting seasons (Leptich and Zager 1991, Canfield et al. 1999). Excessively low male:female ratios ultimately can reduce productivity by affecting pregnancy rates and juvenile survival (Noyes et al. 1996). Furthermore, many states and provinces have mule deer management objectives for post-season buck-doe ratios, buck age structure, and/or recreational hunting opportunity. Motorized access and vulnerability can affect the ability of mule deer to meet state management objectives.

Figure 69. Mule deer often appear acclimated to human development, but this situation misrepresents typical deer behavior and habitat needs at a population scale. (Photo by Gerianne Hummel, Carson City resident).

Managing motorized access to provide adequate security cover is a critical component to mule deer habitat management. Unfortunately, there is no simple "cook-book" method for determining proper motorized access management prescriptions (Hillis et al. 1991). Hunter densities, topography, and vegetative cover all influence the relative impact of motorized access on vulnerability (Edge and Marcum 1991, Unsworth et al. 1993). Steeply dissected terrain with substantive vegetative hiding cover can tolerate relatively higher motorized road and trail densities without substantially altering vulnerability. Conversely, vulnerability substantially increases in relatively flat terrain with limited vegetative hiding cover even with relatively low motorized access densities.

Displacement through Disturbance

Extensive research has documented that wildlife modify their behavior to avoid activities they perceive as threatening, (e.g., elk avoidance of roads with larger traffic volumes). However, this avoidance is generally temporary, and once the disturbance is removed, wildlife return to their prior routine. Although avoidance behavior is very common, research has rarely documented population level responses (e.g., decreased fitness, recruitment, conception) as a direct result of disturbance. Direct and frequent disturbance of Coues white-tailed deer (*Odocoileus virginianus couesi*) during breeding season did not result in any population level responses (Bristow 1998). However, Shively et al. (2005) attributed declines in elk calf:cow ratios to experimental disturbance during the peak calving period and Noyes et al. (2001) observed changes in conception dates and pregnancy rates possibly associated with archery hunting during breeding season.

Information regarding responses of deer to roads and vehicular traffic is scarce and imprecise (Mackie et al. 2003). Perry and Overly (1977) found main roads had the greatest impact on mule deer, and primitive roads the least impact. Further, they indicated roads through meadow habitats reduced deer use, whereas roads through forested habitat had less effect. Johnson et al. (2000) surmised that proximity to roads and trails has a greater correlation with deer distribution than does mean road density (derived from crude calculations based on area). Off-road recreation is increasing rapidly on public lands. The USFS estimated off highway vehicle (OHV) use increased 7-fold 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 (Wisdom et al. 2005), especially for elk.

Some white-tailed deer in the eastern U.S. have apparently acclimated to relatively high densities of people and disturbance. Similarly, mule deer are commonly observed in close association with human developments in many areas however, these deer may represent relatively small proportions of overall populations existing in a more natural state (Fig. 69). In northeastern Utah fawn:doe ratios and densities of mule deer in an urban setting were 30-40% lower than for rural counterparts (McClure et al. 1999). Domestic dogs are a common component of human developments and can cause additional disturbance to deer,

particularly when allowed to freely roam. Dog harassment of deer is most likely to occur, and be most detrimental, during winter when deer are concentrated on winter range. Repeated harassment when deer are in negative energy balance and hindered by snow further depletes energy reserves necessary for survival.

In and of themselves, disturbance factors have generally not been implicated in reduced mule deer population performance. However, given the nutritional and energy requirements of deer, it seems reasonable to assume such factors could work in subtle and undetected ways with a number of other factors to negatively impact deer.

Figure 70. Two mule deer bucks were both hit and killed by the same vehicle traveling on U.S. Highway 93 in eastern Nevada. (Photo courtesy of Nevada Department of Transportation).

Figure 71. Wildlife-unfriendly fences cause direct mortality of mule deer. (Photo by Tony Wasley/NDOW).

Direct Mortality

Direct loss of deer and other wildlife due to collisions with motor vehicles is a substantial source of mortality affecting populations (Fig. 70). Romin and Bissonette (1996) conservatively estimated that >500,000 deer of all species were 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 deer populations, wildlife-vehicle collisions annually cause many human injuries and deaths. Conover et al. (1995) estimated collisions resulted in 29,000 human injuries and 200 deaths annually. Further, deer-vehicle collisions result in substantial loss of recreational opportunity and revenue associated with deer hunting, and damage to property is tremendous (Reed et al. 1982, Romin and Bissionette 1996). Deer-vehicle collisions are a particularly severe problem on winter ranges to which deer populations historically have migrated in concentrated densities (e.g., Gordon and Anderson 2003). The problem of collisions is further compounded by the dramatic explosion of human residential and other development within mule deer winter range in the IMW.

Lesser amounts of direct mortality can be attributed to entanglement with fencing, but fences certainly cause thousands of deer mortalities each year (Fig. 71). Fencing may further increase deer-vehicle collisions in situations where deer become confined to roadways by adjoining fences (Wasley 2004). An often overlooked aspect of fence-related mortality derives from reduced ability to escape predators, particularly for fawns, when escape routes are blocked or escape is hindered by fences (Hölzenbein and Marchinton 1992).

Canals and reservoirs also cause direct mortality of mule deer. Canals with steep sides or those lined with concrete or other hard surfaces can trap deer that fall into them, eventually leading to drowning. Drowning also occurs when deer break through ice while attempting to cross reservoirs.

Although usually not considered a significant source of overall mortality, free-ranging and feral dogs certainly kill deer. Under some circumstances, such as periods of heavy snow on winter ranges, predation by dogs can be a serious problem (Boyles 1976).

Guidelines

A. Planning and Coordination

1. Develop and maintain interagency coordination in land planning activities to protect important habitats and reduce negative impacts to mule deer (Fig. 72).
2. Land and wildlife management agencies should play a proactive role in state, county, and city planning, zoning, and development where decisions affect the

integrity of adjacent private or public lands containing critical mule deer habitat.

3. Identify important habitats, seasonal use areas, migration routes, and important populations of mule deer. Discourage development, including recreation sites, in these areas.
4. Coordinate with agricultural producers to consider wildlife needs in 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 or railroads are developed or altered for higher speed and greater volume traffic.
6. Coordinate with agencies responsible for regulating high-fenced, private wildlife facilities. Locate facilities outside of mule deer habitat, particularly important winter ranges or migration corridors.
7. Encourage state and federal transportation agencies to fund positions to coordinate road planning and mitigation issues.

B. Minimizing Negative Effects of Human Encroachment

1. Develop consistent regulations and identify designated areas for OHV use.
2. Develop and maintain interagency coordination in enforcement of OHV regulations.
3. Encourage use of native vegetation in landscaping human developments to minimize loss of usable habitat.
4. Examine records of deer-vehicle collisions to identify major impact areas and evaluate need for wildlife passage structures. Consider railroads, canals, and other impediments to natural movement when evaluating need for passage structures.
5. Along highway segments where high levels of deer-vehicle collisions have been documented, encourage appropriate regulatory agencies to
   - Seed unpalatable plants in highway rights-of-way to decrease attractiveness.
   - Reduce highway speed limits.
   - Erect temporary warning signs during migration events (Sullivan et al. 2004).
6. Encourage practices that reduce vehicle trips at times or seasons of elevated deer-vehicle collisions (e.g., flex-time, carpooling, public transportation).
7. Construct overpasses and underpasses along wildlife corridors known to be mule deer travel routes. In the case of canals, escape ramps may reduce drowning mortality.
8. Provide ungulate-proof fencing to direct wildlife to right-of-way passage structures or away from areas of numerous deer-vehicle collisions.
9. Monitor activities that may unduly stress deer at important times of the year. Reduce or regulate disturbance if deemed detrimental. When applicable, encourage enforcement of regulations regarding dogs running at large or chasing wildlife.

Figure 72. Area of critical mule deer winter range on public land was part of a land swap and will be sold for a housing development due to lack of interagency coordination and funding. (Photo by Mike Cox/NDOW).

10. Direct new development toward previously disturbed areas (clumped rather than dispersed distribution).
11. Enhance alternate habitats to mitigate for habitat loss, including components like water availability.
12. Encourage use of wildlife-friendly (permeable) fencing in appropriate areas to minimize habitat fragmentation and direct mortality. Evaluate existing fences for purpose and need; remove redundant fences and retrofit needed fences to allow greater wildlife passage.
13. Provide private landowner incentives, such as conservation easements, for protecting habitat (e.g., prevent ranches from being developed).
14. Purchase important mule deer habitat subject to likely development or other detrimental use (e.g., when habitat connectivity and migration corridors are at stake [Neal et al. 2003]). If necessary, land can be resold with appropriate conservation easements or deed restrictions.
15. Work with conservation groups (e.g., Mule Deer Foundation, Rocky Mountain Elk Foundation) to leverage funds for management, mitigation, or land acquisition projects.
16. Develop informational brochures or internet resources describing methods and activities for reducing impacts of human development. Widely distribute materials to a variety of individuals or groups including county and city planning departments, homeowner associations, conservation groups, livestock associations, developers, state and federal agencies, extension agents, 4-H clubs, automobile associations (e.g., AAA), recreation groups, etc. Potential items to include are cleaning vehicles and equipment to reduce spread of invasive weeds, wildlife-friendly fence design, value of native vegetation,

methods for reducing deer-vehicle collisions, control of dogs, negative impacts of feeding ungulates, etc.

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 structural design characteristics were of secondary importance 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 1981b, Foster and Humphrey 1995, Ruediger 2001, Clevenger and Waltho 2003, Ng et al. 2004). 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 (Clevenger and Waltho 2000, Gordon and Anderson 2003).
6. Underpasses designed specifically for mule deer should be >20 feet wide and 8 feet high (Forman et al. 2003, Gordon and Anderson 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 underpasses (e.g., earthen sides and naturally vegetated) promote use by ungulates (Dodd et al. 2007). In Banff National Park, Alberta, deer strongly preferred (10 times more use) crossing at vegetated overpasses compared to open-span, bridged underpasses (Forman et al. 2003). Based on behavioral traits of pronghorn, it stands to reason that where pronghorn and mule deer coexist, an overpass structure may be more beneficial to both species.
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 natural 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 (Huijser and McGowen 2003).

Water Availability

Background

Precipitation, a major habitat component that regulates and recharges water availability, is a key influence on the distribution and relative abundance of mule deer (McKinney 2003). The IMW experiences a wide variation in annual precipitation. The Great Basin, a large portion of the IMW, is considered the most arid of all North American mule deer habitats. Not only are the average annual precipitation levels low, but precipitation is highly irregular from year to year, and the driest season is the hot summer months, unlike the monsoonal, summer rain season of the Southwest Deserts. Typical mule deer summer range varies from alpine habitats on mountain tops that may receive >30 inches of precipitation annually (with ≥200 in. of snowfall) to arid mountain brush habitats on lower mountain tops and rim rock tables that average only 10-12 each year (with 20 in. of snowfall). Free water sources are either snowmelt fed streams or point source springs recharged by annual precipitation that "leak" from subsurface ground waters. Many of these streams and springs can be ephemeral during drought years and dry during the heat of summer.

Within higher elevation alpine summer ranges, water is not typically a limiting factor for mule deer. Summer range in mid-elevation shrubland communities may be water limiting during prolonged drought cycles and where water sources have been depleted due to manmade diversions, P-J woodland encroachment, and over use by nonnative ungulates. Mid- to low-elevation migratory or transition habitats and year-round habitats may have some areas that have limited water availability during late summer and fall. Wintering deer on low elevation sagebrush and intermixed shrub-woodland habitats are typically not constrained by water because snow is often available and low evaporation rates persist, which allow deer to meet their low water demands during this time of year.

State wildlife and public land management agencies have constructed only a limited number of water developments

or enhancements specifically for the benefit of mule deer in the IMW (Fig. 73). But as more and more impacts occur to mule deer habitats and their associated water sources, wildlife and habitat managers are evaluating opportunities to mitigate these losses.

Figure 73. Mule deer buck utilizing a water development at night in western UT. (Photo courtesy of Randy Larsen and Dr. Jerran T. Flinders/Brigham Young University).

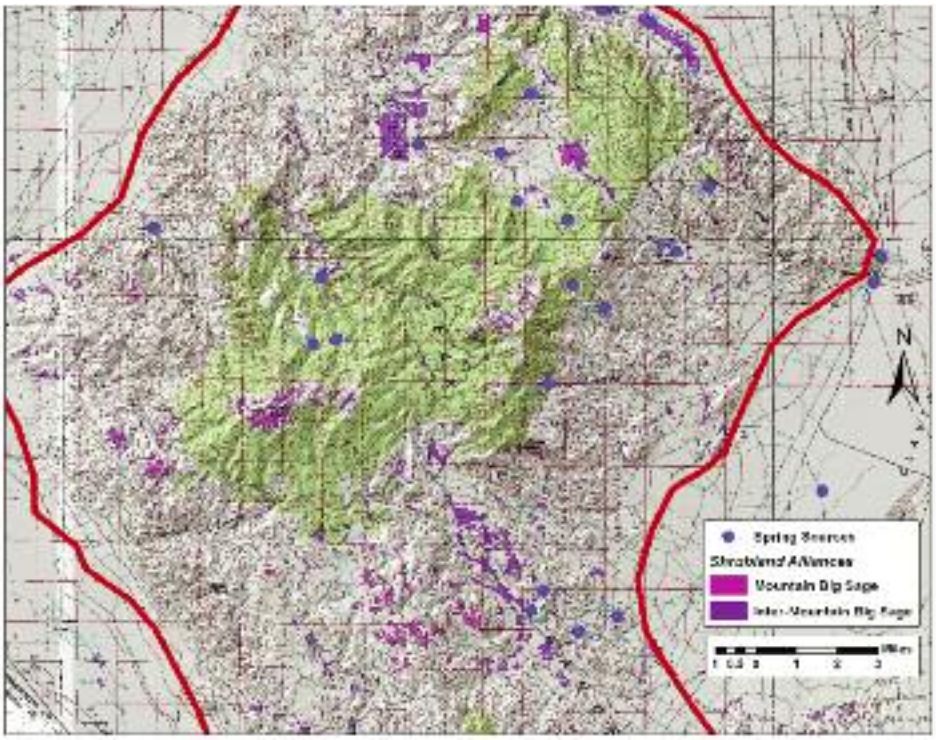

Figure 74. GIS map of mule deer habitat in southwestern Nevada limited by water availability and P/J encroachment to help assess potential opportunity for enhancing water resources. The map displays all point-source springs known to exist along with pixels of mountain big sagebrush plant communities (pink and purple areas based on vegetation modeling by Southwest Regional Gap Analysis and LANDFIRE GIS mapping projects), representing possible mule deer foraging areas. (Provided by Ralph Phenix/NDOW).).

Issues and Concerns

Water Requirements

In northern California, Boroski and Mossman (1996) found mule deer regularly moved ≤1.6 miles to use free water sources. Studies in the Southwest Deserts showed mule deer will move or change home range size in response to changes in water availability across the landscape (Wood et al. 1970, Ordway and Krausman 1986, Rautenstrauch and Krausman 1989). During summer, does have been found to use habitat closer to reliable water sources compared to bucks, likely due to their increased water demands to maintain lactation (Hervert and Krausman 1986, Main and Cobletz 1996). Mule deer in the Southwest Deserts have been found to drink from 0.40 to 1.6 gallons of water/day, with the highest rates occurring during hot summer months (Hervert and Krausman 1986, Hazam and Krausman 1988).

Deer that consume succulent forage high in water content require less free water in order to properly digest and assimilate nutrients (Verme and Ullrey 1972). Clarkson and Sturla (1990) estimated the critical dry-season period is 120 days for big game water use in areas of 10-15 inches of annual precipitation. During this critical dry season, dehydration of vegetation and increased physiological needs compel mule deer to increase their intake of free water. In many of the limited mule deer habitats of the IMW, the combination of low moisture content of late-summer forage and limited water availability may result in reduced food consumption, weight loss, and ultimately lower survival even several months later due to inadequate fat reserves during severe winter conditions.

Adequate Forage to Support More Water Sources

Though some localized mule deer habitats may be water limited, a more overriding limitation may be adequate distribution of quality and quantity of forage. By only adding water, you may do nothing to enhance mule deer use of a particular habitat. Water enhancement should only occur where an evaluation of forage availability shows the area could support more mule deer. Wakeling and Bender (2003) stressed the importance and fundamental role of high quality forage items with readily digestible nutrients to the health and productivity of a mule deer herd.

Geographic Information Systems (GIS) can assist in the initial stages of a

Figure 75. Natural spring source in southern NV used by mule deer and degraded by feral horses and elk. (Photo by Mike Cox/NDOW).

broad landscape evaluation of suitable vegetation associations that mule deer require to meet nutritional demands (Fig. 74). Once distribution of plant associations are known, field investigations should confirm the condition and adequacy of shrubs, availability of forbs and grasses, and existence of appropriate topographic features for enhancing water sources.

Mitigating Impacts to Natural Water Sources

In many cases, the opportunity to enhance water availability for mule deer is based on the fact that historic water sources have been degraded or lost. Throughout the IMW, water sources continue to be impacted by a multitude of uses and landscape changes. If productive mule deer forage still exists in a given habitat but water availability has been compromised, mitigating lost or unavailable water sources can be vital to maintain viable and productive mule deer herds. Factors that have contributed to the elimination or reduced reliability of water sources for mule deer in the IMW are:

1. Landscape-scale plant succession in the form of P-J woodland encroachment and continued increases in density and age of long term P-J stands. Woodland encroachment and succession has substantially increased the draw on subsurface ground water and over time has caused reduced flow rates or even resulted in elimination of natural springs;
2. Man-made diversions of water sources to support mining, agricultural, livestock production and local municipal water uses;
3. Interbasin water transfers to supply large urban centers with municipal water;
4. Competition from other native and nonnative ungulates that degrade spring sources and small streams to the point that water availability and associated riparian forage and hiding cover are no longer adequate (Fig. 75).

In addition to direct impacts to water sources, large scale habitat conversions caused by rangeland fires, invasive weeds, fire suppression in forested habitats, and urban development have caused mule deer to shift or abandon historic home ranges, thereby eliminating use of some water sources. Many of these same landscape-scale habitat changes have also forced mule deer to utilize less palatable forage plants, decadent browse, and cured annual grasses, that increase demands on free water for proper digestion.

Benefits and Negative Aspects of Water Developments

Only limited anecdotal observations exist on the effects of wildlife water developments on mule deer in the IMW. In south-central Oregon, mule deer densities have increased from the addition of several big game guzzlers where natural water sources are scarce due to the region's highly permeable volcanic substrate but mountain shrubs are abundant and of high quality. Water developments in the Southwest Deserts have apparently contributed to increased mule deer populations, indicating water developments can be beneficial in arid habitats when adequate forage is available (Rosenstock et al. 1999).

Mule deer distribution in water-limited landscapes 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 or fawning or hiding cover. This approach effectively increases the habitat carrying capacity and reduces the frequency of long range movements that increase risk of mule deer to predation and other mortality factors. Increased water availability may also allow consumption of a greater variety of foods, including very dry forage. If this results in a better overall nutritional intake for deer, their health and survival would be improved.

Deer will often negotiate hazards such as fences, residential areas, or highways when seeking water sources. In such cases, the benefits of water developments located away from hazards may have more to do with minimizing risks of injury than meeting a physiological requirement. Creating additional water sources to minimize movements by deer to waters associated with hazards may be justified.

Before proceeding with any water development or enhancement for mule deer, consideration should be given to the cost:benefit potential and the possibility of unintended negative consequences. For example, you may want to expand water distribution for a migratory deer herd on a transition area. Instead, you may inadvertently increase use of limited forage in this transition habitat by resident deer before the migratory herd arrives. Depending on access restrictions to a water development built to

benefit mule deer, you may increase habitat use by unintended ungulates such as livestock, feral horses, or even elk.

Broyles (1995) speculated water developments in water-limited habitats might increase predation rates by concentrating prey and providing water for predators. Although conclusive evidence is not available, it is reasonable to assume predation on deer on or near the water development might occur if ambush terrain and cover exists.

Water quality and disease transmission are potential issues that can negate benefits of water developments (Broyles 1995). During summer months wildlife water developments can contain warm, stagnant water that promotes growth of algae and other micro-organisms. Fortunately, available evidence in the Southwest Deserts indicates water quality is not likely to be a major health issue with most wildlife water developments (Rosenstock et al. 2004).

Mule deer that have become dependent on water developments can experience great physical stress if the development goes dry during the critical summer months and they must move great distances to seek other sources of free water (Hervert and Krausman 1986).

Guidelines

Needs Assessment

As part of the North American Mule Deer Mapping Project, wildlife agencies were asked to identify the top 3 limiting factors for each seasonal habitat area delineated. Of the total 225.5 million acres of non-winter habitat within the IMW ecoregion, 11% (25.3 million acres) was identified as being limited by water availability (Fig. 76). Specifically looking at Nevada's habitat in the arid Great Basin portion of the ecoregion, 31% (730,000 acres) has water availability issues. Extensive acreage of water-limited mule deer habitats also exists in south-central and southeastern Oregon and adjacent Idaho.

Gathering information and conducting ground surveys to assess the need and opportunity to enhance water availability are important steps. Questions to answer include:

1. What is the current spacing and seasonality of all existing water sources?
2. What is the general mule deer distribution and seasonal habitat use patterns?
3. Is there adequate forage to support more mule deer?
4. Could water developments disrupt established migratory patterns of mule deer in the area?
5. Are movements to existing water sources causing conflicts or placing mule deer at risk?

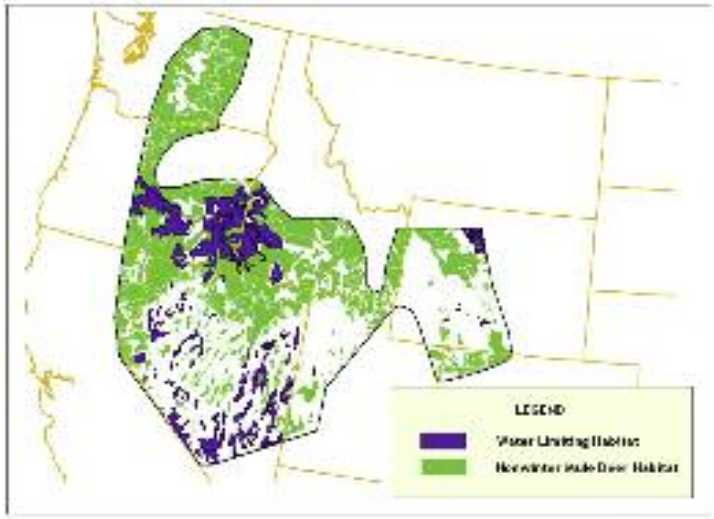

Figure 76. Map of IMW depicting nonwinter mule deer habitat identified by state wildlife agencies as limited by water availability. (Map derived from Mule Deer Mapping Project, MDWG 2005).

Figure 77. Mule deer doe and fawns drinking from natural spring source in western Utah. (Photo courtesy of Randy Larsen and Dr. Jerran T. Flinders/Brigham Young University).

Figure 78. Example of a big game water development in northwestern Nevada. (Photo by Clint Garrett/NDOW).

Site Selection

After the need and opportunity to enhance water availability has been determined, the following should be considered when selecting a site for a water development or enhancement:

1. Before identifying new sites for water sources, improve storage of current water sources and improve site character by reducing visual barriers to predators and enhancing riparian vegetation to make it more attractive to mule deer (Fig. 77);
2. Position multiple water development sites in a fashion similar to a spring complex so animals are not tied to a single water source; this allows for broader use of forage across the landscape and may reduce predation risk;
3. For summer range and more arid seasonal habitats, water sources should be < 3 miles apart so all mule deer habitat is within 1.5 miles of a permanent water source (Brownlee 1979, Dickinson and Garner 1979);
4. Utilize sites that would benefit from both the addition of a water development and a habitat improvement project such as P-J thinning or invasive weed eradication.
5. Place water developments at the head of a draw or ravine to mimic the location of a natural spring source;
6. Always consider having P-J, mountain mahogany, or similar vegetation as edge to a water development site; if no major cover type exists, consider a rocky rim or shelf feature adjacent to proposed water development;
7. Select sites with a strong forb and perennial grass component;
8. If slinging materials in by helicopter is possible, choose a remote site away from roads to reduce vandalism and human disturbance;
9. Without compromising use by mule deer, consider site selection to accommodate other wildlife species use.

Design and Storage Capacity

There are 4 types of big game water developments: 1) artificial collection systems and diversions (guzzler and dugouts), 2) natural water source modifications, 3) wells, and 4) natural spring enhancements. Artificial collection systems primarily use man-made catchment surfaces (metal apron is preferred) to collect water and store it in lined basins or tanks (Fig. 78). Modifications to natural water sources are to increase water retention or enhance storage capacity through use of a reservoir, tank, or dam. Natural water flows can also be diverted to distribute water to other areas making use of elevation gradients. Wells use devices such as windmills or 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 2005b). The most appropriate type and design will depend on a variety of conditions and available water sources.

Storage capacity of a water source is a critical part of the design. How much use do you anticipate and how much rain and snow will you capture in the driest of years? The amount of storage needed is equal to number of animals x number of critical water-use days x amount of water used/animal/day. For example, a 3,600-gallon guzzler can support approximately 12 deer year-round with a 200-day critical water-use period assuming an average of 1.5 gallons used/animal/day. The other important factor to calculate is the amount of water that can be collected from precipitation events for a given site. For every 160 square ft. of catchment surface, 100 gallons will be captured for each 1 inch of rainfall. Evaporation rate of exposed water should be minimal if the guzzler includes enclosed storage tanks and a separate small drinking basin. To estimate the amount of water that can be collected in a given year, use a value less than the average annual precipitation, especially if year-to-year fluctuations are large.

Specific guidelines for big game guzzlers are

1. The preferred design includes a metal apron raised above the ground on metal support beams, thick polypipe from gutter on low side of apron to ≥ 2 1,800-gallon storage tanks buried just below ground level with polypipe delivering water to a separate drinking basin or drinker that has its top edge level with the top edge of the buried tanks (passive design with no valves or float leveling system).
2. Drinker should be small (< 4 feet diameter or length)

Figure 79. Example of a pipe rail fence surrounding the drinker of a big game water development in western Nevada to exclude feral horses and cattle. (Photo by Clint Garrett/NDOW).

to prevent drowning and minimize insect breeding and evaporation. Make sure there's an escape ramp for birds and small mammals.
3. Water developments intended to benefit mule deer primarily should be fenced with wildlife-friendly fencing (3-wire fence to exclude cattle and heavy pipe-rail preferred if excluding feral horses) to restrict use by domestic animals and undesirable species (Fig. 79, Brigham 1990).
4. The waterline between the collection surface and storage drinker should be screened to prevent debris and sediments from clogging the system.
5. Use designs to reduce accumulation of sediments in storage tanks and drinker to eliminate moist substrates used by disease vectors.

Other Considerations
1. Use guzzler materials that will require minimal maintenance and that are not flammable in the likelihood of a future wildfire.
2. Have a clear goal of how the added water source will benefit local mule deer herd.
3. Have adequate collection surface and storage tanks for expected use so that no water hauling is needed during drought years.
4. Minimize visual impacts by blending the structure with the surrounding landscape.
5. Periodically remove organic debris, silt, dead animals, floating algae, and accumulated sediment.
6. In some instances, filing for water rights may be necessary to secure a natural water source for wildlife in the future. Be aware of "beneficial use" water law requirements when filing for wildlife use of a water source. If water laws stipulate that private water sources must "leave water at the source for wildlife," ensure compliance and report violations.

Timber Management

Background

Most of the harvestable timber that exists in the IMW occurs at higher elevations where mule deer spend most of their summers. Higher elevations usually contain ponderosa-Jeffrey pine, Douglas-fir-white fir, fir-spruce, lodgepole, or mixed conifer stands that provide both cover and foraging habitat for deer. Due to lack of disturbance, however, summer range is now occupied by dense, even-aged forest cover that is heavily overstocked. Fire suppression and a decrease in logging activity on public lands over the years have allowed forests to become dense with very little understory vegetation that provides hiding cover and forage for deer.

Historically, logging occurred over large expanses of land where all vegetation was removed, all remaining slash was burned, and sites were densely replanted with seedling trees. The result of this intensive forest system created an overstocked, even-aged forest with a high risk for catastrophic wildfires. As time went on, the negative impacts of large-scale clearcutting became evident. Because negative impacts outweighed the positive impacts of clearcutting, this system is rarely used on public land today. If clearcutting does occur, it is usually on a much reduced scale. The one positive benefit of clearcutting was the increase in early seral species post-harvest. After fire suppression became the norm, clearcutting replaced the effect fires had on the landscape for creating early seral habitat. Currently, there are a variety of even and uneven-aged management strategies that managers have at their disposal for creating quality mule deer habitat.

The overall decline of deer in the IMW is directly related to habitat quality and quantity. Habitat quality for deer is often a function of past management practices such as logging and fire exclusion or natural disturbances such as fire and insect outbreaks (Vavra et al. 2005). For timber management to enhance mule deer habitat quality in the IMW, the following concerns need to be addressed: the adjacency of security cover and forage, establishing early successional species important for forage, understory management post-harvest (including herbicide treatment), lack of fire to recreate and recycle early successional species, and restoration of declining aspen stands which provide a valuable microhabitat to mule deer.

The North American Mule Deer Conservation Plan contains a goal to "evaluate timber management strategies to ensure mule deer habitat quality is maintained or enhanced, or that post-removal restoration is conducted to improve mule deer habitat" (Mule Deer Working Group 2004). Evaluation of timber management in this section is structured to address how various silvicultural treatments may affect mule deer habitat. Issues and concerns surrounding timber harvest in mule deer habitat include: 1) juxtaposition of cover and foraging habitat, 2) succession and early seral species, 3) understory management, 4) forest management strategies, 5) aspen restoration, and 6) pinyon-juniper management. Managers will need to evaluate their specific setting to develop appropriate management actions.

Issues and Concerns

Habitat Adjacency

Timber management can affect quality and quantity of food, shelter, and water; the essential components of mule deer habitat. Mule deer have specific forage requirements due to their relatively small rumen and body size (Hanley 1982, Hoffman 1989). Consequently, management of forested habitats, if designed to address deer nutritional

requirements, needs to be structured not only to increase forage quantity, but more importantly, to increase the amount of high quality forages at crucial times of year. The general pattern of diet composition indicates deer select forages that are higher in digestible energy, even though those forages may be relatively uncommon on the landscape. Regelin et al. (1974) compared forage values between clearcut and uncut areas and found there was little difference between digestibility, protein, or moisture content. However, there was a difference between plant composition and the quantity of forage available between uncut and clearcut areas.

At the same time, mule deer need shelter from weather and hiding cover from predators. Climatic and structural characteristics of forests most likely play a role in deer selecting habitats that meet their thermoregulatory needs. Germaine et al. (2004) found day bed site temperature and canopy closure had the most influence on day bed placement in untreated ponderosa pine forests when compared to ponderosa pine forests that were thinned. High tree densities aid in higher bed concealment and selecting sites to aid thermoregulation appeared a higher priority for deer in untreated forests. Parker and Gillingham (1990) estimated the upper and lower critical temperatures for thermal regulation for mule deer in summer and winter and found sunlight and wind speed were critical to moderating critical body temperatures.

Mule deer move seasonally from summer to winter ranges in response to snow depth, with little use of ranges where snow depth is >18 inches (Gilbert et al. 1970). Forest canopy can intercept snow, resulting in shallower snow depth on the ground, decreased energetic costs of locomotion, and increased forage availability (Poole and Mowat 2005). Where deer winter in forests with deep snow conditions, removal of forest canopy may have deleterious effects on deer survival (Hanley 2004). Canopy cover provided by trees may also reduce wind speeds at the ground and decrease severity of winter conditions. Leckenby and Adams (1986) developed a weather severity index for mule deer winter range and found that juniper cover decreased winter severity by reducing wind speed and providing cover.

The physiological basis for the need for shelter in summer for deer is less clear. Germaine et al. (2004) examined diurnal bed sites for mule deer in thinned and unthinned ponderosa pine stands and found soil temperatures at bed sites under closed canopy were cooler by 7° F. Reynolds (1966) found that deer stayed close to cover and didn't venture far into clearcuts while feeding, presumably for reasons of security.

Although cover is important to deer, habitat quality on potentially or currently forested lands usually is considered in terms of forage (Wallmo and Schoen 1981). Appropriate spacing of foraging and cover habitat that represents seral stages important to deer including early-, mid-, and late-seral habitats should be considered in creating good quality deer habitat. Incorporating more than 1 seral stage will create more edge that deer seem to prefer.

Understory Management and Silvicultural Techniques

Nutritional condition of deer is fundamental to highly productive deer populations, and timber harvest can have a profound effect on forage production on some sites. Biomass of herbaceous vegetation increases after timber harvest in response to decreased competition for sunlight, soil minerals, and precipitation (Moir 1966). For example, in ponderosa pine stands, herbaceous vegetation can increase from near 0 lb/acre when canopy cover is 100% to >678 lb/acre with little conifer canopy cover (Jameson 1967). In Douglas-fir-ninebark (*Physocarpus* spp.) habitat types associated with drier coniferous habitat in the IMW, early seral stages following timber harvest have the greatest species diversity and forage values, but as succession advances, forage biomass drops to lower levels (Steele and Geier-Hayes 1989).

Bitterbrush is important deer forage on summer and winter ranges (Griffith and Peek 1989), and often occurs in the understory of ponderosa pine and lodgepole pine stands. Reestablishment of mature stands of bitterbrush may require ≤40 years (Riegel et al. 2006). Without some type of understory disturbance, the amount of understory shrub species can decrease dramatically. Peek et al. (2001, 2002) concluded that mule deer populations in south-central Oregon declined over a 35-year period due to a long-term decline in biomass of understory forage as canopy cover became closed. Understory productivity is controlled to a large extent by density of the overstory. For every 1 foot increase in pine spacing, there can be a 9 lb/acre increase in grass yield and a 2 lb/acre increase in shrub yield (McConnell and Smith 1977). Using different silviculture methods to meet forest management objectives can benefit not only wildlife species, but also maintain conifer growth in order to yield more wood products from the forest. Below is a summary of how different silvicultural techniques can be used to restore understory shrub species important to deer.

Selection

As an uneven-aged stand management strategy, selection harvesting maintains some level of canopy cover either in a uniform distribution (single tree selection) or by leaving small gap openings (group selection) throughout the stand. There is usually a wide variety of tree age classes represented in an uneven-aged stand, ranging from saplings and poles to late-seral or old-growth trees. Single tree

Figure 80. Example of a clearcut with aggregated habitat retention areas on private timberlands, Tehama County, CA. (Photo by Robert Sullivan/CDFG).

Figure 81. Example of a clearcut with dispersed habitat retention area on private timberlands, Trinity County, CA. (Photo by Jennifer Carlson/CDFG).

selection involves removing dominant trees where crowding exists. This can have a positive impact for deer by increasing forage and providing cover. Group selection units are used to create small openings (1-2 acres) and are also part of an uneven-aged stand management system. Group selection units can mimic natural processes of disturbance such as blow-down areas, fire, or trees that have fallen and are no longer part of the canopy. Because the tree is no longer living and the amount of sunlight has increased, early seral species can become established if there is a seed base present, which can potentially provide hiding cover and forage for deer.

Regeneration (Clearcut)

The clearcut harvest system is an even-aged management strategy that removes all trees in a given area. Clearcutting can have both positive and negative effects on mule deer. From a foraging standpoint, clearcuts provide benefits in the amount of early seral species that establish after harvest. In subalpine fir (*Abies lasiocarpa*) stands that were clearcut, forage production more than doubled and deer spent 72% of foraging time in clearcuts (Wallmo et al. 1972). In Utah, Collins and Urness (1983) found that 18 years after clearcutting lodgepole pine stands, forage production was 13 times that of adjacent uncut stands. Deer obtained most of their digestible energy and protein from clearcuts when given the choice between foraging in clearcuts versus uncut lodgepole pine-spruce-fir forests in Colorado (Regelin et al. 1974). Regelin and Wallmo (1978) found that 20 years after a clearcut treatment, forage availability in cut stands was 36% greater compared to uncut stands. Edgerton et al. (1975) found bitterbrush crown closure decreased 71% following slash disposal in lodgepole pine stands and they anticipated crown closure would exceed pretreatment levels after only 12-15 years.

Recent techniques used in conjunction with clearcuts include retaining habitat that would provide cover for deer in an area otherwise devoid of this type of deer habitat. These areas can include dispersed or aggregated retention, where a certain proportion of trees are retained within the harvest unit. These retention areas act as “islands” that allow some species to recolonize or use an area earlier than would otherwise be possible, given availability of habitat. If an adequate amount of retention is used, it can provide a valuable microhabitat for mule deer (Figs. 80 and 81).

Thinning from Below

Thinning is one of the more widely used silviculture systems in restoring forests throughout the West. Because many forests are overstocked with conifers, thinning is a valuable tool that spaces trees further apart, not only creating more growing space, but also opening the canopy to allow more sunlight through so shrubs, forbs, and herbs can become established (Fig. 82). Vavra et al. (2005) summarized studies on understory productivity and found understory production increased dramatically after ponderosa pine stands were thinned. Standing biomass increased in all categories of understory vegetation. In order to restore mixed conifer-shrub community types, the USFS conducted 2 different restoration treatments to evaluate which would have the best result (Arno and Fiedler 2005). They implemented non-merchantable thinning and comprehensive thinning. The non-merchantable thinning system removed enough

Figure 82. Paired photos of mechanical thinning in a pine plantation in NE California prior to and after treatment to allow for reestablishment of understory forage species. (Photo by Dean Lofthus/Fruit Growers Supply Company).

trees to prevent catastrophic wildfire for ≤20 years. However, it did not open the stand enough to allow establishment of early seral shrubs. Comprehensive thinning produced better results, with increased soil moisture and enough sunlight to establish pine seedlings and early seral shrubs.

Potential negative impacts to deer can occur under a thinning system. Kucera and Barrett (1995) concluded thinning a stand may not benefit deer in the short-term because it decreases the amount of cover and availability of browse. If thinning is applied across hundreds or thousands of acres, the uniformity of the harvest could also present a more long-term effect where it may take some time before cover is reestablished.

Fire

Fire was a frequent occurrence in western forests throughout the IMW in the 19th century, with low intensity burns occurring every 5-10 years (Arno and Fiedler 2005), keeping fuel loads low and maintaining an open understory. Without these burns, forests became overcrowded with conifers and began to take over meadows and grassland areas that were important to wildlife species, as well as decreasing biodiversity on the landscape. Due to fire suppression, early successional vegetation communities in timberlands have declined. Fire as a tool, if used correctly, can have a positive effect on deer populations. Fire can restore and increase grass, forb, and shrub layers that are reduced when timber stand canopies are dense.

Forests are being restored to pre-settlement conditions using not only timber harvesting, but also prescribed burning to reduce fuel loads in the forests of the Southwest. The current conditions in ponderosa pine forests throughout most of the IMW tend to be overcrowded, with heavy fuel loads in the understory which cause high intensity, larger fires. When coupled with logging, fire can have a positive effect on ground cover. Grifantini (1991) found salvage-logged sites that were also burned had less shrub cover, but more hardwood cover, and greater plant diversity than the sites that were not salvage logged.

Grifantini (1991) found unburned old-growth sites had low graminoid, forb, and shrub cover, and vascular plant diversity, but moderate amounts of horizontal screening for hiding cover. However, in burned stands, regardless of post-fire management, there was greater shrub and forb cover and vascular plant diversity than in unburned, old-growth stands. He also found post-fire management influenced early seral stand development and the quantity and diversity of deer forage. He concluded density of shrub species in burned stands increased for the next 12 years following the fire. On the other hand, Busse et al. (2000) found burning to remove slash or thin ponderosa pine stands reduced biomass to below treatment levels for ≤6 years following treatment. Geier-Hayes (1989) found herbaceous and shrub biomass was greater in cut stands compared to uncut stands 10 years after treatment, but lower in a high intensity broadcast burn, compared to low intensity or no burning in Douglas-fir habitat in Idaho. Moore et al. (2006) found no difference in herbaceous biomass in ponderosa pine stands that were thinned from below and then periodically burned over a 12-year period.

Herbicides

Management of the understory by applying herbicides is widely used in site preparation activities before and after timber harvest. Herbicides are also used following a fire and prior to replanting conifer seedlings. Herbicide treatments can have a negative impact on shrubs and alter natural disturbance pathways by removing early successional vegetation that mule deer depend on as forage. Aerial spraying is used regularly on private industrial timberlands, and may adversely impact not only harvested

areas, but also shrub species adjacent to harvested areas due to spray drifting.

Hardwood species, particularly oaks, provide a food source in the fall for mule deer with the acorn crops they generate. Prior to harvest, use of herbicides are also used to kill and remove commercially non-desirable species, including California black oak (*Quercus kelloggii*). A "hack-and-squirt" method is used where oaks are inoculated and removed prior to timber harvest activities, so they do not complicate the harvest of the commercial species at a later date. Removal of oaks also frees up some of the water and soil nutrients for the commercial conifer species to reclaim. In northern California, DiTomaso et al. (1997) evaluated the influence of herbicides on long-term plant richness in treated and untreated sites. Initially, vegetative cover and diversity was drastically greater in untreated plots and lower in treated plots. Therefore, using herbicides over large blocks of land could potentially have a significant impact on deer forage.

Figure 83. This mid-seral aspen stand depicts the optimal functioning condition for mule deer fawning and foraging habitat that managers should strive for that includes a mixed- age class of aspen and healthy understory of forbs and grasses (Photo by Mike Cox/NDOW).

Aspen Restoration

Aspen is the most widely distributed tree in North America (Di Orio et al. 2005), yet it is declining rapidly in the western U.S. Aspen is considered a keystone species and an indicator of ecological integrity and biodiversity (Di Orio et al. 2005). Most of the aspen in the West (75%) occurs in Utah and Colorado.

Factors contributing to aspen decline include >100 years of fire suppression and excessive browsing. As a result, a significant portion of aspen stands have been heavily encroached upon by lodgepole pine and other conifers, thus reducing water availability and site suitability for aspen. Prior to European settlement, natural fire regimes helped balance the abundance and distribution of tree species that occupied a specific area.

Although no wildlife species is totally dependent on habitats dominated by aspen, this cover type adds significantly to species richness of wildlife in areas where it occurs. Aspen habitat can provide some of the best quality food and cover for mule deer (Beck and Peek 2005) (Fig. 83). However, mule deer are not the only species that utilize aspen communities. Other ungulate species, such as moose (*Alces alces*) and elk, use aspen stands for the same purposes as mule deer. Cattle also use aspen stands heavily during summer months for grazing and resting.

Studies recently have focused on the decline of aspen in the West. Di Orio et al. (2005) found a 24% decline in aspen stands in California between 1946 and 1994. In addition, the aspen stand distribution was more fragmented, with smaller units spaced further apart in 1994. On Lassen National Forest in northeastern California, Jones et al. (2005) found 77% of aspen stands were in decline and at risk of being lost. They found that the declines were due to competition from conifers becoming established within aspen stands as a result of loss of natural fire regimes and excessive browsing by livestock (Fig. 84).

Aspen stands, in contrast to coniferous stands, present additional challenges when managing them for succession, age, and forage. Collins and Urness (1983) found that mule deer preferred logged aspen stands over logged and unlogged lodgepole pine and meadow complexes, and total herbage production in logged aspen stands doubled 3 years after logging. Aspen stands can be successfully regenerated with commercial timber harvest (Crouch 1983), but herbivory of regenerating stands can impede growth and establishment of sprouting aspen (Bartos et al. 1994). Conifer removal encourages aspen regeneration in northeast California and in the interior western U.S. (Fig. 84) (Shepperd 2001). Conifer removal should significantly reduce competitive interactions for light and water between

Figure 84. Declining aspen stand in Lassen County, CA with conifer encroachment (upper photo), and the same aspen stand in Lassen County, CA a few years after timber harvest (lower photo). (Photo by Joe Croteau/CDFG).

conifers and aspens. Conifer removal will likely also lead to increased soil temperatures within the stand. These changes should encourage healthy growing conditions for existing stems, as well as encourage the production of new stems by root suckering. Jones et al. (2005) found that using mechanical treatments to remove conifers resulted in a significant increase in total aspen stem density post-treatment.

Restoration of aspen stands should be a management priority in areas of significant decline. Restoration of aspen through mechanical treatments, prescribed burning, and cattle exclusion have been demonstrated in the Sierra Nevada and Cascade regions of northeastern California and across the West.

Pinyon-Juniper

A major vegetation change in the West has been the recent expansion and increase in density of juniper and pinyon, beginning in the late 1800s (Miller and Wigand 1994, Miller et al. 1995, Miller and Rose 1999, Miller et al. 2008). The most rapid expansion occurred between 1880 and 1920, with a decline in expansion rates after 1950 (Miller et al. 2008). Anthropogenic factors, primarily livestock grazing, reduction in natural fires, and even climate change are widely believed to be key factors in woodland expansion (Burkhardt and Tisdale 1976, Heyerdahl et al. 2006, Tausch 1999). Grazing by domestic livestock (primarily cattle and sheep) can increase juniper establishment by distributing seeds and disturbing the soil (Johnsen 1962). Reduction of grass cover can also shift the competitive balance in favor of some woody vegetation (Johnsen 1962). Long-term persistence of savanna communities on ungrazed, relict areas supports the role that grazing has played in successional changes.

Previously burned areas protected from fire are often reinvaded by juniper (Arnold et al. 1964, Barney and Frischknecht 1974). Tress and Klopatek (1987) estimated post-fire succession from grassland to mature woodland sites required approximately 200 years. Prior to European settlement, fires set by aboriginal inhabitants likely played an important role in maintaining southwestern grasslands (West 1984).

Reproduction of pinyon, oneseed juniper (*J. monosperma*), and Utah juniper (*J. osteosperma*) is entirely by seed; consequently, seed dispersal by mammals and birds plays an important role in juniper establishment and expansion (Arnold et al. 1964, Balda and Masters 1980). Junipers, particularly oneseed, are well adapted to animal dispersal, providing large, abundant, readily accessible, and nutritious fruits.

Evidence suggests that recent expansion of P-J woodlands may be attributable to other processes, including climatic changes (Miller et al. 2008). Johnsen (1962) found competition for moisture was important in juniper establishment, particularly during dry years. Short-term drought resulted in decreased juniper seedling survival, but increased site dominance by larger, established trees. Long-term climatic changes have been correlated with elevational and geographic ranges of southwestern P-J woodlands, which have expanded and contracted considerably over the last 2 million years. For example, during the late Pleistocene, P-J woodlands were present on low elevation sites (approx. 800 feet) currently occupied by desert scrub vegetation (Betancourt 1987). Post-European

settlement woodland expansions may, therefore, reflect ongoing species migration and response to climatic change.

Guidelines

Mule deer habitat is highly variable across the IMW and managers will need to consider a variety of factors when designing timber management activities. Silvicultural treatment of forest stands has the potential to greatly increase forage production, and with careful management of fire and herbicide use, high quality forage can be created for mule deer in and adjacent to timbered sites. At the same time, providing cover for fawns and adults are important considerations when designing timber management strategies. Finally, the manager should note that there is no one recipe for creating mule deer habitat and what currently exists on the landscape should be looked at critically before a desired outcome is pursued. The following are general guidelines to consider when planning timber harvest to create and optimize mule deer habitat while still pursuing a market for wood products.

A. Habitat Adjacency

1. On winter ranges where snow depth is >18 inches, insure integrity of canopy to intercept snow (Day et al. 2000).
2. Maintain patches of saplings and pole trees to provide hiding cover in clumps >0.1 acre interspersed within thinned ponderosa pine stands (Germaine et al. 2004).
3. Maximize time intervals between underburnings to remove slash and promote shrub regrowth, particularly where bitterbrush is a dominant shrub.
4. Open closed-canopy forests to promote growth of herbaceous vegetation (Peek et al. 2001).
5. Implement management strategies that promote development of a diversity of understory species to provide adequate nutrition to deer later in the season.
6. Optimize the landscape for mule deer to include 40% cover and 60% foraging habitat.
7. Provide 1 sapling thicket/100 acres for bedding cover, and retained basal areas of 40-80 square feet/acre (Clary et al. 1975).

B. Understory Management and Silvicultural Techniques

For many management objectives, a mix of dispersed and aggregated retention will likely provide the greatest ecological and microclimatic benefits. With advances in technology, there are many more options in using silviculture techniques to create a desired future condition using both even- and uneven-aged management.

1. Selection
- In an uneven-aged management strategy, where openings are fewer, incorporate group selection units ≥1-2 acres to establish early successional shrub species important as deer forage.
- Consideration should be given to the juxtaposition of group selection openings so that a mosaic of openings and timbered stands are present throughout the stand.

2. Regeneration (Clearcuts)
- Create habitat retention areas within clearcut units where approximately 2.5 acres of the clearcut is retained in a cluster of trees. Retention areas provide cover for deer in places that are normally used for foraging only.
- Maintain approximately 15% of the harvest area in green-tree retention to counteract harvest impacts.

3. Thinning
- Retain clusters of dense vegetation that maintains hiding cover within the thinned stand (Kucera and Barrett 1995).
- Retain patches of ≥25 unthinned acres in a block for every 200 acres of the project (approx. 12%) to provide cover for deer.
- Leave blocks of untreated areas adjacent to meadows and streams or other habitat features that benefit deer. Screens of unthinned material that are approximately 100 feet wide should be retained along roads.

4. Fire
- Prescribed fire should be applied at times of the year when the greatest likelihood of achieving the desired plant response will be achieved. Dry season burns (fall) result in more effective regeneration of shrub species from seed than moist season burns.
- Prescribed fires to enhance deer habitat should be 400 acres, planned as a component of a watershed approach to establish mosaics in varying successional stages, and conducted where wildlife value is a priority (as opposed to fuel reduction or timber stand improvement).
- Post-fire deer habitat recommendations (Grifantini 1991): 1) minimize use of post-salvage burns, 2) disperse post-fire management schemes throughout the landscape, and 3) maintain all available screening cover in locations likely to have high deer use.

5. Herbicides (Di Tomaso et al. 1997)
- Identify areas that may be more beneficial for mule deer and delay spraying unless absolutely necessary. Retaining clumps of vegetation that are not treated with herbicide would be beneficial.
- Create a mosaic of suitable habitat adjacent to cover on a percentage of the watershed being treated.

C. Aspen Restoration

Pre-project planning to restore aspen should be guided by the following questions (Sheppard 2004):

1. Is the aspen stand in decline, as evidenced by abundant dead trees, downed logs, or holes in the overstory canopy? If not, the stand may be adequately stocked with the correct hormonal balance and may not be

attempting to regenerate;
2. Are aspen suckers present in the stand? If so, the stand may be naturally regenerating and not in need of management intervention;
3. If the stand is in decline and no successful suckers are present, are scattered browsed or clipped sprouts present in the understory? If browsed sprouts are present, fencing the stand will allow them to release and grow;
4. If no browsed suckers are evident, competing trees, or dense understory vegetation may be preventing an adequate environment for sucker growth. Removing competing vegetation may initiate suckering without cutting any aspen;
5. Declining clones with no suckers may be an indication that the stand is a root rot epicenter, which cannot be fixed with management action.

Management techniques that can be used to restore aspen (Sheppard 2004):
1. Clearfell-coppice harvest: requires large areas of aspen to be applied successfully, does not work well for small stands unless cut units are completely fenced from browsing;
   - Must be a commercial market for aspen nearby;
   - Introduces a new age class of aspen, but requires removing old trees which have high ecologic value;
2. Mechanical root stimulation: Severing the lateral root at a distance from the parent tree while still maintaining the old tree component;
   - Relies on root habit of aspen to establish suckers in locations that have a better growth environment;
   - Bulldozed areas produce more suckers than cut areas;
   - More suckers are produced in fenced areas compared to areas where logging slash is left as a deterrent to browsing;
   - Ripping techniques can be used to sever lateral roots 8-10 yards away from the parent tree;
3. Prescribed fire: fire is used to provide hormonal stimulation of sucker production by killing overstory stems and injuring lateral roots;
   - Removes competing vegetation;
   - May not provide protection to new sprouts if area is not large enough to sustain local browsers;
   - Difficult to burn completely due to lush vegetation and moisture associated with aspen stands;
   - Time fire when fuels are dry, and distribute fuels through aspen stand for fuel continuity.

Recommendations for aspen restoration that CDFG (2008) has provided in timber harvest plans:
1. All conifers (both merchantable and non-merchantable) within existing aspen stand and within 100 feet of all aspen stems shall be removed;
2. All tree tops and associated slash shall also be removed from the stand, employ whole-tree yarding;
3. Equipment use within the stand shall only occur to the extent necessary for conifer and slash removal;
4. All existing aspen stems shall be protected to the extent feasible during harvest operations;
5. Remove remaining non-merchantable trees encroaching into meadows that contain aspen through biomass silviculture or hand felling.

Recommendations from Jones et al. (2005) to consider when aspen restoration is a management goal:
1. Mechanical harvesting of conifers acts as a slight disturbance mechanism (hormonal stimulation), but its primary function is to create a proper growth environment (sunlight) for aspen regeneration;
2. Pre-treatment density of aspen may be a useful selection tool for treatment application;
3. Aspen density increased 4 years after treatment compared to control stands for all size classes; however a decrease in size class 2 and 3 occurred during the first 2 years following treatment;
4. Burning hand piles near the aspen trees will kill roots, and expose dominant trees to sooty bark canker.

D. Pinyon-juniper
1. Recovery of sagebrush-steppe habitats dominated by P-J encroachment involves 1) reduction of tree densities, 2) establishment of conditions that encourage grasses, forbs, sagebrush and other browse plants, and 3) maintenance of shrublands to prevent future conversion to woodland. Following tree removal, a combination of actions, including crushing of cut trees and shrubs, selective application of herbicides, seeding of grasses, forbs and shrubs, and burning regimes, may be used to prolong the site in an early to mid seral stage.
2. In closed canopy pinyon-juniper woodlands. create openings by felling trees with a hyrdo-axe (Fig. 85). Windrow felled trees into piles for later burning. Young juniper and pinyon trees (< 50 years) can be killed with surface fires, but larger trees are difficult to burn, and these stands are generally devoid of fine fuel for low intensity surface fires. Windrows of dried, cut trees are used to generate sufficiently hot fires in crowns that will kill most remaining live trees on a site.
3. Chaining is relatively inexpensive and provides soil disturbance in preparation for seeding. Although Tausch and Tueller (1995) projected that treated sites will revert to pre-chaining levels of production and deer use within 20 years following treatment, the number of years to reversion depends heavily on post-treatment land management practices. Sites that have greater understory present before chaining occurs exhibit greater vegetation response and are more heavily used by deer after treatment.

4. Roller-chopping is a cost effective method to control regenerating woody vegetation and improve site productivity on previously chained sites (Sorenson 1999). The roller-chopper is not designed for use in closed-canopy P-J woodlands. The pipe harrow is gaining wide use to retard pinyon and juniper encroachment in Wyoming sagebrush parks. Because encroaching trees are younger, they have a small trunk diameter and can be dragged out of the ground with the debris pile and fins on the pipes of the pipe harrow. This practice also works well to improve Wyoming sagebrush plant communities by creating multiple plant age classes and by preparing the soil for the application of seed.
5. Sites with moderate woody encroachment can be recovered through lopping pioneering trees using chainsaws and scattering them. The spaces occupied by felled trees will be quickly repopulated through natural reseeding from surrounding shrubs, grasses, and forbs if they are present. This method causes little disturbance of seedbeds, and therefore has promise for areas threatened by cheatgrass invasion. However, thinning by hand is more expensive than chaining (Chadwick et al. 1999).
6. Herbicides that target annual plants are effective in controlling cheatgrass on P-J sites following thinning with fire or mechanical treatments.
7. Managers should consider the need to maintain cover for deer as they plan to manage sites with woody encroachment where longer winters with more severe weather occur. Retain dense P-J stands within 200 meters of treated openings (deer foraging areas) to maintain cover during severe winter weather periods (Fairchild 1999).
8. Consider the value of pinyon pine nuts as a component to mule deer diets in low precipitation zones where deer herds have limited browse species on transition and winter ranges.

Figure 85. Hydro-axe mounted on a bobcat compact track loader (upper photo) causes very little soil disturbance and can be used on top of snow in the winter to lessen soil disturbance to cut pinyon and juniper trees flush with the ground. The lower photo is the result of the hydro-axe in felling trees on a moderately dense P-J stand. (Photos courtesy of Kreig Rasmussen/USFS – Fishlake).

SUMMARY

Wildlife and land managers are faced with a daunting task of maintaining vegetation communities, wildlife habitat, and mule deer numbers at levels sufficient to ensure viability of mule deer populations throughout their range and to satisfy society's various requirements. Clearly, the key to approaching this task is instituting nationwide policies affecting land use practices such as livestock grazing, urban sprawl, transit systems, appropriate fire management, timber harvest, and mineral exploration. Land management agencies responsible for a significant portion of the mule deer's range must prioritize wildlife habitat management issues higher than they are now. Continued efforts to work with land management agencies and private landowners on a cooperative basis to maintain and enhance wildlife habitat is critical.

Issues affecting habitats key to mule deer in the West are both simple and complex. We can identify factors such as fire suppression, excess livestock grazing, and mineral exploration that have had significant negative impacts on mule deer habitats. We can even provide solutions to address those issues. This is simple. Institutionalizing and implementing solutions on a scale large enough to make a difference is complex.

In most cases, several factors are working in concert causing mule deer declines. While discussion continues on how important different factors are in affecting mule deer and other species, it is important to keep in mind what primarily drives population densities. Wallmo (1981:238) stated: "In my view, the only generalization needed to account for the mule deer decline throughout the West is that practically every identified trend in land use and plant succession on the deer ranges is detrimental to deer. Hunting pressure and predators might be controlled, and favorable weather conditions could permit temporary recovery, but deer numbers ultimately are limited by habitat quality and quantity."

Management should be directed toward protecting and enhancing sagebrush, bitterbrush, and other important browse species for mule deer, particularly on winter ranges. Hobbs (1989) developed a model linking energy balance with mule deer survival, and implicit in the model was availability of a shrub component with winter snow on the ground. Prevention and early suppression of summer wildfire on deer winter ranges should be given higher priority, because the resultant invasion by annuals, such as cheatgrass, decrease the value of the deer range. Efforts to enhance deer ranges through plantings of desirable browse species should continue to be evaluated and implemented where feasible.

In the more northern reaches of this region, forest management is key for deer management. The goal should be to maintain significant areas of forest in early stages of succession (Wallmo 1981). Balancing this need for secondary succession while retaining sufficient cover for security needs and winter use is a challenge.

More effective management of livestock grazing during summer months is needed throughout the ecoregion, particularly in important riparian and aspen habitats. Land management agencies typically have adequate standards and guidelines for these key habitats. However, monitoring and compliance with those standards are often not met. Where livestock graze on deer winter ranges, allocating forage to mule deer and other wildlife is needed to ensure overuse of important browse species does not occur.

Increasing efforts to control and reduce P-J invasion are needed through the use of fire or mechanical treatments. Further efforts to develop a cost-effective approach are needed. Treated areas should not be at such a large scale that they eliminate cover within a reasonable distance for mule deer.

It seems apparent after all these years, and studies, and successes and failures, that we know what to do, we just can't do it on a large enough scale, or circumstances don't allow us to do what is necessary. The need is to apply management practices to reverse current trends in vegetative communities and land uses over large areas on either watershed or landscape scales. We need to manage in order to make mistakes in order to learn. As Mitchell and Freeman (1993:10) put it: "No matter how much data are collected and analyzed, some level of ignorance will always exist. A land manager must make decisions with the information available and continue to learn from both mistakes and accomplishments."

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APPENDIX A.

Plants and Animals Listed in Document

Alphabetical listing by category of [Common name (*scientific name*)] species cited in the text.

Trees and Shrubs

Aspen, Quaking (*Populus tremuloides*)
Bitterbrush, Antelope (*Purshia tridentata*)
Bitterbrush, Desert (*Purshia glandulosa*)
Ceanothus (*Ceanothus* spp.)
Chokecherry (*Prunus virginiana*)
Cliffrose (*Purshia mexicana*)
Douglas-fir (*Pseudotsuga menziesii*)
Fir, Subalpine (*Abies lasiocarpa*)
Fir, White (*A. concolor*)
Juniper (*Juniperus* spp.)
Juniper, Oneseed (*J. monosperma*)
Juniper, Utah (*J. osteosperma*)
Juniper, Western (*J. occidentalis*)
Manzanita, Greenleaf (*Arctostaphylos patula*)
Mountain-mahogany, Curl-leaf (*Cercocarpus ledifolius*)
Mountain-mahogany, True (*C. montanus*)
Ninebark (*Physocarpus* spp.)
Oak, California Black (*Quercus kelloggii*)
Oak, Gambel (*Q. gambelii*)
Pine, Bristlecone (*Pinus longaeva*)
Pine, Jeffrey (*P. jeffreyi*)
Pine, Limber (*P. flexilis*)
Pine, Lodgepole (*P. contorta*)
Pine, Ponderosa (*P. ponderosa*)
Pinyon (*Pinus* spp.)
Rabbitbrush (*Ericameria* spp., *Chysothamnus* spp.)
Rabbitbrush, Rubber (*Ericameria nauseosa*)
Rose, Wild (*Rosa* spp.)
Sagebrush (*Artemisia* spp.)
Sagebrush, Big (*Artemisia tridentata*)
Sagebrush, Black (*Artemisia nova*)
Sagebrush, Low (*Artemisia arbuscula*)
Sagebrush, Mountain Big (*Artemisia tridentata vaseyana*)
Sagebrush, Silver (*Artemisia cana*)
Sagebrush, Tall Threetip (*Artemisia tripartita tripartita*)
Sagebrush, Wyoming Big (*Artemisia tridentata wyomingensis*)
Saltbush, Fourwing (*Atriplex canescens*)
Serviceberry (*Amelanchier* spp.)
Skunkbush (*Rhus trilobata*)
Snowberry (*Symphoricarpos* spp.)
Snowbrush (*Ceonathus velutinus*)
Spruce (*Picea* spp.)
Willow (*Salix* spp.)
Winterfat (*Krascheninnikovia* spp.)

Forbs and Grass

Alfalfa (*Medicago sativa*)
Aster (*Aster* spp.)
Balsamroot, Arrowleaf (*Balsamorhiza sagittata*)
Bluegrass (*Poa* spp.)
Bluegrass, Sandberg (*Poa secunda*)
Brome, Smooth (*Bromus inermis*)
Buckwheat (*Eriogonum* spp.)
Burnet, Small (*Sanguisorba minor*)
Cheatgrass (*Bromus tectorum*)
Cryptantha (*Cryptantha* spp.)
Fescue (*Festuca* spp.)
Fescue, Idaho (*Festuca idahoensis*)
Globemallow (*Sphaeralcea* spp.)
Goldenweed (*Machaeranthera* spp.)
Grama, Blue (*Bouteloua gracilis*)
Grass, Mutton (*Poa fendleriana*)
Junegrass (*Koeleria macrantha*)
Knapweed, Diffuse (*Centaurea diffusa*)
Knapweed, Spotted (*Centaurea biebersteinii*)
Kochia, Forage (*Kochia prostrata*)
Lupine (*Lupinus* spp.)
Medusahead (*Taeniatherum caput-medusae*)
Mustard, Tumble (*Sisymbrium* spp.)
Needle and Thread (*Hesperostipa comata*)
Needlegrass (*Stipa* spp., *Heterostipa* spp. *Achnatherum* spp.)
Needlegrass, Thurber's (*Achnatherum thurberianum*)
Orchardgrass (*Dactylis glomerata*)
Penstemon (*Penstemon* spp.)
Phlox (*Phlox* spp.)
Ricegrass, Indian (*Achnatherum hymenoides*)
Sainfoin (*Onobrychis viciifolia*)
Sagebrush, Fringed (*Artemisia frigida*)
Sagewort (*Artemisia* spp.)
Salt cedar (*Tamarix pentandra*)
Skeletonweed, Rush (*Chondrilla juncea*)
Snakeweed (*Gutierrezia* spp.)
Spurge, Leafy (*Euphorbia esula*)
Squirreltail, Bottlebrush (*Elymus elymoides*)
Star-thistle, Yellow (*Centaurea solstitialis*)
Wheatgrass (*Agropyron* spp., *Pseudoroegneria* spp., *Pascopyrum* spp.)
Wheatgrass, Bluebunch (*Pseudoroegnaria spicata*)
Wheatgrass, Crested (*Agropyron cristatum*)
Wheatgrass, Western (*Pascopyrum smithii*)
Wildrye, Basin (*Leymus cinereus*)

Animals and Other

Bison (*Bison bison*)
Cattle, Domestic (*Bos taurus*)
Deer, White-tailed (*Odocoileus virginianus*)
Deer, Coues White-tailed (*Odocoileus virginianus couesi*)
Deer, Mule (*Odocoileus hemionus*)
Elk (*Cervus elaphus*)

Fungi (*Glomus* spp.)
Goat, Domestic (*Capra hircus*)
Goose, Canada (*Branta canadensis*)
Horse, Domestic (*Equus caballus*)
Moose (*Alces alces*)
Sapsucker (*Sphyrapicus* spp.)
Sheep, Domestic (*Ovis aries*)

APPENDIX B.

Important Intermountain West Mule Deer Forage Plants

Alphabetical listing of important forage plants [Common name (*scientific name*)] eaten by mule deer in the Intermountain West. Names based on USDA (2008).

Trees and Shrubs

Aspen, Quaking (*Populus tremuloides*)
Bitterbrush, Antelope (*Purshia tridentata*)
Bitterbrush, Desert (*Purshia glandulosa*)
Ceanothus (*Ceanothus* spp.)
Chokecherry (*Prunus virginiana*)
Cliffrose (*Purshia mexicana*)
Mountain-mahogany, Curl-leaf (*Cercocarpus ledifolius*)
Mountain-mahogany, True (*C. montanus*)
Oak, California Black (*Quercus kelloggii*)
Oak, Gambel (*Q. gambelii*)
Pine, Jeffrey (*Pinus jeffreyi*)
Pine, Ponderosa (*P. ponderosa*)
Rabbitbrush (*Chysothamnus* spp., *Ericameria* spp.)
Rabbitbrush, Rubber (*Ericameria nauseosa*)
Rose, Wild (*Rosa* spp.)
Sagebrush, Big (*Artemisia tridentata*)
Sagebrush, Black (*Artemisia nova*)
Sagebrush, Low (*Artemisia arbuscula*)
Sagebrush, Mountain Big (*Artemisia tridentata vaseyana*)
Sagebrush, Silver (*Artemisia cana*)
Sagebrush, Tall Threetip (*Artemisia tripartita tripartita*)
Sagebrush, Wyoming Big (*Artemisia tridentata wyomingensis*)
Saltbush, Fourwing (*Atriplex canescens*)
Serviceberry (*Amalanchier* spp.)
Skunkbush (*Rhus trilobata*)
Snowberry (*Symphoricarpos* spp.)
Willow (*Salix* spp.)
Winterfat (*Krascheninnikovia* spp.)

Forbs and Grass

Alfalfa (*Medicago sativa*)
Aster (*Aster* spp.)
Balsamroot, Arrowleaf (*Balsamorhiza sagittata*)
Bluegrass (*Poa* spp.)
Bluegrass, Sandberg (*Poa secunda*)
Buckwheat (*Eriogonum* spp.)
Burnet, Small (*Sanguisorba minor*)
Cryptantha (*Cryptantha* spp.)
Fescue (*Festuca* spp.)
Fescue, Idaho (*Festuca idahoensis*)
Globemallow (*Sphaeralcea* spp.)
Goldenweed (*Machaeranthera* spp.)
Grama, Blue (*Bouteloua gracilis*)
Grass, Mutton (*Poa fendleriana*)
Junegrass (*Koeleria macrantha*)
Lupine (*Lupinus* spp.)
Needle and Thread (*Hesperostipa comata*)
Needlegrass (*Stipa* spp., *Heterostipa* spp. *Achnatherum* spp.)
Penstemon (*Penstemon* spp.)
Phlox (*Phlox* spp.)
Ricegrass, Indian (*Achnatherum hymenoides*)
Sainfoin (*Onobrychis viciifolia*)
Sagebrush, Fringed (*Artemisia frigida*)
Sagewort (*Artemisia* spp.)
Snakeweed (*Gutierrezia* spp.)
Squirreltail, Bottlebrush (*Elymus elymoides*)
Wheatgrass, Bluebunch (*Pseudoroegnaria spicata*)
Wheatgrass, Western (*Pascopyrum smithii*)
Wildrye, Basin (*Leymus cinereus*)

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