



HABITAT GUIDELINES FOR MULE DEER
CALIFORNIA WOODLAND CHAPARRAL ECOREGION

A Product of the
Mule Deer Working Group
Sponsored by the
Western Association of Fish and Wildlife Agencies

2007

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Cover photo by: California Department of Fish and Game (CDFG)

Suggested Citation: Sommer, M. L., R. L. Barboza, R. A. Botta, E. B. Kleinfelter, M. E. Schauss and J. R. Thompson. 2007. Habitat Guidelines for Mule Deer: California Woodland Chaparral Ecoregion. Mule Deer Working Group, Western Association of Fish and Wildlife Agencies.

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INTRODUCTION

Mule and black-tailed deer (collectively called mule deer, *Odocoileus hemionus*) are icons of the American West. Because of their popularity and wide distribution, mule deer are one of the most economically and socially important animals in western North America. A survey of outdoor activities by the U.S. Fish and Wildlife Service in 2001 showed that over 4 million people hunted in the 18 western states. In 2001 alone, those hunters spent almost 50 million days in the field and 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 can be used as a barometer 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 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 deer and their habitats. Within the geographic distribution of this species, however, areas can be grouped into “ecoregions” within which deer populations share certain similarities regarding issues and challenges that land managers must face. Within these guidelines we have designated seven separate ecoregions: 1) California Woodland Chaparral, 2) Colorado Plateau Shrubland and Forest, 3) Coastal Rain Forest, 4) Great Plains, 5) Intermountain West, 6) Northern Forest, and 7) Southwest Deserts. The diversity among the ecoregions presents different challenges to deer managers and guidelines for managing habitat must address these differences (deVos et al. 2003). In many ecoregions, water availability is not a major limiting habitat factor. However, in others, such as the Southwest Deserts ecoregion, water can be important. A significant factor affecting deer population fluctuations in the Northern


Forest is severe winterkill. Winterkill is not a problem in the Southwest Deserts, but heavy grazing and drought can seriously impact populations in this ecoregion.

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

Across these different ecoregions, the core components of deer habitat are consistent: water, food, and cover. An important aspect of good mule deer habitat is juxtaposition of these components; they must be interspersed in such a way that a population can derive necessary nutrition and cover to survive and reproduce. We have learned much about mule deer foods and cover, but more remains to be learned. For example, cover is not a simple matter; the relief 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 the many issues affecting their populations.

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

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 (Schaefer et al. 2003). Disturbance is a key element to maintaining high



quality deer habitat, especially where shrubs compose the climax community. In the past, different fire cycles and human disturbance, such as logging, resulted in higher deer abundance than we see today. Although yearly weather patterns, especially precipitation, have a short-term influence on deer populations, landscape-scale habitat improvements promote long-term gains in mule deer abundance in many areas. Mule deer are known as a “K-selected” species, meaning that populations will have a tendency to increase until carrying capacity is reached. Carrying capacity is considered the number of individuals in a population that the resources of the habitat can support. If deer populations remain at or grow beyond carrying capacity they begin to impact their habitats in a negative manner. Carrying capacity is affected over the long-term by drought conditions and vegetation succession. Even when a drought period ends, a time-lag effect can cause carrying capacity to remain low for many years. This may 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 occurs on government owned land (federal, state, local) and must be achieved under authority of land management agencies. Mule deer habitats are facing substantial threats from a wide variety of human-related activities on public lands. 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, meant to improve game species populations, have also benefited many other un hunted 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, such as mule deer, are now sometimes 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 hard question to answer. Treated areas must be sufficiently large to produce a “treatment” effect. There is no one “cookbook” rule for scale of treatment. However, the manager should realize the effect of the treatment applied properly can be larger than the actual number of acres treated because deer will move in and out of treatment areas. In general, several smaller treatments in a mosaic or patchy pattern are more beneficial than one 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 designed to work as part of an overall large-scale habitat improvement strategy. For example, treatments should begin in an area where benefits to deer will be greatest and then subsequent habitat improvement activities can be linked to this core area.

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

Sections of these guidelines were adapted from the Habitat Guidelines for Mule Deer – Southwest Deserts Ecoregion (Heffelfinger et al. 2006).

THE CALIFORNIA WOODLAND CHAPARRAL ECOREGION

DESCRIPTION

The California Woodland Chaparral Ecoregion includes the Coast Range of California from the San Francisco Bay, south to northern Baja California, Mexico, and lower elevations on the west slope of the Sierra Nevada adjacent to the Central Valley. A modified version of chaparral extends eastward into central Arizona (Fig. 1). The vegetation communities in this ecoregion in California are characterized by oak woodland and chaparral with an annual grass-forb understory, annual and perennial grasslands, small amounts of riparian habitat, and other minor types (deVos et al. 2003). Of 51 California habitat types described by Mayer and Laudenslayer (1988), about 30 occur in this ecoregion. Shrub dominated habitats include Mixed Chaparral, Chamise (*Adenostoma fasciculatum*)-Redshank (*Adenostoma sparsifolium*) Chaparral and Coastal Scrub. Dominant woodland habitats include Coastal Oak (*Quercus agrifolia*), Valley Oak (*Quercus lobata*), and Blue Oak (*Quercus douglasii*) Woodlands, Blue Oak-Gray Pine (*Pinus sabiniana*), Montane Hardwood, and Montane Hardwood Conifer, with localized habitats dominated by redwood (*Sequoia sempervirens*) and ponderosa pine (*Pinus ponderosa*). There are also relatively small areas of vitally important Montane and Valley Foothill Riparian and Fresh Emergent Wetland habitat. Throughout the ecoregion, large areas have been converted to Cropland, Orchard, and Urban use. The chaparral in Baja California is an extension of vegetation in Southern California (Mearns 1907, Leopold 1959). In Arizona chaparral species are structurally similar, although less complex than the chaparral in California (Urness 1981). Herbaceous growth is generally sparse, with occasional bursts of short-lived annual growth (Swank 1958).

Soils in most of the chaparral of this ecoregion are shallow, poorly developed, and in some cases nonexistent (Swank 1958, Urness 1981). Better soil conditions support various oak woodland habitats. Selenium deficiencies are widespread in deer herds of California and are common in parts of this ecoregion (Flueck, 1994). Low selenium levels may contribute to fawn mortality in deer in many parts of California (Oliver et al. 1990). Deer may benefit from placement of selenium mineral blocks, however use of salt or mineral blocks containing sulfur should be avoided. Increased exposure to sulfur may interfere with selenium function, resulting in an increased demand for selenium (Fleming et al. 1977, Flueck 1994).

Climate of the California Woodland Chaparral Ecoregion in California is characterized by hot dry summers and mild wet winters with periodic droughts (Cronmiller and Bartholomew 1950). In California most precipitation falls from November to April, varying from 8 to 30 inches annually depending on elevation and latitude, and can also vary greatly from year to year. Arizona chaparral climate is

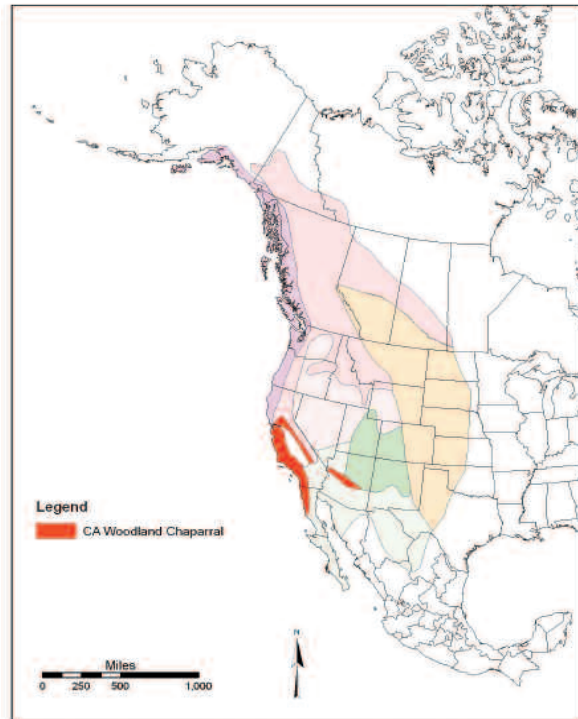


Figure 1. The California Woodland Chaparral Ecoregion (Sue Boe/Arizona Game and Fish).

similar, except that there are two wet seasons – one in winter and one in late summer (Wallmo et al. 1981). The critical dry period occurs earlier in the year, beginning around 1 May and extending until the summer rains begin in July (Swank 1958). Chaparral in Arizona receives from 14 to 25 inches of annual precipitation, of which 40% occurs as summer rains (Swank 1958, Hibbert 1979). Winter snows occur in the California Woodland Chaparral Ecoregion, but are not generally considered a limiting factor for deer. However, in some years unusually low winter temperatures over prolonged periods, or cold rains during fawning season, may increase mortality rates.

ECOREGION-SPECIFIC DEER ECOLOGY

The term mule deer applies to all subspecies of *O. hemionus*, including black-tail deer. California Woodland Chaparral Ecoregion is home to at least 4 subspecies of mule deer; Columbian black-tailed deer, *O. h. columbianus*, California mule deer, *O. h. californicus*, Southern mule deer, *O. h. fuliginatus*, and Rocky Mountain mule deer, *O. h. hemionus*. The following describes the subspecies distribution within this ecoregion, however for a complete subspecies map see figure 2. Columbian black-tailed deer inhabit the most northwestern portion of the ecoregion along the coast of California to southeast of Monterey Bay, where they intermix with California mule deer (Wallmo 1981). The range of California mule deer runs from this area south to the Los Angeles vicinity, and extends around the

edge of the Central Valley into the mountains of central and southern California. Southern mule deer occupy the area south of Los Angeles and into Baja California (Wallmo 1978). The most common mule deer occupant of the Arizona chaparral is the Rocky Mountain mule deer, although the edge of the chaparral coincides with the delineation of the boundary of desert mule deer, *O. h. eremicus* (Heffelfinger 2000), previously known as *O. h. crooki* (Wallmo et al. 1981).

Considerable variation exists in fawning periods of California deer, and those of the California woodland chaparral are no exception, with births ranging from April through August. Peak fawning period of Columbian black-tailed deer typically occurs in April and May, which is as much as one month earlier than most mule deer (Wallmo 1978, Schauss and Coletto 1986, unpublished California Department of Fish and Game report). Southern mule deer may breed as early as black-tailed deer, and in coastal San Diego County the breeding period starts earlier and ends later than for other herds in California (Schaefer 1999). However, Southern mule deer generally deliver their fawns during May to August, peaking in mid May and June. Salwasser et al. (1978) found that California mule deer of the North Kings River herd (southern Sierra Nevada foothills) began fawning in June, which is typical of this subspecies. Peak of parturition tends to occur in mid June and July for California mule deer.

In the Arizona chaparral the peak breeding period takes place from mid-December through mid-January. Fawning occurs from mid July through the first week in September, with most during the last week of July and first week in August (Hanson et al. 1955, Heffelfinger 2006, Swank 1958).

Except for the Sierra Nevada and San Geronio mountains populations, most deer populations in this ecoregion are not migratory and live yearlong in habitats dominated by a mix of oak woodland shrub and tree species, and chaparral with a diversity of shrub species (Longhurst et al. 1952, Nicholson 1995). Non-migratory, or resident deer, exhibit seasonal shifts within home ranges to take advantage of microclimate and vegetative differences between south and north facing slopes (Taber and Dasmann 1958). Deer densities tend to decrease from north to south in this region of California (Longhurst et al. 1952). The most nutritionally demanding time of year for deer in the California Woodland Chaparral Ecoregion occurs in late summer and early fall before onset of fall and winter rains

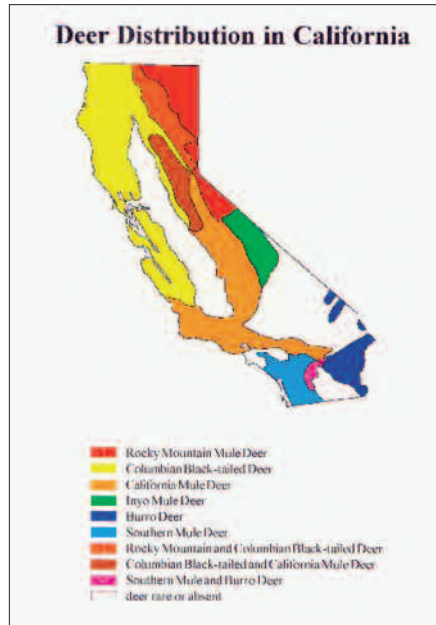


Figure 2. Subspecies of deer in California (California Department of Fish and Game).

that result in germination of annual grasses and forbs (Longhurst et al. 1952), and before acorn mast becomes available. Does have their greatest nutritional demand at this time (e.g., Hanley 1984) because they are nursing fawns. Nutritional quality is diminished because most herbaceous forage has cured and dried by early summer and crude protein content of major browse species begins to decline about mid-summer (Taber and Dasmann 1958). Abundant acorn production and early fall rains that stimulate annual plant growth are important for deer to gain some reserves for the breeding season and coming winter. Throughout the ecoregion, drought can have a profound effect on reproductive success and fawn recruitment.

Woodland chaparral habitats were historically maintained by frequent, low-intensity fires that produced a mosaic of burned and unburned areas (Biswell 1989). Deer populations respond favorably to fire in chaparral with increased body weights and reproductive success (Taber and Dasmann 1957). With advances in fire suppression, the natural fire regime has been altered from what occurred historically. The net result has been maturation of chaparral to a decadent level that provides poor quality forage and declining condition for deer populations. Ownership of the deer range in the California Woodland Chaparral is a mixture of public and private, with a significant amount of land privately owned. Consequently, development pressure is a significant factor that directly results in habitat loss. In remaining wildlands, there is public pressure for fire suppression, as well as pressures for use as livestock range, recreational use, and other human-influenced activities. In contrast, the Arizona chaparral is primarily public land. Terrain is steep and extremely rugged, which limits accessibility (Swank 1958).

In Mexico, wildlife conservation and habitat protection have been hampered by an unstable government infrastructure, lack of funding and ineffective law enforcement (Heffelfinger, 2006). Climate and habitat effects on deer populations are overshadowed by these factors. There is no public land, so large private ranches that protect deer against illegal harvest and manage them as a renewable resource often provide the best deer management (Heffelfinger, 2006). Leopold (1959) observed that subsistence hunting was depressing deer populations in many areas of Mexico. This situation may still exist and could limit distribution of mule deer on the southern periphery of their range.

MAJOR IMPACTS TO MULE DEER HABITAT IN THE CALIFORNIA WOODLAND CHAPARRAL

Loss of usable habitat due to human encroachment and associated activities. Mule deer habitat is completely lost or fragmented due to expansion of urban/suburban areas and other associated activities such as road building, vineyard establishment and motorized recreation. Related human activity can also displace mule deer from otherwise suitable habitat.

Nutritional quality has decreased. Increasing age of woody shrubs can result in forage of lower nutritional quality and plants growing out of reach of mule deer. Many browse plants eventually become senescent and die if not disturbed. Disturbance is needed to revitalize decadent shrubs and increase nutritious new growth.

Vegetation structure has been modified. Both increases and decreases in woody species can decrease mule deer habitat quality. Increasing woody canopy cover in some cases decreases the amount and diversity of herbaceous species. Conversely, decreases in some woody species often results in less forage or hiding and thermal cover.

Plant species composition has been modified.

In some cases noxious or invasive species have proliferated in native plant communities, frequently reducing species richness by replacing native flora in near-monocultures. More subtly, some less desirable species have become more abundant at the expense of more desirable species (e.g., manzanita [*Arctostaphylos* spp.] replacing more palatable wedgeleaf ceanothus [*Ceanothus cuneatus*]).



(Photo by CDFG)

LONG-TERM FIRE SUPPRESSION

BACKGROUND

Fire is the dominant ecological influence on most plant associations within the California Woodland Chaparral Ecoregion. The Mediterranean climate, characterized by mild wet winters and hot, dry summers, generates periods of persistent drought and recurrent fires. Plants in this ecoregion have evolved traits such as fire-induced germination of dormant seeds, re-sprouting from burned stumps, and physiological adaptations that increase flammability (Conrad et al. 1986). Fire can also influence plant re-growth by reducing competition, releasing nutrients and minerals to the soil, and scarifying seeds (Keeley and Keeley 1984, Hanes 1988).

The importance of fire in shaping and maintaining shrub-dominated landscapes is well documented (Hanes 1971, Christensen and Muller 1975, Odion and Davis 2000, Brown 2000, Montenegro et al. 2004). However, the historical record of fires in the California Woodland Chaparral Ecoregion is difficult to interpret. In tree-dominated communities, past fire events can be reconstructed by examining fire scars in growth rings of trees. However, fires in the chaparral system remove above-ground vegetation, leaving no physical record of the event. Therefore, traditional views regarding the role of fire in chaparral ecosystems have been the subject of recent debate (Keeley and Fotheringham 2001).

Before the arrival of Anglo-Americans during the 18th century, Native Americans used fire to manage wildlands, although it is not clear how long or to what extent this was put into practice (Conrad et al. 1986). European settlement brought about significant landscape level changes including alteration and/or removal of natural processes such as fire. As the human population increased, protection of property and life became a high priority for land management agencies, and fire suppression continued throughout much of the 20th century. The condition of present day woodland-chaparral landscape can be attributed to fire suppression and land-use changes brought about by Anglo-American settlement. In some areas, fire suppression has promoted development of mature vegetation, which may not provide optimal ecological conditions for browsing herbivores like mule deer (Gruell 1986). Recent studies suggest climate (primarily “Santa Ana” or foehn winds, and prolonged drought) has pre-disposed this region to periodic catastrophic fires, regardless of vegetation age-class or use of prescribed fire to reduce fuel loads (Keeley 1992, Keeley et



Figure 3. Mule deer can utilize newly-burned areas that are in close proximity to cover (Photo by Rebecca Barboza/CDFG).

al. 1999, Keeley 2002). Furthermore, an increase of accidental and intentional human ignitions has resulted in as many or more burns along the urban-wildland interface than occurred prior to onset of active fire suppression (Keeley and Fotheringham 2001). Even so, use of well-planned prescribed fire and/or mechanical treatment in chaparral to create early-successional, high-quality browse in close proximity to cover can provide substantial benefits to deer (Figure 3).

ISSUES AND CONCERNS

Following a disturbance (such as fire), vegetation undergoes various successional stages. The pattern and rate of change in plant communities are controlled by the physical environment, which has substantial implications for mule deer populations. As habitat shifts from young, open stands to more mature, closed stands, wildlife species and habitat use will change accordingly (Ashcraft and Thornton 1985).

Fire has various effects on plant communities, even within similar habitat types. However, specific patterns of fire can be classified into fire “regimes” which take into account burn intensity, severity, seasonality, frequency, and size. Fire intensity refers to the amount of heat energy produced by a fire, while severity measures potential post-fire changes in plant communities. Fire frequency is an indication of the average number of years between fires on a given landscape (Brown 2000).

Fire regime, plant size, soil type, elevation, and slope influence plant regeneration. Plants that produce seedlings from dormant seeds (obligate-seeders) are generally more common on xeric (dry) slopes and along ridges (Figure 4). Facultative sprouting plants (which re-sprout from stumps, root-crowns, or other specialized underground structures) are more common on mesic slopes that have moist deep soil (Figure 5; Keeley and Keeley 1984).

In general, facultative sprouting species such as chamise and live oaks respond favorably to low-intensity fires (Figure 6), and unfavorably to high-intensity fires. Low-intensity fires result in higher seed bank survival among obligate-seeding species such as *Ceanothus* spp. However, because



Figure 4. Species that sprout from dormant seeds such as wedgeleaf ceanothus are common on dry slopes and ridges. (Photo by Hazel Gordon, U.S. Forest Service Remote Sensing Lab).



Figure 5. Scrub oak re-sprouts from stumps after fires and cutting, and is commonly found on lower-elevation slopes with moist soil. (Photo by Hazel Gordon, U.S. Forest Service Remote Sensing Lab).

these plants regenerate from dormant seeds, they can survive high-intensity fires as well (Keeley and Keeley 1984).

Fire is most beneficial when it occurs as part of the “natural” fire regime. But in woodland chaparral communities, ignitions (either natural or human-caused) frequently occur before or after the autumn “fire season.” In addition, prescribed burns are sometimes applied during spring and winter because of concerns regarding fire control and air quality, or to manipulate burn intensity.

Effects of out-of-season burning can be hard to predict, and vary from species to species (Miller 2001). Fires ignited under inappropriate prescriptions, or during inappropriate seasons or conditions may negatively influence seed germination, reduce post-fire sprouting, and may contribute to type-conversion and loss of species diversity. Furthermore, frequent occurrence of fire can damage young or re-sprouting obligate-seedling shrubs before they become reproductively mature, depleting the seed bank (Brown 2000).

Sustained fire suppression (as well as diminished habitat manipulations and other enhancements) can contribute to the following conditions:

- Reduction or loss of herbaceous plants as canopy cover increases.
- Decreased reproduction and abundance of plant species important for mule deer as the canopy structure changes.
- Increased plant susceptibility to disease and insect infestation as woody plants become decadent.
- Reduction or elimination of disturbances that cycle nutrients and maintain early and mid-successional habitats.
- Increased age, leading to decreased palatability, nutritional quality and availability of important browse species for mule deer.
- Monotypic communities of similar age and structure resulting in a lack of abundant and diverse high quality forage.
- Dense stands of vegetation reduce access to areas of higher quality forage.

The landscape-level deterioration of habitat is a key factor responsible for diminishing mule deer populations in many areas of the Southwest. Reintroducing ecologically appropriate fire regimes holds the most potential for sustaining and creating mule deer habitat in this ecoregion.



Figure 6. Facultative sprouting species such as chamise can proliferate following low-intensity fires. Newly-sprouted chamise provides forage for mule deer, while mature plants are often utilized for cover. (Photo by Robert Vincik/CDFG).



Figure 7. Appropriate cover and mosaic burn pattern to rejuvenate browse following a prescribed burn in a mixed-chaparral community in southern California (Photo by Kim McKee/CDFG).

FIRE MANAGEMENT GUIDELINES

Most woodland chaparral vegetation communities are adapted to, and are reliant on, periodic fire for regeneration, but some are not. Chaparral, coastal sage scrub, and grassland communities respond favorably to fire while oak woodland and riparian communities generally do not. Managers need to strive to restore appropriate fire regimes in fire-adapted shrub communities to maintain health and productivity. In these communities fire is the primary mechanism acting to improve accessibility, palatability, and nutritional value of forage species used by mule deer

(Dasmann and Dasmann 1963, Hobbs and Spowart 1984). Prescribed fire, either management ignited or naturally ignited, that is designed or managed to mimic natural fire regimes can serve as an efficient and cost-effective tool for enhancing mule deer habitat (Table 1). Changes in vegetation composition and structure after a fire influence how mule deer populations respond to post-fire landscapes (Figure 7).

Because fire does not act in isolation of other environmental factors, managers should integrate prescribed fire into overall habitat management planning to ensure that short-and long-term objectives are achieved. Further, considerations of size, timing, frequency, and intensity of fires are critical for achieving site-specific burn objectives (Table 2). Key to successful use of prescribed fire is the development of a plan that integrates both scientific (weather, topography, vegetation, and fire regime) and social (economics and air quality) considerations (Keeley 2001).

A. Fire Management Plan

The first step to a successful prescribed burn is thorough planning. Preparation of a burn plan should be undertaken by an interdisciplinary team of resource specialists having extensive knowledge of plant and fire ecology, fire behavior, fire suppression, post-fire monitoring, and wildlife habitat management. Fire management plans are required for all federal and state land management agency lands and prescribed burns conducted by most local fire management agencies and departments. Detailed federal and state agency burn planning and implementation guidelines exist which identify elements such as burn objectives, safety measures, ignition procedures, control and escape contingencies, and air compliance. Elements that should be incorporated into any plan include:

1. Burn area description (topography, vegetation, and structures)
2. Management objectives, including total acreage to be burned, and desired burn pattern
3. Preparations (site, personnel, and equipment)
4. Desired prescription (weather conditions and timing)
5. Special considerations (endangered species, erosion potential, impacts on riparian and other sensitive habitats, and other potential adverse impacts)
6. Implementation (ignition, suppression measures, and smoke management)
7. Notification procedures (regulatory agencies, local fire departments, law enforcement, and adjoining landowners)
8. Post-burn management activities (remediate erosion and invasion of non-native weedy species)
9. Project evaluation and monitoring strategies

B. Effects of Fire on Critical Habitat Components

1. Food: Fire in woodland chaparral is closely linked to quantity, quality, and diversity of food plants necessary

Table 1. Potential beneficial effects of fire on shrub-dominated communities and habitat requirements (food, cover, and water) of mule deer in fire-adapted plant communities (Dasmann and Dasmann 1963, Hobbs and Spowart 1984, Severson and Medina 1983).

Food	<ul style="list-style-type: none"> ➤ Improves nutrient cycling ➤ Increases palatability of forages ➤ Stimulates crown or root sprouting ➤ Creates a mosaic of different successional stages ➤ Elimination of undesirable plant species ➤ Reduces undecomposed organic materials and litter that inhibit growth of grasses and forbs 	<ul style="list-style-type: none"> ➤ Increases nutrient value of plant species ➤ Removes dense, rank, or over mature growth ➤ Provides for early successional species and communities ➤ Encourages early spring green-up of grasses and forbes ➤ Stimulates seed germination
Cover	<ul style="list-style-type: none"> ➤ Creates/maintains appropriate cover levels ➤ Creates edge ➤ Provides control of young invasive undesirable woody plants ➤ Improves fawning cover through the promotion of seed germination and growth of perennial bunchgrasses (fawning cover) 	<ul style="list-style-type: none"> ➤ Produces temporary openings ➤ Modifies use patterns by deer ➤ Improves detection of predators
Water	<ul style="list-style-type: none"> ➤ Improves water yield ➤ Improves water infiltration, retention, and deep percolation (through increased ground cover) 	<ul style="list-style-type: none"> ➤ Increases spring recharge

Table 2. Effects of fire and season on chaparral vegetation (Severson and Medina 1983, Gordon and White 1994).

TIMING	EFFECTS OF FIRE	
	FORBS	WOODY PLANTS
Cool Season <small>(early-mid winter)</small>	<ul style="list-style-type: none"> ➤ Improved germination ➤ Improved growth and vigor of desirable grasses and forbs ➤ Promotes cool-season annuals and perennials ➤ Maximum forb growth 	<ul style="list-style-type: none"> ➤ Temporary suppression ➤ Reinvigoration of desirable browse
Cool Season <small>(late winter)</small>	<ul style="list-style-type: none"> ➤ Reduces abundance of annual forbs ➤ Promotes perennial grasses ➤ Improved grass quality and species composition 	<ul style="list-style-type: none"> ➤ Temporary suppression ➤ Reinvigoration of desirable browse
Warm Season	<ul style="list-style-type: none"> ➤ Reduces abundance of annual forbs ➤ Promotes perennial grasses ➤ Improved grass quality and species composition 	<ul style="list-style-type: none"> ➤ Maximum mortality

for successful reproduction and survival of deer populations. In mature or late seral stage chaparral communities, browse quality, quantity, availability, and diversity are primary limiting factors during much of the year (Figure 8; Ashcraft and Thornton 1985). A diverse mix of woody plants, forbs, and grasses in an early to intermediate seral stage provide deer with highly nutritious and palatable forage. According to Bowyer (1981) and Dasmann and Dasmann (1963), deer thrive on early successional vegetation that is prevalent from 1-10 years after a fire. Availability of diverse, high quality forage provides deer the opportunity to obtain year-round dietary requirements of protein, carbohydrates, crude fat, vitamins, and minerals. Fire can be an effective tool for returning early successional stages to woodland chaparral communities that are fire adapted (Figures 9-10). In some communities however, frequent fire can damage or

eliminate herbaceous food plants, leading to short-and long-term reductions in forage (Hobbs and Spowart 1984). Generally, prescribed burn intervals of 10-20 years are appropriate in chaparral dominated communities.

2. Cover: The importance of woody plants in providing thermal, hiding, and escape cover in chaparral communities is well described by Ashcraft and Thornton (1985). Habitat quality for mule deer is influenced as much by availability of cover and its proximity to forage, as by presence of diverse and nutritious food plants. In woodland chaparral communities, cover is generally provided by tall woody plants, which in some seasons may also be an important food source. In shrub communities dominated by woody plants, lack of disturbance over time results in a shift to late seral stage vegetation that is dense and unsuitable for mule deer.



Figure 8. Mature chaparral, like this area of the Palomar Mountains in southern California fails to provide important habitat requirements of mule deer, such as quality forage (Photo by Randy Botta/CDFG).



Figure 9. Fire can be used to regenerate desirable woody species and increase availability of forbs in chaparral dominated landscapes to benefit mule deer (Photo by Randy Botta/CDFG).



Figure 10. In shrub-dominated landscapes fire is the most efficient tool available to managers for enhancing mule deer habitat over large areas (Photo by Randy Botta/CDFG).

Managing woody plants with fire can provide benefits to deer through enhancement of forage and cover condition. However, in some situations, use of fire can be detrimental to mule deer. This can occur when fire results in loss of a substantial number of oak trees, eliminating availability of mast and cover structure (Fig. 11; Nichols and Menke 1984), or where fires become so large that thermal cover and hiding cover are eliminated (Ashcraft 1979) over large areas. According to Ashcraft and Thornton (1985), optimum mule deer habitat in chaparral is composed of approximately 40% cover (20% hiding cover, 10% thermal cover, and 10% escape cover) and 60% feeding area. Leckenby et al. (1982) reported that optimum mule deer habitat in shrub-steppe vegetation for Oregon was composed of 45% cover (20% hiding cover, 10% thermal cover, 10% fawn-rearing cover, and 5% fawn habitat) and 55% feeding area. Managers should consider a goal of providing < 40% canopy cover as a general rule of thumb.

C. Additional Tools to Consider

Two other options for enhancing mule deer habitat in woodland chaparral vegetation are mechanical and chemical treatments. These methods may be useful options for plant communities that are not fire-adapted, or in areas where prescribed fire is not feasible. In some situations, mechanical treatments such as hand cutting or crushing, or chemical treatment, can be used to prepare areas for prescribed burning. As with prescribed burning, proper planning and execution of these treatments is critical for achieving success. Each method has advantages and disadvantages and should be considered in relation to management objectives (Table 3). Success in meeting management objectives rests in treatment selection and location, based on knowledge of available treatment alternatives and their effects on plant species and the treatment site (Ashcraft and Thornton 1985).

1. Mechanical treatments vary from hand cutting and grubbing to the use of heavy equipment for clearing, disking, mashing, mowing, or mulching woody vegetation (Fig. 12; Ashcraft and Thornton 1985). According to Richardson (1999) mechanical treatments can be classified into two categories, those designed to remove the above ground portions of plants (shredding and roller chopping) and those designed to remove the entire plant (root-plowing, grubbing, chaining, crushing, and ripping). Above ground removal treatments generally provide increased browse availability by reducing plant height while also increasing browse palatability by stimulating sprouting of tender, highly nutritious regrowth. However, given the rapid regrowth potential of most brush species, benefits to deer derived from improved forage conditions may be temporary. Longer-term benefits from top removal can be achieved by designing brush management programs that treat a



Figure 11. Significant oak tree mortality in woodland communities due to fire results in the reduction of forage and cover for mule deer, as in this interior live oak woodland in the Volcan Mountains of southern California (Photo by Randy Botta/CDFG).



Figure 12. Mechanical treatments can be used to reduce undesirable woody species or rejuvenate desirable forage species, as depicted here following mastication of chaparral on the San Bernardino National Forest in southern California (Photo by Rebecca Barboza/CDFG).

portion of an area over multiple years providing a continuing supply of nutritious, readily available browse. Total removal treatments may be used to eliminate dense stands of woody vegetation to favor increased forb production or to allow for the planting of herbaceous species. However, under proper conditions total removal methods can be effective in knocking down, uprooting, and thinning dense stands of woody vegetation, increasing browse availability and forage values while also increasing forb production through soil disturbance and reduced shrub density. Mechanical treatments are among the most selective tools available for rejuvenating mature chaparral stands to a mix of woody plants, forbs, and grasses favored by mule deer, but may also have higher costs than chemical or prescribed fire treatments.

2. Chemical treatment involves the use of herbicides to control undesirable plants or stands of vegetation. Advantages of chemical control are that complete kill of selected plants can be obtained more easily than with mechanical methods, and that fewer equipment-related hazards may exist. Disadvantages, however, are that much trial and error may be needed to determine proper chemical, application rate, and timing of application for individual plant species in the treatment area. In addition, after treatment the entire woody portion of the plant remains erect and intact requiring mechanical treatment, fire, or a combination of both for complete removal. Application of herbicides in pellet form may be made directly to the soil, or more commonly applied as a liquid directly to the plant surface. Treatment of large areas generally requires broadcast application by aerial or ground spraying, using aircraft or vehicle boom sprayers. Criteria for the selection, use, and application of herbicides are well described by Ashcraft and Thornton (1985) and Richardson (1999). Method and rate of application must be carefully selected to maximize success, and to minimize adverse impacts to mule deer, other wildlife, and non-target plant species.

Table 3. Advantages and disadvantages of mechanical and chemical treatments (Ashcraft and Thornton 1985, Richardson et al. 2001).

TREATMENT	ADVANTAGES	DISADVANTAGES
Mechanical	<ul style="list-style-type: none"> ➤ Selective ➤ Produces immediate forb response ➤ Promotes a variety of herbaceous plants through soil disturbances and decreased competition ➤ Encourages sprouting of palatable and nutritional browse plants 	<ul style="list-style-type: none"> ➤ Cost ➤ High erosion potential ➤ Limited by topography ➤ Archaeological concerns ➤ Most methods only provide temporary control of woody plants
Chemical	<ul style="list-style-type: none"> ➤ Provides for treatment of large areas in a short time period (aerial) ➤ Erosion potential is less (no ground disturbances) ➤ Not limited by topography (aerial) ➤ Selective (individual plant treatment) ➤ Useful as a preparatory treatment before prescribed burning 	<ul style="list-style-type: none"> ➤ Cost ➤ Some woody plants are resistant to herbicides ➤ Short-term suppression of desirable plants (1-2 years after treatment) ➤ Non-selective (non-target damage or mortality to desirable plants) ➤ Woody plants and litter not totally consumed (standing dead woody plants)

HUMAN ENCROACHMENT

BACKGROUND

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

ISSUES AND CONCERNS

Displacement by Occupation

Wildlife habitat is attractive in many ways to humans. Because of the appealing nature of landscapes occupied by wildlife, and simply availability of unoccupied space, humans are increasingly moving to these areas to live. Occupation of deer habitat brings with it construction of homes, fencing, roadways, agriculture, business developments, public buildings, schools and other supporting infrastructure (Figure 13). People who occupy these areas frequently bring domestic dogs and livestock that may jeopardize wildlife through direct mortality, habitat degradation, or disease transmission. These communities are sometimes located in habitats that fill critical wildlife needs during periods of migration or seasonal stress. During the mid 1990s alone, this development occupied 5,436,200 acres (2.2 million hectares) of open space in the West (Lutz et al. 2003).

In California and Arizona, vast amounts of deer habitat are vanishing due to urban sprawl, residential development and agriculture (Wallmo 1978). Oak woodland habitats of California are especially vulnerable, because they are often privately owned and located in areas desirable for development. During the 1970s and 1980s the growth of large cities such as San Jose and San Diego and the many associated suburbs expanded to eliminate about 7,400 acres of oaks per year (Pavlik et al. 1991). Urban growth into neighboring open space continues to degrade California's oak woodlands (Hagen 1997) and cause losses to deer habitat while creating a severe wildfire hazard (Kucera and Mayer 1999). This type of development is the greatest impact of human disturbance on wildlife populations.

Along with negative impacts, human occupation may provide some advantages to local wildlife populations (Tucker et al. 2004). Wildlife in some urban areas may have more water from artificial sites (e.g., ponds) and enhanced forage (e.g., lawns, landscaping, golf courses, agricultural fields) than in surrounding areas (Figure 14). Reduced populations of predators in these habitats may also decrease mortality for wildlife that inhabit the area, however predator-prey relationships often continue even after people move into the vicinity. Enhanced forage conditions and



Figure 13. Human development is spreading in Santa Clara County, near Gilroy, California (Photo by Martha Schauss/CDFG).



Figure 14. Residential development can provide additional resources for local deer (Photo courtesy of CDFG).

decreased predation may result in unhealthy densities of wildlife that will be susceptible to diseases. Predators may move into urban areas from surrounding areas to prey on naïve wildlife. Ultimately, these predators may supplement their natural diet with domestic pets. So while there are some benefits to deer in these situations, resulting problems often outweigh advantages.



Figure 15. Vineyard developments can create an increase in mule deer food and cover but are often surrounded by deer-proof fencing (Photo by Robert Vincik/CDFG).



Figure 16. Hillside cleared of oak trees in the Central Coast range of California (Photo by Henry Coletto).

Some agricultural developments also make habitats more desirable to mule deer. However, these same developments often include efforts by those managing agricultural lands to limit wildlife use of the area. Maintenance of a mature vineyard may provide forage and cover, but often deer-proof fences are constructed that prevent entry (Figure 15). In this case, the habitat is lost to deer use. Where deer-proof fences are not constructed, landowners sometimes kill high numbers of deer under depredation permits. Recently, progress has been made in promoting biodiversity in California vineyard development by increasing use of cover crops and hedgerows (Hilty and Merenlender 2002), but this practice is not widespread. Establishing buffers of natural vegetation along riparian corridors and leaving those areas unfenced may also be a beneficial practice.

A concern for mule deer is encroachment upon and development within important habitats. For resident deer in this ecoregion, development will usually have a year-round

negative effect. Migratory deer experience similar impacts to critical portions of their range habitats, although they may only experience these impacts for part of each year. Improved forage and a potential for decreased predation notwithstanding, increased housing density can result in decreased mule deer abundance (Vogel 1989). Road development can limit access to important areas as well, and may be a locally important mortality factor.

Reduction of Habitat Suitability

Human activity has the ability to alter habitat suitability through direct alteration of habitat characteristics, thereby influencing habitat quality. Unregulated use of off-highway vehicles (OHVs) can alter habitat characteristics through destruction of vegetation, soil compaction, and increased erosion. Perry and Overly (1977) found roads through meadow habitats reduced deer use, whereas roads through forested habitat had less effect. Excessive livestock grazing, or grazing at the wrong time of year, may also alter habitat suitability by removing forage and cover species that deer rely on.

Removal of oaks for agricultural clearing and fuelwood is recognized as a threat to quality and quantity of habitat for deer (Figure 16; Kucera and Mayer 1999). In California Woodland Chaparral deer rely heavily on oak leaves in summer months, and oak mast to augment poor nutritional conditions that typically exist in fall, before rains begin to stimulate new plant growth (Urness 1981, Schauss and Coletto 1986, unpublished California Department of Fish and Game report, Kucera and Mayer 1999). According to Pavlik et al. (1991), studies have demonstrated a correlation between acorn availability and reproductive success in deer. Approximately 14,000 acres of oak woodland are lost to development each year, while a comparable amount is impacted by woodcutting (Hagen 1997). Agricultural clearing in the 20th century was recognized as the single most consumptive use of oak woodlands and riparian forests in California (Pavlik et al. 1991).

Various additional activities such as energy developments, landfills, and mining tend to be small in scale in this ecoregion, and individually have little effect on mule deer in their vicinity. However, when considered cumulatively, the habitat lost to these activities may represent a significant amount of deer range. Reservoir construction and hydroelectric development can inundate large portions of deer range, and while these projects were common in the past, new construction is rare. Oil, gas, and wind energy developments have potential to limit deer use by new road installation and vegetation conversion. Aggregate mining (Figure 17; gravel quarries) can also cause deer and other wildlife to discontinue use of the immediate area, although if properly rehabilitated, can often be returned to usable wildlife habitat. The most obvious negative impact on

habitat suitability is the elimination of linkages between important habitats.

Displacement through Disturbance

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

The human population in California in the year 2000 was nearly 34 million people. Even in National forests and parks California's vast population may be impacting deer behavior beyond what occurs elsewhere. The Cleveland National Forest in Southern California reported nearly 800,000 forest visitors during the period of October 2000 – September 2001 (Kocis et al. 2002). Nearby is the Cuyamaca Rancho State Park, where only 26,000 acres is subject to more than half a million visitors annually (Sweanor et al. 2004, Wright 2001). Most of this type of recreational use occurs in critical summer months, during fawning and lactation periods. In describing the Central Coast bioregion, Kucera and Mayer (1999) indicated negative effects resulted from human recreational activities, particularly in riparian habitats. Recreational use continues to grow as California's populace expands.

Information regarding response 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. Proximity to roads and trails has a greater correlation with deer distribution than does crude calculations of mean road densities (Johnson et al. 2000). Off road recreation is increasing rapidly on public lands. The U. S. Forest Service estimates that OHV use has increased 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.

Highways

Highways are a unique situation in which all three types of displacement may occur. Displacement through occupation occurs when habitat is eliminated by actual construction of roads and highways. Habitat suitability is reduced in areas related to roads, shoulders, rest stops, road cuts, etc.

Alteration of the area surrounding roadways may include clearing of native trees and other vegetation which is replaced by less desirable species, or in some cases, no vegetation. If palatable species are planted close to roadways, deer mortality may increase. Fences and streetlights are further modifications that may affect deer. And finally, displacement by disturbance occurs by traffic noise and increase in human presence (Figure 18).

Recognition and understanding of the impact of highways on wildlife populations has increased dramatically in the past decade (Forman et al. 2003). In fact, highway-associated impact has 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). These impacts are especially severe in western states where rapid human population growth and development are occurring at a time when deer populations are depressed. Human population growth has resulted in increased traffic



Figure 17. Aggregate mine impacts oak woodland and riparian habitats (Photo by Mary Sommer/CDFG).



Figure 18. Highways, with their associated roads and fences, effectively inhibit deer movement in many areas (Photo by Mary Sommer/CDFG).

volume on highways, upgrading of existing highways, and construction of new highways, all serving to further exacerbate highway impact to mule deer and other wildlife.

Direct loss of deer and other wildlife due to collisions with motor vehicles is a substantial source of mortality affecting populations. Romin and Bissonette (1996) conservatively estimated > 500,000 deer of all species are killed each year in the U.S. Schwabe and Schuhmann (2002) estimated this loss at 700,000 deer/year, whereas Conover et al. (1995) estimated > 1.5 million deer-vehicle collisions occur annually. These losses result in substantially less recreational opportunity and revenue associated with deer hunting. Additionally, roadways fragment habitat and impede movements for migratory herds (Lutz et al. 2003). Some highway transportation departments have used overpasses and underpasses for wildlife to mitigate highways as impediments, and to reduce collisions between deer and vehicles. Recently, temporary warning signs have been demonstrated to be effective in reducing collisions during short duration migration events (Sullivan et al. 2004).

Of all the impacts associated with highways, the most important to mule deer and other wildlife species is attributable to barrier and fragmentation effects (Noss and Cooperrider 1994, Forman and Alexander 1998, Forman 2000, Forman et al. 2003). Highways alone act as barriers to animals moving freely between seasonal ranges and to special or vital habitat areas. Use of median walls exacerbates this effect. Safety, economics, and convenience of vehicular travel often take priority over wildlife considerations (Reed 1981a). This barrier effect fragments habitats and populations, reduces genetic interchange among populations or herds, and limits dispersal of young; all serve to ultimately disrupt the processes that maintain viable mule deer herds and populations. Furthermore, effects of long-term fragmentation and isolation render populations more vulnerable to influences of random events like weather and disease, and may lead to extinctions of localized or restricted populations of mule deer. Other human activity impacts directly tied to increased travelways include increased poaching of mule deer, unregulated off-highway travel, and ignition of wildfires. Highways also serve as corridors for dispersal of nonnative invasive plants that degrade habitats (Gelbard and Belnap 2003).

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

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

Structures designed to promote wildlife passage across highways are increasingly being implemented throughout North America, especially large crossings (e.g., underpasses or overpasses) designed specifically for ungulate and large predator passage (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 wildlife needs (Farrell et al. 2002). However, there is increasing expectation that such structures will indeed yield benefit to multiple species and enhance connectivity (Clevenger and Waltho 2000), and that scientifically sound monitoring and evaluation of wildlife response will occur to improve future passage structure effectiveness (Clevenger and Waltho 2003, Hardy et al. 2003).

GUIDELINES

A. Transportation and Community Planning and Coordination

1. Maintain interagency coordination in land planning activities to protect critical habitats.
2. Become involved in master planning of developments and communities.
3. Identify and prioritize critical habitats, seasonal use areas,

- migration routes, and important populations of mule deer.
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 collisions with deer as new roads are developed or altered for higher speed and greater volume traffic.
 6. Incorporate “deer-friendly” designs into median barriers and other highway features to reduce impacts.
 7. Include consideration of deer and other non-listed wildlife species in community general plans.
 8. Include requirements for clustered housing to minimize impacts of development on open space.
 9. Identify lands with high quality deer habitat for protection by purchase, conservation easement, or inclusion in mitigation bank, to be owned or managed by government agency or conservancy organization.
 10. Encourage counties to pass ordinances for better protection of oak woodlands by more stringent regulation of woodcutting, etc.

B. Minimizing Negative Effects of Human Encroachment

1. Develop consistent regulations for off-highway vehicle (OHV) use, and designate areas where vehicles may be legally operated off road. Maintain interagency coordination in enforcement of OHV regulations.
2. Recommend conservation of oak resources.
3. Examine records of vehicle-killed deer to determine where major collision areas exist and evaluate need for wildlife passage structures.
4. Construct over and underpasses for wildlife corridors.
5. Monitor activities that may unduly stress deer at important times of year (e.g., recreational activity or harassment by dogs during fawning). Reduce/regulate disturbance if deemed detrimental.
6. Enhance alternate habitats to mitigate for habitat loss, including components like water availability.
7. Provide ungulate-proof fencing to direct wildlife to right-of-way passage structures or away from areas of high deer-vehicle collisions.
8. Encourage use of fencing that allows for safe passage of wildlife in appropriate areas to minimize habitat fragmentation.
9. Educate rural homeowners and recreational users of undeveloped areas about negative impacts of dogs on deer and other wildlife. Encourage parks departments and other land management agencies to enforce leash laws and dog restrictions.

C. Wildlife Passage Structures

1. Work with State and Federal transportation agencies and agency engineers early in the planning process to facilitate the planning and funding of passage structures for deer.

2. To maximize use by deer and other wildlife, passage structures should be located away from areas of high human activity and disturbance.
3. 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).
4. Spacing between structures is dependent on local factors (e.g., known deer crossing locations, areas of excessive deer-vehicle collisions, deer densities adjacent to highways, proximity to important habitats).
5. Where appropriate and available, use models and other tools to assist in location of passage structures (Clevenger et al. 2001, Barnum 2003, and Claar et al. 2003).
6. 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). Underpasses designed specifically for mule deer should be at least 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. Reductions in underpass width influence mule deer passage more than height (Clevenger and Waltho 2000, Gordon and Anderson 2003).
7. More natural conditions within underpass (e.g., earthen sides and naturally vegetated) have been found to promote use by ungulates (Dodd et al. 2006). In Banff National Park, Alberta, deer strongly preferred crossing at vegetated overpasses compared to open-span bridged underpasses (Forman et al. 2003).
8. Use ungulate-proof fencing in conjunction with passage structures to reduce deer-vehicle collisions (Clevenger et al. 2001, Farrell et al. 2002). Caution should be exercised not to construct extensive ungulate-proof fencing without sufficient passage structures to avoid creating barriers to free deer movement.
9. Where possible, fences should be tied into existing natural passage barriers (e.g., large cut slopes, canyons; Puglisi et al. 1974).
10. When fencing is not appropriate to reduce deer-vehicle collisions, alternatives include dynamic signage (sometimes with flashing lights) 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).

WILD AND DOMESTIC HERBIVORES

BACKGROUND

An impressive variety of herbivores, including deer of the genus *Odocoileus*, populated much of what is now the California Woodland Chaparral Ecoregion until about 10,000 years ago (Edwards 1992). Mule deer were one of the few large herbivores that survived the Pleistocene (12,000 to 10,000 years ago), and then a period of heat and drought that occurred from 8,000 to 5,000 years ago. Mule deer, tule elk (*Cervus elaphus nannodes*), and pronghorn (*Antilocapra americana*) greeted Spanish explorers that entered California in the 16th Century (Edwards 1992). In 1769, Spanish missionaries established the San Diego Mission, the first European settlement on the California landscape, which began the system of domestic livestock grazing that continues today (Burcham 1957). The Spanish propagated large herds of livestock (primarily cattle, sheep, and horses) for their missions located along the coast. By the 1830's, sheep are thought to have numbered around 300,000, and cattle between 140,000 and 420,000 (Burcham 1957).

Large numbers of livestock (mainly cattle and horses) also accompanied the thousands of immigrants who came to California after gold was discovered in 1848 (Longhurst et

al. 1952). Burcham (1957) estimated that by 1860, 3.5 million livestock occupied California and by 1880 the number increased to approximately 6 million sheep and >1 million cattle (Wagner 1989). As these large numbers of livestock (Figure 19) heavily grazed range all over California, the carrying capacity for both livestock and mule deer decreased rapidly. Grazing, combined with unregulated hunting, severe weather, and conversion of habitat to urban and agricultural uses, resulted in a "drastic decrease in deer numbers during the second half of the nineteenth century" (Longhurst et al. 1952). The beginning of the twentieth century brought improved management of grazing on public lands, a development that contributed significantly to the recovery of the deer herds of California (Longhurst et al. 1952).

European settlement of California also brought the introduction of annual grasses and forbs that, being better adapted to intense grazing, eventually replaced many of the native perennial grasses that originally dominated the understory of the California Woodland Chaparral (Harris 1967, Evans and Young 1972, Heady 1977). Conversion of native perennial grasslands to annual grasslands and forbs has had mixed impacts on deer habitat quality. Longhurst et al. (1976) found that conversion of perennial grasslands to

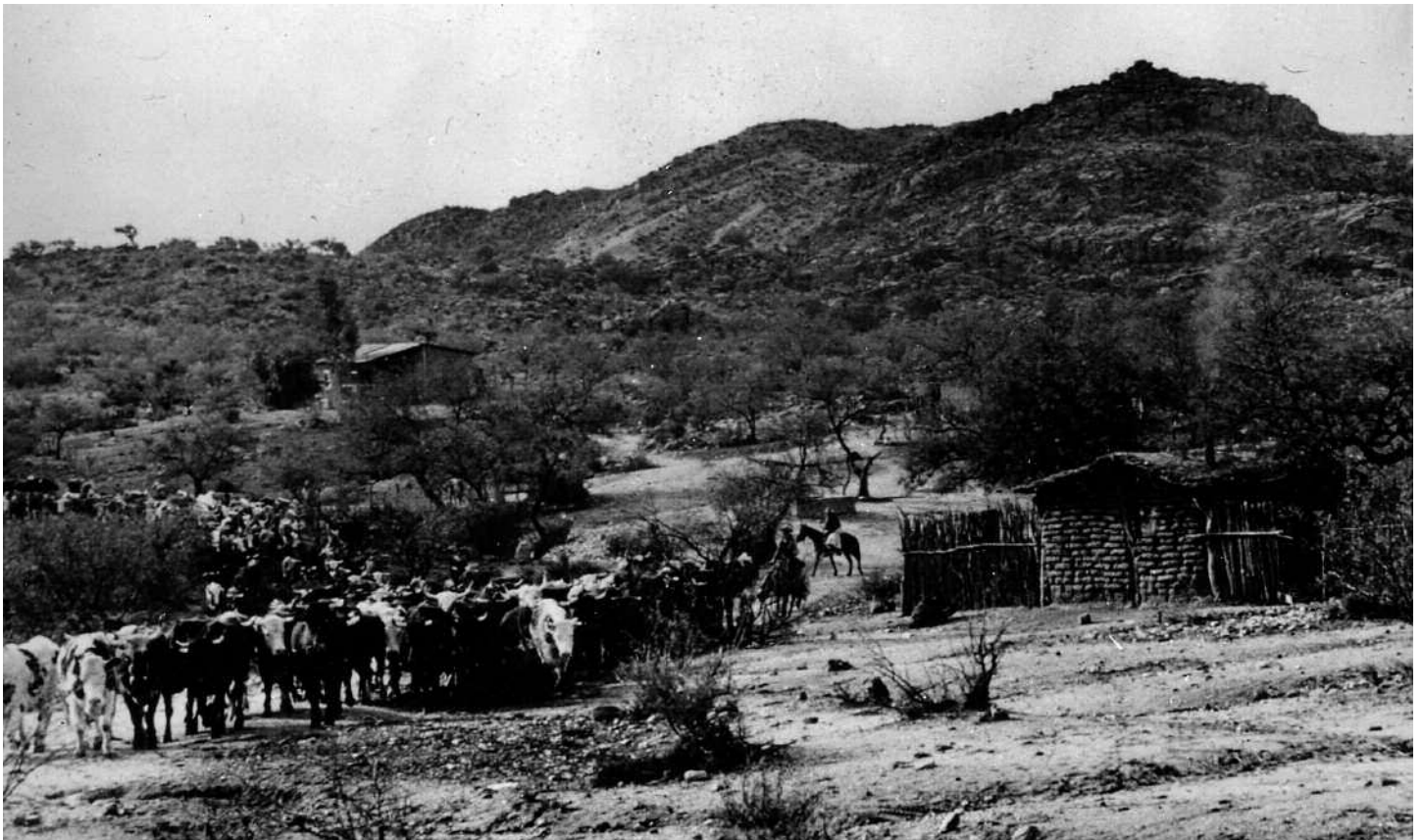


Figure 19. Historical photo of vaqueros (cowboys of Spanish or Mexican descent) gathering cattle in the 1890s in Tulare County, San Joaquin Valley, California (Photo courtesy of Tulare County Public Library, Annie Mitchell History Room).

annual plant species actually may have improved forage conditions and carrying capacity of deer habitat in some areas by introducing non-native forbs such as filaree (*Erodium* spp.) and clover (*Trifolium* spp.). However, perennial forbs that mule deer prefer still grow in this ecoregion, such as California buttercup (*Ranunculus californicus*), brodiaea (*Brodiaea* spp.), popcorn flower (*Plagiobothrys* spp.), lotus (*Lotus* spp.), miner's lettuce (*Claytonia perfoliata*), and fiddleneck (*Amsinkia* spp.) (Bertram 1984). Perennials stay green longer than annuals, therefore they usually provide more long-term nutrition to deer than annuals, especially in locations where deer live year-round and can not migrate to habitats with green summer feed.

ISSUES AND CONCERNS

Livestock Grazing Impacts

Grazing by domestic livestock is the most common land use practice on undeveloped lands in the western United States (Wagner 1978, Crumpacker 1984), with as much as 70% of western lands being grazed in a given year (Council for Agricultural Science and Technology 1974, Longhurst et al. 1983, Crumpacker 1984). California Woodland Chaparral is no exception, with livestock grazing (predominantly cattle) occurring on approximately the same percentage of rangeland within the ecoregion (Huntsinger and Hopkinson 1996, Standiford and Barry 2005). Therefore, depending on location and grazing strategy, mule deer and cattle may share a large majority of the habitat during part of any given year. Approximately 19 million acres of this habitat (more than 35% of private land in California) are owned by livestock ranchers. Many ranchers supplement their private rangeland with federal grazing permits (the majority on National Forests or BLM lands) that often serve as summer range for their cattle. Eleven National Forests and seven BLM Districts have ownership of California Woodland Chaparral habitat: six in the Coast Mountain Range or inland Southern California, five along the west slope of the Sierra Nevada, and seven in a narrow band across central Arizona. The habitat on these private and federal lands functions as winter range for the migratory deer herds that summer in the Sierra Nevada and San Geronio mountains, and as year-round range for the remainder (majority) of deer in the region.

As reported by Bertram and Ashcraft (1983), Bertram (1984), Kie and Loft (1990), Bronson (1992), and Kucera and Mayer (1999), cattle grazing can have significant impacts to deer in oak woodlands and chaparral (Figures 20 and 21) if practiced at a time of year or under conditions that cause:

- excessive competition for browse or forage
- degradation of cover required for sheltering fawns
- degradation of cover to hide from predators and obtain relief from extreme winter or summer weather

- degradation or exhaustion of water sources important to deer
- changes in plant composition that reduce forage or cover for deer
- introduction and spread of non-native invasive plant species.

When managed at an appropriate stocking rate (maintaining species composition and plant vigor), and at a time of year that does not result in any of the above impacts, the effects of livestock grazing on mule deer habitat may be minimized (Longhurst et al. 1976, Kie and Loft 1990, Kie and Boroski 1995). Depending on location, grazing may actually benefit mule deer by reducing dense stands of annual grasses that compete with forbs, as well as stimulating growth of low-profile forbs that deer prefer



Figure 20. The fence line in this photo allows a post-grazing season comparison of the effects of cattle grazing levels in woodland chaparral habitat. The area in the foreground is heavily grazed, while the area in the background has been lightly grazed. There is much more browse and hiding cover for deer behind the fence, even during early fall, the driest part of the year (Photo by Eric Kleinfelter/CDFG).



Figure 21. Heavy grazing in woodland chaparral habitat can produce "high-lined" oak trees that provide little browse within the reach of deer (Photo courtesy of CDFG).

(Longhurst et al. 1976, Kie 1988, Kie and Loft 1990). One important consideration, however, is that timing, stocking rates, and location of grazing practices needed to conserve and/or improve mule deer habitat may not be in the best financial interests of the livestock operator (Longhurst et al. 1976).

As mentioned above, plant communities of this ecoregion are dominated by oak woodland or chaparral, with an annual grass-forb understory, and by annual grassland-forb communities with no overstory component. Due to their unique ecological responses, descriptions of how mule deer and their habitat respond to cattle grazing and the presence of other herbivores are addressed separately in the following section as oak woodland or chaparral components. Annual grassland-forb communities are not addressed separately because they will respond similarly regardless of overstory presence

1. Oak Woodlands with Annual Grass and Forb Understory

Oak leaves and acorns are important components of the mule deer diet, particularly in the summer and fall (Taber and Dasmann 1958, Schauss and Coletto 1986, unpublished California Department of Fish and Game report). Acorns are high in digestible energy (Harlow et. al 1975), and are an important source of nutrition for mule deer, especially in the fall and early winter, when nutritional stress is often most intense. Acorn mast production is particularly important for does that need to improve their body condition for reproductive success in the spring (Longhurst et al. 1979, Bertram and Ashcraft 1981, 1983, Bertram 1984). Acorns can provide significant benefits to both non-migratory and migratory deer. Acorns provide forage to migratory deer returning to winter ranges that often lack significant new grass or forb growth (Dixon 1934, Leach and Hiehle 1957, Bertram and Ashcraft 1983, Bertram 1984). They are especially important to fawns arriving on the winter range from their summer birthplaces in the Sierra Nevada. Fawns are often nutritionally stressed by their first winter migration and recent weaning (Bertram 1984.) Holl (1974, 1975) reported improved physical condition of fawns that consumed large amounts of acorns on the North Kings Deer Herd winter range in Fresno County. Similarly, non-migratory deer benefit from acorn mast to alleviate nutritional deficiencies of the dry summer months. Acorn crops vary from year to year and may be a scarce food item in some fall seasons. Because different species produce "bumper crops" during different years, a diverse oak woodland or forest rarely experiences a year without acorns (Pavlik, et al. 1991). Therefore, managers should strive to maintain diverse oak assemblages.

Long-term viability of oak trees in California has become a topic of great interest in recent years. This subject has implications for cattle management, but even more so for deer, given the importance of oaks to the survival of fawns, and to the health and nutrition of adults. Some researchers reported negative impacts caused by domestic and wild herbivores on the regeneration and recruitment of oaks. Muick and Bartolome (1987), Bolsinger (1988), Standiford et al. (1991), and Swiecki and Bernhardt (1993) reported that excessive populations of domestic and/or wild herbivores contributed to decreased recruitment and survival of oak trees to the sapling size class. Holzman (1993), on the other hand, reported increased canopy density and basal area in blue oak woodlands from 1932 to 1992, with the presence of "typical livestock grazing practices and fire exclusion policies...due to residual tree growth and recruitment of new individuals." Pocket gophers (*Thomomys bottae*) are a significant predator of oak seedlings (Griffin 1980). Other rodent species (Bernhardt and Swiecki 1997, McCreary and Tecklin 1997, Tyler et al. 2002), as well as cattle, mule deer, and insects consume acorns and oak seedlings.

Annual grasses may also play a role in the oak recruitment process by out-competing oak seedlings for water. Griffin (1973), Bernhardt and Swiecki (1991), and Danielson and Halvorson (1991) reported that competition from annual grasses may have contributed to inadequate regeneration of valley oaks in the 20th century. Tyler et al. (2002) speculated that a thick layer of thatch along with new grass growth may have attracted high densities of grasshoppers to a research site in Santa Barbara County that resulted in defoliation of many oak seedlings (Figure 22). A thick



Figure 22. Grasshoppers feeding on leaves of a planted oak seedling on the central California coast. These insects can rapidly defoliate young oak trees (Photo courtesy of Claudia Tyler, University of California, Santa Barbara).

herbaceous cover may also attract more rodents such as voles that damage oak seedlings (Bernhardt and Swiecki 1997). Impacts of livestock use should be included when monitoring the cumulative impacts of animals on oak sapling recruitment, particularly in locations where recruitment appears inadequate.

2. Chaparral with Annual Grass and Forb Understory

Considerations for grazing cattle in chaparral habitats are similar to oak woodlands. Because chaparral grows in dense stands and has tough, thick leaves designed to tolerate hot, dry summers, mature chaparral shrub species are generally quite resilient to grazing, if they are grazed by cattle at all. However, the most important habitat features for chaparral deer (the herbaceous understory, new growth on browse species, riparian areas, and acorns) are vulnerable to impacts by cattle (Kucera and Mayer 1999). Mule deer populations in this habitat will be healthier and will experience better fawn survival if use of these habitat features by cattle is limited. Small enclosures or cages may be used to differentiate between wildlife and livestock use.

Compared to oak woodlands, chaparral often has very limited annual grass and forb components. Unproductive soils, often associated with chaparral habitat, and a thick, decadent overstory of shrubs often create poor conditions for the growth of annuals. Cattle grazing in chaparral should be monitored closely to prevent heavy use of forbs and browse that cattle may eat in the absence of annuals (Bertram 1984).

Swank (1965) reported that most research in Arizona found increased competition between deer and cattle in the fall and winter when green grass is scarce and cattle turn to browse, compared to the summer when cattle concentrate on abundant quantities of green grass. In years when chaparral browse in Arizona may achieve little or no growth during the summer (due to low rainfall or other weather influences), forbs and grasses that grow along washes and on flats may be particularly important to deer.

According to Cronemiller and Bartholomew (1950), Taber and Dasmann (1958), and Ashcraft (1979), late spring through summer is a critical period for deer in chaparral because 1) typically, fawns are born in mid-April through July; and 2) most deer are usually undergoing some form of nutritional deficiency in the late summer as they endure the period when green forage is limited and before the onset of fall or winter rains. Acorns are important to deer in this habitat, particularly in late summer and early fall, when resident deer are struggling to find the nutrition they require to survive until fall precipitation stimulates the growth of new grasses and forbs (Cronemiller and Bartholomew 1950, Ashcraft 1979, Pine and Mansfield 1980). However, compared to oak woodlands, acorn crops in chaparral are

smaller and less reliable (Cronemiller and Bartholomew 1950, Taber and Dasmann 1958, and Ashcraft 1979).

By nature of its dominance on the landscape, and its moisture holding properties, chamise is the most important browse species for deer in this habitat. This shrub is the "fall back" species that deer in this harsh environment can always count on for basic nutrition and moisture when all other plant sources have dried or been consumed (Taber and Dasmann 1958, Ashcraft 1979). Chamise, like most chaparral shrubs, responds vigorously with new growth to disturbance (e.g., fire or mechanical manipulation). To be nutritionally productive and accessible to deer, frequent disturbance in chaparral brushfields is important. Disturbance also mimics natural processes in chaparral that recycle decadent vegetation and promote new growth. Cattle, however, are not a natural component of these processes, and their presence in chaparral brushfields following disturbance should only be allowed when excessive browsing of new growth and trampling of new plants will not occur (Bertram 1984, Bronson 1992).

3. Riparian Areas

Riparian and wetland areas take on special importance when fawns are born. Fawns need cool, shady places to hide and rest, and does require nutritious forage from these areas during lactation. All age classes of deer living in chaparral habitat use riparian and wetland areas to survive the hot, dry summers. Maintaining quality riparian and wetland habitat, by limiting cattle use during late spring, summer, and fall, by frequent pasture rotation, or by completely excluding them from all or portions of these areas, will provide deer with the resources they need to make it through physically-demanding chaparral summers (Figures 23 and 24). Ward et al. (2003) provide photo references to assess the condition of riparian areas.

Impacts from Wild Pigs

Wild pigs (*Sus scrofa*) are not native to California. Most are descended from domestic pigs that were introduced by Spanish settlers in the late 1700s (Pine and Gerdes 1973, Mayer and Brisbin 1991). After being released to graze and browse in oak woodlands, some of the original domestic stock escaped their human hosts and became feral. European wild boar were introduced to Monterey County in the central coast area of California in the 1920s, and have since been relocated to other parts of the state (Hoehne 1994), where they have interbred with the feral pigs. In the mid 1980s Mansfield (1986) estimated California's wild pig population at 70,000 to 80,000. They are a popular game animal, with approximately 4,500 to 8,000 harvested per year from 1993-2004 (CDFG website <http://www.dfg.ca.gov/hunting/pig/takeindex.htm>, accessed June 20, 2007), but their population has continued to increase despite this high level of harvest. Wild pigs are



Figure 23. Heavy, poorly managed livestock grazing in riparian areas may result in habitat that has very little value to mule deer (Photo by Eric Kleinfelter, CDFG).



Figure 24. Light, managed grazing in riparian areas can accommodate livestock use and still result in quality mule deer habitat (Photo by Eric Kleinfelter, CDFG).



Figure 25. Rooting by wild pigs can be very destructive in oak woodland and grassland habitats (Photo by Martha Schauss/CDFG).

intelligent, possess excellent sense of smell and hearing, and are adapted to thrive in California's Mediterranean climate. They are also very prolific, with sows producing a litter of five to six young every year, and sometimes two litters when conditions are exceptionally favorable or when the first litter is lost (Barrett 1978, Schauss 1980). Updike (CDFG personal communication, 2006) estimated the current population of wild pigs in California at 200,000 to 1,000,000.

Wild pigs are omnivorous and forage by rooting through soil with their snouts, a behavior that is often very destructive to the habitats they occupy. Sweitzer and Van Vuren (2002) reported that wild pig rooting significantly reduced above-ground plant biomass in oak-dominated habitats, possibly reducing forage available to deer and other herbivores. Pigs may also reduce germination and survival of oak seedlings, and compete with deer for acorns (Sweitzer and Van Vuren 2002). As the photo in Figure 25 illustrates, wild pigs can effectively imitate a rototiller with their ability to root-up and destroy native vegetation. Individuals who have witnessed this damage in the field can attest to the significant destruction that wild pigs can accomplish in a short amount of time.

Although the spread of wild pigs has stopped in most parts of the United States (Waithman et al. 1999), they continue to expand their range in California. If this trend continues, they may increase their level of competition with deer for resources, particularly in the California Woodland Chaparral Ecoregion. Intensive control programs may help to reduce wild pig populations at local levels, and reduce the negative impact they have on deer habitat. Wildlife managers, ranchers, and the public should consider the potential impact of pigs when assessing range conditions for deer and cattle.

Tule Elk

There are three sub-species of elk that occur in California: Tule elk, Rocky Mountain elk (*Cervus elaphus nelsoni*), and Roosevelt elk (*C. e. roosevelti*). Tule elk likely evolved from Rocky Mountain elk during or after the Pleistocene (McCullough 1969) and are the smallest of the three sub-species. Tule elk are the only sub-species of elk with a significant population in the California Woodland Chaparral Ecoregion (California Department of Fish and Game 2004). Journals and diaries of early explorers indicate that approximately 500,000 tule elk inhabited much of the oak woodland and oak grassland habitat types in the state (McCullough 1969). Market hunting, competition with livestock, conversion of perennial grasslands to annual grasslands,

and the conversion of large amounts of tule elk habitat to agricultural land uses extirpated all but one small population that survived in the southern San Joaquin Valley in the late 1860s (McCullough 1969). Through conservation efforts and relocation of individuals from this last remaining herd, the tule elk population slowly recovered, however, and continues to grow into the twenty-first century in California.

In 2006, between 3,800 and 3,900 tule elk, occurring as 21 separate herds, inhabited California (Hobbs, CDFG personal communication, 2006). Populations in the Coast Mountain Range of California are doing well, but competition between mule deer and elk has not been documented to be a problem in California (California Department of Fish and Game 2004). Nelson and Legee (1982) stated that "It would appear, therefore, that neither the elk nor the mule deer is affected seriously by the other, mainly because of differences in primary forage species and habitat choice." This also appears to be the case in the California Woodland Chaparral Ecoregion. However, research conducted in northeastern Oregon indicates that competition between elk and mule deer may occur when forage is limited (typically late summer and early fall) or during drought conditions (Coe et al. 2005, Findholt et al. 2005).

In recent years, the potential for competition between deer and elk has received considerable attention in the western states and provinces of North America. Many states and provinces have reported a decline in deer population numbers, coinciding with an increase in elk numbers. It has not been proven that elk consistently displace deer or are a significant factor in suppressing their numbers throughout a broad geographic region. In considering the potential for competitive interaction between deer and elk, a variety of factors may be important such as predation, climate, digestive physiology, energetics, vegetation succession, livestock, and human-related factors. Lindzey et al. (1997), and Keegan and Wakeling (2003) discussed these and other factors in reviewing the potential for competition between deer and elk throughout the west, and compiled an extensive list of references regarding this subject. They concluded that it is appropriate to question whether the growth of elk populations has contributed to apparent deer decline, but found no consistent trends in sympatric areas that would suggest an important cause-and-effect relationship.

GUIDELINES

A. Suggested Grazing Plans

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 (www.wildlife.org) Policy Statement regarding livestock grazing on federal rangelands (The Wildlife Society 1998). Grazing practices such as

appropriate stocking rates, and when practical, rotational use of fields and/or allotments will often promote the establishment and growth of native browse to benefit mule deer. The overall goal of a grazing plan should be based on maintaining appropriate ecosystem functions. Healthy land benefits wildlife, livestock, and people.

Timing of Cattle Grazing

Rainfall and vegetation growth patterns are unpredictable in habitat occupied year-round by deer, which is the majority of habitat occupied by mule deer in California Woodland Chaparral. Cattle grazing in these locations should typically begin in early winter (with the start of the green-up of annuals), and should end by early spring. Cattle allowed to remain in areas until the grasses and forbs have dried will have eaten most of the palatable plants, and will have removed both deer forage and important fawning cover.

In areas occupied by migratory deer, like the southern Sierra Nevada foothills, Kie and Loft (1990) recommend a plan that: 1) favors mule deer by not grazing cattle during fall and early winter to lessen competition for forbs and acorns, and 2) limits grazing to late winter and spring to reduce annual grass growth and encourage the growth of forbs. When soil moisture is high, grazing cattle in late spring in California Woodland Chaparral habitat also benefits deer by stimulating fast-growing forbs (responding to the combination of high soil moisture and warmer temperatures) to further increase their growth. Once spring has ended and annual forbs have matured and dried, cattle should be removed from the range to prevent heavy browsing of new growth on shrubs and trees. Post-spring removal of cattle also protects riparian areas and other water sources (e.g., springs and canyon-bottom seeps) from excessive use and degradation.

Maintenance of Riparian and Wetland Systems

Riparian areas and springs are of critical importance for resident deer, providing them with fresh water, fawning and escape cover, and shade from the summer heat (Kie and Loft 1990). The potential detrimental effects of cattle on riparian systems are well documented (Bleich et al. 2005). Healthy riparian habitats will benefit cattle as well as deer, in the long-term in California Woodland Chaparral (Thomas et al. 1979, Leckenby et al. 1982). Fencing, and providing water and feed supplements at sources away from rivers, creeks, and streams help distribute cattle evenly rather than concentrating them around riparian areas (Bleich et al. 2005). In areas where water is available only from water sources developed for cattle (holding troughs, wells, ponds, etc.), deer will benefit from modifications that allow them to drink safely (Wilson and Hanans 1977, Andrew et al. 1997). Clary and Webster (1989) recommended residual vegetation stubble heights for riparian areas where excluding cattle is not a realistic option (Table 4).

B. Cattle Stocking Rate

1. Stocking rate is usually defined as “the amount of land allocated to each animal unit for the grazable period of the year” (Society for Range Management 1989). In the California Woodland Chaparral allowable forage use is often expressed as Animal Unit Month (AUM). An AUM is the amount of forage consumed by a cow and a calf < 6 months of age in one month of time.
2. Grazing practices to accomplish deer habitat management goals in California Woodland Chaparral may

Table 4. Recommended minimum residual (post-grazing) vegetation stubble heights for riparian zones where grazing by cattle cannot be avoided (Clary and Webster, 1989).

PRE-GRAZING CONDITION	MINIMUM POST-GRAZING TARGET STUBBLE HEIGHT
Excellent	4 - 6 inches (10-15 cm)
Good	6 - 8 inches (15 - 20 cm)
Poor	No grazing; rest until recovers to good condition

require a cattle stocking rate quite different from traditional range management standards (Kie and Loft 1990). Stocking rates may be heavier for a short time if early seral stage browse growth is desired (e.g., in chaparral), or lighter if increased thermal and escape cover is desired. Forage resources and movement patterns of cattle are also important factors to consider when deciding on stocking rates (Kie and Loft 1990). Stocking rates in this ecoregion will often ultimately be determined by Residual Dry Matter (RDM) measurements (the amount of old dry plant material remaining on the ground at the beginning of a new growing season) to determine use levels (Clawson et al. 1982, and Bartolome et al. 2002). RDM is a standard measurement used by many land management agencies to assess grazing use levels in California Woodland Chaparral habitats (George et al. 1996). RDM is an indication of the previous growing season’s forage production minus its consumption by grazing animals, herbivorous wildlife, insects, and decomposition. “The standard assumes that the amount of RDM remaining in the fall, subject to site conditions and variations in weather, will influence subsequent species composition and forage production” (Bartolome et al. 2002). Once stocking rates are determined for desired RDM levels, they can be consistently used year to year in the same location unless severe drought or other event necessitates adjustments. Combined with knowledge of deer habitat requirements, “stocking rates based on RDM standards [see Tables 5 and 6] usually are

Table 5. Recommended residual dry matter (RDM) levels for annual grasslands and associated woodlands for various rainfall levels and slope grades (Clawson et al. 1982, Bartolome et al. 2002, SSNFPA 2004, and modified based on Holechek et al. 1998).

HABITAT TYPE	LOWER OR FLAT SLOPES (0-10%)	AVERAGE SLOPES (10-30%)	UPPER OR STEEP SLOPES (>30%)
	POUNDS OF RDM/ACRE		
DRY ANNUAL GRASSLAND, ANNUAL RAINFALL < 12 INCHES	400	700	Minimal or no grazing
ANNUAL GRASSLAND W/VARIABLE WOODY CANOPY; ANNUAL RAINFALL BETWEEN 12 - 40 INCHES	700	1,000	Minimal or no grazing
COASTAL PRAIRIE (PERENNIAL GRASSES COMMON, VARIABLE WOODY CANOPY, VARIABLE RAINFALL)	1,200	1,500	Minimal or no grazing

Table 6. Qualitative (visual) categories used to estimate grazing intensities and RDM targets for annual grassland (Clawson et al.1982, Bartolome et al 2002).

QUALITATIVE GRAZING INTENSITY CATEGORY	DESCRIPTION	AVERAGE RDM (POUNDS/ACRE)
LIGHT	Unused plant matter > 3" height Range does not appear patchy, small objects hidden from sight	800 – 1,500
MODERATE	Unused plant matter = 2" height, Range appears patchy with small amount of bare soil visible Small objects not visible at 20 feet	400 - 800
HEAVY	Unused plant matter < 2" height Bare soil visible and small objects visible at > 20 feet	< 400

compatible with [deer] habitat management goals" (Kie and Loft 1990).

- The Natural Resource Conservation Service (NRCS) and University of California Cooperative Extension produce range site guides that: 1) give production estimates to aid in the decision process for determining appropriate cattle stocking rates, and 2) provide range guidelines on how to determine range utilization levels. While stocking rates discussed may be appropriate for livestock production, they may require modification for optimal deer management.
- Steep slopes, areas of extremely dense brush and lands distant from water sources will not be used by cattle and should be deleted from grazable land area (Fulbright and Ortega 2006). Holechek et al. (1998) recommend that lands with slopes between 11 and 30% be reduced in grazing capacity by 30%, lands with slopes between 31 and 60% be reduced in grazing capacity by 60%, and lands with slopes > 60% be deleted from the grazable land area. Also, they suggest that lands 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.
- Heavy stocking rates when forbs are at low levels, even when supported by a supplemental feeding program, are likely to have a negative impact on forbs and browse important to deer (Kie and Loft 1990). Supplemental livestock feeding is often an indication that the habitat is overused, and long-term degradation to wildlife habitat may be occurring.

C. Utilization Rates and Stubble Heights

- Traditional concepts of range condition and trend (Dyksterhuis 1949, Stoddart et al. 1975) do not typically

apply to annual grasslands in the California Woodland Chaparral (Smith 1978, 1988; Kie and Thomas 1988). These concepts result in the annual understory being classified as poor because annual plants are the dominant component. McDougald et al. (1991) developed a scoring system that provides a quick estimation of the grazing capacity of a given area. The system combines rainfall, canopy cover, and slope features of an area to provide an efficient estimate of grazing capacity.

- Annual monitoring of grazing intensity is essential for proper management of rangeland resources for deer. Measurements or assessments of RDM are conducted in the fall, before the onset of the rainy season. Fall weather, along with the water-holding capacity of the soil, and RDM levels determine early annual plant growth (Bentley and Talbot 1951). According to Clawson et al. (1982), sufficient RDM "provides favorable microenvironments for early seedling growth, soil protection, adequate soil organic matter, and a source of low-moisture fall forage for livestock feed." Practicing proper cattle grazing management earlier in the year (by removing cattle from the range in early or late spring, for chaparral or oak woodland, respectively) will usually produce good RDM measurements in the fall. Based on RDM standards, the timing of cattle grazing in this ecoregion may be of more interest to wildlife managers than the actual numbers of cattle at a given location (Kie and Loft 1990). Average plant height may also be an important measurement in the spring to aid in determining when livestock should be removed.
- Tables 5 and 6 present minimum recommended quantitative (Holechek et al. 1998, Bartolome et al. 2002, U.S. Forest Service 2004) and qualitative (Clawson et al. 1982, Bartolome et al. 2002) objectives for yearly RDM

in annual grassland and associated woodlands. The qualitative descriptions provide a low-intensity sampling method to visually assess RDM by comparing the subject range to photos of the grazing intensity categories. These minimum RDMs were developed for optimal range management for livestock production, and may require modification for optimal deer management.

4. Bartolome et al. (2002) provide a good description of how to clip dry vegetation and calculate RDM levels. Methods for monitoring browse use by cattle are discussed in *Monitoring California's Annual Rangeland Vegetation* (Clawson 1990). The amount of browse utilization is important to maintain plant vigor. Browse utilization should not exceed 40% of current annual growth. Methods to monitor browse utilization are also discussed in the *Interagency Technical Reference, Utilization Studies and Residual Measurements* (Bureau of Land Management 1996). Acceptable levels of use, and potential impacts on deer must be determined by the wildlife manager and/or cattle manager.

D. Other Considerations

1. Grazing plans should be flexible enough for the landowners, permittees, and/or the land management agencies to adapt to changing environmental conditions.
2. All fences should meet standards for wildlife passage. The specifications illustrated in the following diagram (Figure 26) provide general guidelines to follow for constructing deer-friendly fences. Fences around riparian or wetland areas may require the top wire to be barbed, depending on the pressure exerted on the fence by cattle. Without the barbed wire at the top, cattle grazing near riparian or wetland areas may eventually damage the fence by pushing a smooth top wire down. The 12" gap between the top barbed wire and the second wire (smooth) will prevent adult deer from entangling their feet between the wires when jumping over the fence (Jepson et al. 1983, U.S. Bureau of Land Management 1985).
3. In most locations, maximizing deer habitat values and cattle productivity at the same time is not realistic. Kie and Loft (1990) stated that "while good range management can provide for good wildlife habitat, the best wildlife habitat often requires modifications of existing livestock management practices."
4. Kie and Loft (1990) also made reference to a trend in California that is relevant in 2006: residential development is still occurring in the ecoregion at a rapid

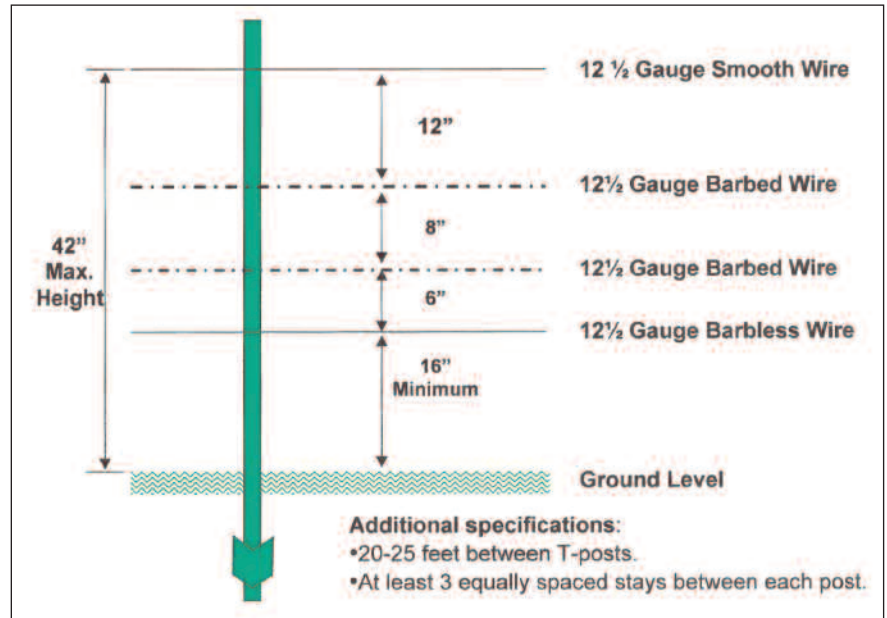


Figure 26. Specifications for a four-strand wildlife-friendly fence. Modification to existing fences can be made by replacing the bottom wire with a smooth wire that is at least 12 inches off the ground. (Adapted from Heffelfinger et al. 2006). Smooth wire (12 1/2 gauge) is recommended for the top wire (barbed wire may be required for protecting riparian or wetland areas).

rate. While cattle ranching may not always result in ideal habitat for mule deer, it is certainly preferable to housing or industrial developments. Resource managers should, whenever possible, work with their local ranching community to preserve open space or risk further loss of deer habitat through development.

WATER AVAILABILITY AND HYDROLOGICAL CHANGES

BACKGROUND

Chaparral habitat typically receives 15-25 inches of precipitation annually. Associated woodland habitats may see up to 40 inches at higher latitudes. In the Mediterranean climate of California, with cool, wet winters and hot, dry summers, < 20% of this precipitation occurs during summer months (Mayer and Laudenslayer 1988). In the chaparral habitats of central Arizona, summer storms are often intense, but much of these torrential rains run off quickly, very little of it serving to recharge groundwater (Swank 1958). On an annual basis these summer storms produce less streamflow than larger, less intense winter storms which yield approximately 90% of annual streamflow. These chaparral habitats are associated with well drained soils and often water courses with intermittent flow under natural conditions. Drought cycles are also a characteristic of chaparral habitats, where there is < 75% of normal precipitation on average occurring every 4.5 years (Hanson and McCulloch 1955).

Human activities have caused lowering of water tables in many areas, which has resulted in seasonal and, in some cases, permanent disappearance of some springs, artesian wells, and streams. As natural water sources have disappeared, artificial sources have been developed throughout the West for livestock and wildlife. These developments provide water for a variety of wildlife species, where natural sources have been depleted. However, in some cases, water is turned off when cattle are moved out of a particular pasture (Scott 1997). Most western wildlife management agencies also have ongoing water development programs specifically for wildlife. At least 5,859 such developments have been built in 11 western states (Rosenstock et al. 1999).

Other than the direct effect of providing freestanding water, precipitation also has indirect effects on deer habitat. Wallmo (1981) noted fawn survival in Arizona chaparral varied with precipitation, attributing the relationship to production of winter-growing forbs. Variations in forb production accounted for approximately 75% of total fawn survival in that area. Also, hydrology and precipitation may have an impact on availability of wetland and riparian vegetation, often important during summer months.

ISSUES AND CONCERNS

Habitat Use and Deer Movements

Mule deer in chaparral vegetation will move 1-1.5 miles to water (Hanson and McCulloch 1955, Swank 1958). Although mule deer may not be completely dependent on free water every day, they do shift areas of activity within home ranges, or even move out of home ranges when water sources dry up (Rogers 1977). Hervert and Krausman (1986) reported when water sources within home ranges of several mule deer does were rendered inaccessible, some does travelled 1-1.5 miles to other water sources to drink. Once they drank, they immediately returned to their home range. In addition, does in later stages of pregnancy have a higher demand for water. Pregnant does use habitat closer to reliable water sources (Clark 1953, Hervert and Krausman 1986). Bowyer (1984) noted a greater need for water by lactating does in Southern California. Does with fawns also remain closer to water in Northern California (Boroski and Mossman 1996).

Sexual segregation may be facilitated by water availability as bucks maintain a greater distance from summer water sources than does in Southern California. Bucks may need less water because larger body size and rumen to body volume ratio reduces rates of water loss. These characteristics may also allow bucks to subsist on drier vegetation (Bowyer 1984).

Water Quality

Small, stagnant pools of water with high evaporation rates

create potential for water quality problems (Kubly 1990). Water quality has been identified as a potential concern for ungulates (Sundstrom 1968, deVos and Clarkson 1990, Broyles 1995). Concerns expressed were potentially toxic algae, bacteria, hydrogen sulfide, and ammonia (Kubly 1990, Schmidt and DeStefano 1996). Although blue-green algae grow in western water sources, Rosenstock et al. (2004) found no evidence of the associated toxins microcystin and nodularin.

Research on quality of water available at artificial and naturally occurring deer water sources in chaparral habitats is lacking. However, deVos and Clarkson (1990) measured water quality variables in 18 wildlife water developments in southwestern Arizona: except for one tinaja (water hole), all sites were within normal limits for conductivity (133-887uS/cm), alkalinity, pH (6.3-9.3, most were 7-8), dissolved oxygen (6-16 mg/l), nitrogen (nitrate), and orthophosphate. The unique site was high in dissolved oxygen, conductivity, and alkalinity.

Broyles (1995) speculated that artificial water in the desert could facilitate spread of ungulate diseases by either providing a growth medium for pathogens or increasing or concentrating populations of a disease vector, such as midges (*Culicoides* spp.). Culicoides gnats carry bluetongue (Trainer 1970) which can impact local deer populations. Rosenstock et al. (2004) found midges widely distributed and locally abundant at both watered and unwatered sites in southwestern Arizona. This distribution was logical, given the discovery that midges could travel > 12 miles from any known or suspected larval development site (Rosenstock et al. 2004). Leptospirosis has been found and suspected in a few deer mortalities, and may be water-borne (Roth 1970).

We are aware of only one documented case of a wildlife water development facilitating spread of a disease. In this case, a bighorn sheep (*Ovis canadensis*) lamb apparently drowned in a water source, with resulting decomposition creating high levels of *Clostridium botulinum*. Growth of this organism in the water likely caused the deaths of > 45 other bighorn sheep due to botulism (Swift et al. 2000).

Benefits of Water

Deer may use water catchments for only part of a year, or not at all during wet years. However, in dry months deer often concentrate around water sources (Wood et al. 1970, Brownlee 1979) and may travel long distances outside their home range to drink (Hervert and Krausman 1986, Rautenstrauch and Krausman 1989). These shifts in distribution indicate water sources are important to deer.

Distribution of water sources is also a potential nutritional factor for deer in chaparral habitats. In some areas where

water is not available year-round, providing permanent water sources might be expected to relieve seasonal concentrations of deer and thereby increase the animals' opportunity for selective foraging (Wallmo 1981). Well-distributed water sources likely distribute deer more evenly through their habitat, thereby allowing them to occupy previously unused areas. This distribution pattern should effectively increase overall carrying capacity of the habitat and reduce frequency of long-range movements out of normal home ranges that could increase susceptibility to predation, energy expenditures, and mortality.

Even if deer do not shift their areas of use, availability of open water allows them to use a greater variety of foods, including very dry forage. If enhanced forage use results in an increased nutritional intake for deer, health and survival should exceed that of deer with less access to water. Hutchings (1946) demonstrated this relationship for domestic sheep, and metabolic use of water by deer is no different than that of sheep (Knox et al. 1969).

Broyles (1995, 1997) expressed concern over the lack of supportive research and potential negative consequences of developing artificial water sources. One concern raised was whether predators are attracted to water sources. If this is the case, more water may result in more predation, which would reduce at least some of the benefit of providing water. DeStefano et al. (2000) found that the presence of predators was seven times greater around water sites than non-water sites. However, scant evidence of predation events near water lead them to conclude water sources were not substantially increasing predation rate on a population-wide level. Scott (1997) speculated that if a new water source is available to cattle and in an area that was formerly lightly grazed, the new water could result in heavier grazing and lead to a reduction in deer cover and forage in that area. If water is not solely for wildlife use, cattle stocking rates may be increased with the addition of more water sources. However, if stocking rates are held constant and new watering sites are established for livestock, grazing pressure can be reduced by better distributing livestock across an allotment. If stocking rates result in overuse in dry periods, this would result in a net decrease in deer habitat quality. Managers need to consider these issues when planning or implementing wildlife water sources. Thus far, however, definitive, population-level negative impacts of water developments are not supported by the data and remain largely speculative (Arizona Game and Fish Department 1997; Rosenstock et al. 1999, 2004).

Water developments for wildlife, however, are not a panacea, and projects should only be initiated where there is a demonstrable need and when other limiting factors are being addressed. Developing a water source without regard for availability of food and cover may be a waste of time

and money and, possibly, degrade habitat. Consideration also must be given to livestock that may use water developed for deer. In some cases these interactions may lessen positive effects of the water project for deer (Kucera and Mayer 1999). There are several ways to make water sources available to livestock and deer, as well as designs to exclude livestock from water sources developed to enhance deer habitat (Payne and Bryant 1994).

GUIDELINES

A. Spacing-Location

1. Mule deer will readily travel 1.5 miles to water, but are found at decreasing densities at greater distances from a water source (Wood et al. 1970, Boroski and Mossman 1996). At a minimum, water sources should not 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). Because deer congregate even closer to water sources during dry periods and fawning, optimum spacing would be one mile between water sources.
2. Actual placement of additional water sources should take into consideration all the resources mule deer need. New water sources alone will not create more usable deer habitat unless they are located near food and cover. Well thought-out placement of water sources will greatly improve their usefulness to deer (Figure 27).

B. Water quality

Managers do not normally need to worry about water quality. If a problem is suspected, a local university or Cooperative Extension Agent may be able to test a water sample. Rosenstock et al. (2004) offered several suggestions

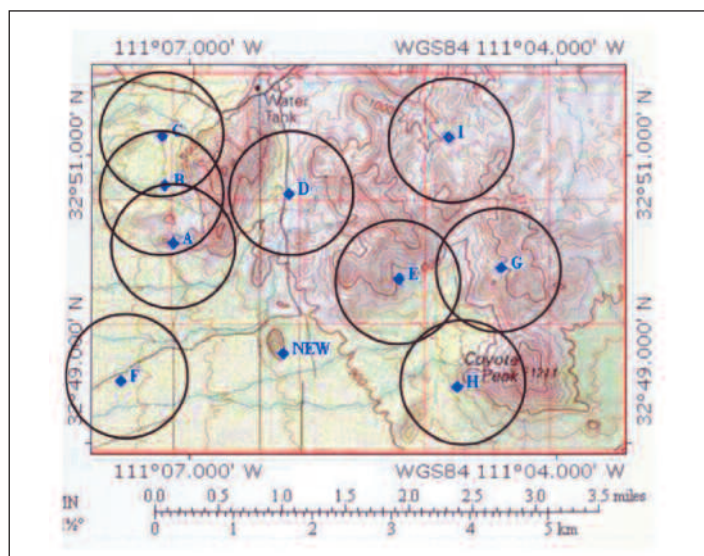


Figure 27. The spacing of available water should be evaluated by mapping sources and circumscribing with 0.5-mile radius buffers. Visualizing water distribution in this way helps to identify areas needing water ("NEW") as well as redundant water sources ("B") (Heffelfinger et al. 2006).

to promote water quality in southwestern water sources that may also be applied in chaparral:

1. For natural catchments, water quality depends on frequency of flushing during rainfall events. These types of water sources should be designed or modified to promote periodic flushing.
2. Where possible, provide natural or artificial shade over the water source to reduce evaporation and growth of algae.
3. Periodically remove organic debris, dead animals, floating algae, and accumulated sediment. If any protected amphibian species (e.g. California tiger salamanders (*Ambystoma californiense*), California red-legged frogs (*Rana aurora draytonii*), or other listed species) is likely to be present, remove debris or sediments in late summer or early fall to avoid impacts. Contact with the U.S. Fish and Wildlife Service may be necessary prior to disturbance or modification of potential amphibian habitat.
4. Use designs that reduce accumulation of sediment at the water's edge to avoid encouraging growth of unwanted vegetation and eliminate the presence of moist substrate used by disease vectors such as midges.

C. Design

Four primary types of water developments have been constructed in the western United States: 1) modified natural tanks, 2) artificial catchments, 3) developed springs, and 4) wells (Figures 28, 29, and 30; Rosenstock et al. 1999). Within these categories, there are an unlimited number of water development designs based on target species and physical features of the site. No single design will be right for every situation. However, with the decades of experience that some agencies have with designs, there are components and systems that have proven to be undesirable. Those interested in building water developments for mule deer should take advantage of this experience to avoid repeating mistakes. Wildlife water development standards are available that describe, in detail, specifications for each component of various water development designs (Arizona Game and Fish Department 2004).

D. Storage Capacity

Storage capacity of an artificial catchment is a critical part of the design. Capacity should consider evaporation rate of exposed water, average amount and timing of precipitation, and number of animals using the water during critical times. Evaporative rates are difficult to calculate because of the complex variables involved, but designs should incorporate effective evaporation control measures. Local precipitation patterns will govern size of the apron needed when designing water catchment systems. For every 160 square feet of catchment apron, 100 gallons will be captured for each 1 inch of rainfall. Depending on



Figure 28. Spring flow diverted to fill water trough for deer in San Diego County, California (Photo by Randy Botta/CDFG).



Figure 29. Upland game guzzler modified to provide water for deer. Note trough in lower left corner is connected to the guzzler to maintain water level (Photo by Randy Botta/CDFG).



Figure 30. Deer guzzler under construction, San Benito County, California. The collection apron will be installed over the underground storage tanks (left of the access ramp) (Photo by Phil Pridmore/CDFG).

topography, a small dam may be used to divert additional rainfall into the storage tank or may be the sole collection apparatus for the catchment. When diverting natural flows, water rights issues must be considered. Number of animals drinking will impact the amount of water needed to sustain year-round availability. When there is very little moisture in forage plants, mule deer may consume 4-10 quarts (average = 6.3 qts.) per day (Elder 1954, Hervert and Krausman 1986).

E. Other considerations

Experience has shown there are criteria that can significantly increase usefulness, dependability, and lifespan of a water source. The Arizona Game and Fish Department (2003) developed such a list of “Criteria for Success”:

1. Has a long lifespan (40-50 years for storage and collection systems, 25 years for drinking troughs);
2. Meets clearly-articulated biological needs;
3. Provides year-round, acceptable water quality for wildlife use;
4. Maximizes passive designs elements, while using proven components applied or installed per manufacturer’s specifications;
5. Does not require supplemental hauling except in rare or exceptional circumstances;
6. Has minimal visual impacts and blends in with surrounding landscape;
7. Has vehicular access to development or close by, to facilitate routine maintenance and inspections;
8. Is built with the greatest possible time and cost efficiency;
9. Requires minimal routine maintenance;
10. Is accessible to and used by target species (including fawns) and excludes undesirable or feral species to the greatest extent possible;
11. Minimizes risk of animal entrapment and mortality; and
12. Camping or other extended, high recreational use should be prohibited in close proximity. In California, camping or occupying areas near wildlife water developments is prohibited (California Code of Regulations, Title 14, section 730).

NON-NATIVE INVASIVE SPECIES

BACKGROUND

European explorers first visited what is now California in 1524, but they did not begin to settle there until 1769. Based on available evidence, most non-native plants that are now established in California were introduced after European settlement (Crampton 1974, Barry et al. 2006). The number of these species increased rapidly from less than 20 in 1824 to 1,045 in 1998 (Bossard et al. 2000). Originally many of these plants were brought in accidentally in ship ballast water and grain shipments, and others were introduced deliberately for use in food and

fiber, in medicine, and for ornamental plantings (Crampton 1974, Bossard et al. 2000). Currently, land managers still use non-native plants for erosion control and livestock forage. Further unintentional infestations may occur as a result of transporting non-native plant propagules in gravel, roadfill, feed, and mulch (Bossard et al. 2000). In some cases these species spread at an exponential rate. For example, the range of yellow starthistle (*Centaurea solstitialis*), in California expanded from 1.2 million acres in the 1950s to possibly as much as 22 million acres today (Figure 31; Holloran et al. 2004).

Deer habitat throughout most rangeland of the western United States has been altered by land management practices to improve livestock production. In addition to direct impacts from cattle grazing, fencing, and changes in water availability, range managers and ranchers in some regions have promoted expansion of non-native plant species (Bossard et al. 2000). In some cases, the introduction of non-natives has been by purposeful plantings for improved livestock forage. In other cases, introductions have been accidental, or incidental to other activities. In many cases, invasion by non-native plants in California has been detrimental to deer habitat. However invasive annual forbs and grasses may also be an important component of mule deer diets in late winter and spring (Kucera and Mayer 1999).

Habitat alterations throughout the western United States have had profound effects on native wildlife (Bock and Bock 1995, Cal-IPC 2006). Invasion by non-native species has resulted in significant environmental impact, causing structural changes and, in some cases, alteration of habitat type, as well as changes in plant composition and diversity (Bossard et al. 2000, Holloran et al. 2004). Invasions of plant pathogens can also impact native species, such as the spread of *Phytophthora ramorum* that causes Sudden Oak Death (SOD). Since its discovery in Mill Valley (Marin County) of California in 1995, SOD has killed tens of thousands of native coast live oaks and tan oaks (*Lithocarpus densiflorus*) along California’s north and central coasts (Cole 2001). SOD is believed to have originated in China and introduced to California in shipments of nursery stock. About 42% of species listed as threatened or endangered in the United States are at risk because of factors related to non-native species. In addition, economic losses due to non-native species are estimated to be in excess of \$138 billion per year (Pimentel et al. 1999).

The California Woodland Chaparral Ecoregion contains a variety of shrub and woodland plant communities (Mayer and Laudenslayer 1988). This diversity of habitats in the ecoregion has provided opportunities for invasion by many non-native plant species (Cal-IPC 2006). Twenty-two (22) species of plants in the California Woodland Chaparral are



Figure 31. Dense infestations of yellow star thistle displace native plants and animals (Photos by Martha Schauss/CDFG and J.S. Peterson/USDA-NRCS PLANTS Database).

rated highly invasive by the California Invasive Plant Inventory (Table 7; Cal-IPC 2006).

ISSUES AND CONCERNS

Impacts of Non-native Plants on Deer and Deer Habitat

Many areas in California Woodland Chaparral habitats have non-native plant species that were never planted purposefully, but have invaded and become dominant in what is now a modified plant community (Crampton 1974, Holloran et al. 2004, Barry et al. 2006). Spreading and subsequent dominance of non-natives may be greatest on areas that have been heavily impacted by overgrazing or other disturbances. Roads are major contributing factors to the ongoing spread of exotic plants (Gelbard and Belnap 2003).

The most destructive of the invasive plant species are capable of affecting the entire ecosystem, so that native plants have difficulty competing and surviving. Alteration of ecosystem processes such as nutrient cycling, fire regimes, hydrological cycles, sediment deposition, and erosion can have disastrous effects, placing many species at a severe disadvantage. Some invasive plants completely change community structure of invaded habitat, often excluding beneficial native plants (Figure 32; Stein and Flack 1996, Bossard et al. 2000, Cal-IPC 2006).

Some non-native invasive plants, such as filaree and bindweed (*Convolvulus arvensis*), benefit deer by improving forage availability during part of the year. Bertram (1984) reported that the following non-native forbs were important to migratory deer occupying winter range in the Sierra Nevada foothills: red-stem filaree (*Erodium cicutarium*), broad-leaved filaree (*Erodium botrys*), lotus (*Lotus* spp.), and clover, although identification of the two latter genera as to native or non-native was not indicated. There are other non-native plants that seem to have little impact on deer habitat, or for which benefits and detriments are not known. The following discussion will address species that are generally considered to influence deer habitat, either through reduction in nutritional levels, or by less direct impacts to deer food, water, or cover availability. More detailed species accounts can be found in “*Invasive Plants of California’s Wildlands*” (Bossard et al. 2000).

Grassland Invasives

Perhaps the most widespread habitat alteration throughout the California Woodland Chaparral ecosystem has been the almost complete replacement in the grasslands and in the woodland understory of native perennial grasses by non-native annual grasses and forbs (Barry et al. 2006, Mayer and Laudenslayer 1988). Annual grasses provide significant forage for deer only in early growing stages, with little nutritional value or palatability when dry. Native perennial bunchgrasses may have provided higher quality forage later

TABLE 7. HIGHLY INVASIVE PLANTS OF THE CALIFORNIA WOODLAND CHAPARRAL

SCIENTIFIC NAME	COMMON NAME
<i>Aegilops triuncialis</i>	barb goatgrass
<i>Arundo donax</i>	giant reed
<i>Brassica tournefortii</i>	Saharan mustard, African mustard
<i>Bromus madritensis</i> ssp. <i>rubens</i> (= <i>B. rubens</i>)	red brome
<i>Bromus tectorum</i>	downy brome, cheatgrass
* <i>Centaurea solstitialis</i>	yellow starthistle
<i>Cortaderia selloana</i>	pampasgrass
<i>Cytisus scoparius</i>	Scotch broom
<i>Delairea odorata</i>	Cape-ivy
* <i>Euphorbia esula</i>	leafy spurge
<i>Foeniculum vulgare</i>	fennel
<i>Genista monspessulana</i>	French broom
<i>Hedera helix</i> , <i>H. canariensis</i>	English ivy, Algerian ivy
<i>Lepidium latifolium</i>	perennial pepperweed, tall whitetop
* <i>Lythrum salicaria</i>	purple loosestrife
* <i>Onopordum acanthium</i>	Scotch thistle
<i>Rubus armeniacus</i> (= <i>R. discolor</i>)	Himalaya blackberry
<i>Spartium junceum</i>	Spanish broom
<i>Taeniatherum caput-medusae</i>	medusahead
<i>Tamarix parviflora</i>	smallflower tamarisk
<i>Tamarix ramosissima</i>	saltcedar, tamarisk
<i>Ulex europaeus</i>	gorse
*Also listed as noxious weeds by the Arizona Department of Agriculture	

into the summer months, the critical period for deer of this region. Displacement of perennials by annuals has also altered fire behavior in both grasslands and woodlands (Bossard et al. 2000). Faster moving, hotter fires carried by dry annuals in summer and early fall can be more destructive than slower burns, and may negate benefits of patchy or mosaic burn patterns.

Dominance by annual grasses may also impact oak regeneration by allowing greater dispersal of rodents that girdle oak seedlings and eat acorns, and by competing with seedlings for water and nutrients (Figure 33; Pavlik et al. 1991, Holloran et al. 2004). As oak leaves and acorns are important components of deer diets, particularly in summer and fall (Taber and Dasmann 1958, Schauss and Coletto 1986, unpublished California Department of Fish and Game

report, Pavlik et al. 1991), any factors reducing oak reproduction and survival must be considered a long-term detriment to deer habitat.

Non-native grasses widely distributed in this region include wild oats (*Avena* spp.), ripgut grass (*Bromus diandrus*), soft chess (*B. hordeaceus*), red brome (*B. rubens*), foxtail chess (*B. madritensis* ssp. *rubens*), and farmer's foxtail (*Hordeum murinum* ssp. *leporinum*) (Mayer and Laudenslayer 1988). Several species, including farmer's foxtail and ripgut grass have mature seeds with stiff awns that are known to cause injury to wildlife that move through grasslands or feed on mature plants (Holloran et al. 2004). Such injuries have been found in juvenile wild pigs and badgers, and may also be problematic for deer fawns. Medusahead (*Taeniatherum caput-medusae*) is a particularly invasive grass, with origins

in Europe. It contains high levels of silica, and is highly unpalatable (Bossard et al. 2000). Barbed goatgrass (*Aegilops triuncialis*) is another particularly noxious and invasive plant that has become established in northern California and parts of the Sacramento Valley and Sierra foothills. Cheatgrass is highly invasive in parts of northeastern California, where it has replaced more palatable forage over large areas. Medusahead has become established in some locations in this ecoregion, and seems to be spreading. Locations of barbed goatgrass and cheatgrass are spotty within the ecoregion, but have the potential to degrade many acres of wildlife habitat.

In addition to annual grasses, there are a number of non-native forb species with little or no value to deer that displace native species and reduce forage availability. These include poison hemlock (*Conium maculatum*), yellow star thistle, artichoke thistle (*Cynara cardunculus*), mustards (*Brassica* spp.), hoary cress (*Cardaria draba*), bull thistle (*Cirsium vulgare*), fennel (*Foeniculum vulgare*), Italian thistle (*Carduus pycnocephalus*), and perennial pepperweed (*Lepidium latifolium*) (Bossard et al. 2000, Holloran et al. 2004). These are often found in areas disturbed by overgrazing, disking, or grading (Figure 34). The extent of displacement of native species by these and other non-native plants is often overlooked. It is common to find grasslands in which native species are nearly absent (Bossard et al. 2000, Barry et al. 2006). While the effect on deer habitat cannot be precisely measured, it has doubtless been substantial.

Shrubland and Woodland Invasives

Non-native shrub and tree species have also been introduced into the California Woodland Chaparral Ecoregion, though most with more limited distribution than non-native grasses and forbs. Pampas grass (*Cortaderia* spp.) is found primarily in coastal areas where it was planted as an ornamental and for erosion control (Bossard et al. 2000). Sharp, cutting leaves of pampas grass make it both unpalatable and impenetrable (Barry et al. 2006). Both castor bean (*Ricinus communis*) and tree-of-heaven (*Ailanthus altissima*) are rapid-spreading ornamentals that can displace large amounts of native habitat where they become established. Leaves, and particularly the seeds, of castor bean are highly toxic (Bossard et al. 2000). French broom, Scotch broom, and Spanish broom are all shrubby invasives that may dominate plant communities where they become established, and have little or no forage value for wildlife (Bossard et al. 2000, LeBlanc 2001). Seeds of Scotch broom are toxic to ungulates, and shoots are unpalatable (Holloran et al. 2004). Gorse is a spiny shrub that is found in numerous locations in the central coast of California, particularly in disturbed areas. Once established, it is difficult to control, and may invade grassland habitats. Eucalyptus (*Eucalyptus* spp.) has been planted in many



Figure 32. Tamarisk invasion in the foreground has completely replaced native riparian vegetation (Photo by Mary Sommer/CDFG).



Figure 33. Annual grasses compete with oak seedlings and may facilitate rodent damage (Photo by Mary Sommer/CDFG).



Figure 34. Strong competitors for nutrients and moisture, mustard and hoary cress dominate this hillside displacing vegetation used by wildlife (Photo by Martha Schauss/CDFG).

areas as ornamental trees, windbreaks, and for commercial purposes. It is highly unpalatable to deer and other wildlife, is highly flammable, and inhibits growth of other plants beneath it by production of allelopathic chemicals and large amounts of debris (Bossard et al. 2000, Holloran et al. 2004).

While the spread of non-native shrubs and trees is not as prevalent as that of non-native grasses and forbs, these plants can have large impacts in local areas, substantially reducing deer forage availability and quality.

Riparian Invasives

Poison hemlock and castor bean are among invasive non-native plants that thrive in disturbed riparian areas as well as other habitats. Several highly invasive plants are also found primarily in riparian habitats in this region: giant reed or arundo (*Arundo donax*), tamarisk or salt-cedar (*Tamarix* spp.), and cape ivy (*Delawarea odorata*) (Bossard et al. 2000, Holloran et al. 2004).

These species have greatly impacted riparian habitats in many locations in the California Woodland Chaparral Ecoregion, and threaten many more. Arundo forms large stands that displace willows (*Salix* spp.), cottonwoods (*Populus* spp.), and other native species, and may cover entire river channels (Holloran et al. 2004). Like arundo, tamarisk can dominate riparian communities, alter channel morphology and stream structure, and reduce groundwater (Bossard et al. 2000). Tamarisk can also increase soil salinity, further inhibiting growth of native plant species (Stein and Flack 1996). Cape ivy can smother other plants in moist areas, including riparian trees, substantially reducing plant diversity (Figure 35). In addition to devastating riparian vegetation, cape ivy contains compounds that are toxic to mammals, and has no forage value (Bossard et al. 2000).

GUIDELINES

A. Planning

While there are numerous methods of approaching the management of non-native invasive species, the following adaptive management approach detailed by Bossard et al. (2000) uses a straight-forward rationale for actions to be taken. This process progresses as follows: (1) establish management goals and objectives for the site; (2) determine which plant species or populations, if any, block or have potential to block attainment of management goals and objectives; (3) determine which methods are available to control the weed(s); (4) develop and implement a management plan designed to move conditions toward



Figure 35. Cape ivy blankets trees and other riparian plants, while providing no forage value for wildlife (Photo by Joel Trumbo/CDFG).

management goals and objectives; (5) monitor and assess the impacts of management actions in terms of effectiveness in moving toward goals and objectives; and (6) reevaluate, modify, and start the cycle again.

Mapping is an excellent tool to aid in prioritizing work, monitoring progress, and documenting what has been done. Maps can be created by hand or by using a Geographical Information System (GIS) and data collected with a Global Positioning System (GPS). More information can be found on both mapping methods in the California Department of Food and Agriculture's weed mapping handbook at cain.nbii.gov/weedhandbook (Holloran et al. 2004). On a small scale, managers may want to use maps of vegetation associations to record and track mule deer sightings or other data. Trend data from changes in deer occurrence or abundance may help identify habitat use and preferences to guide future habitat manipulations.

Managers must always consider all other social and economic demands for management of the land. In areas of predominantly private land, habitat management plans will not be successful without cooperation and coordination with the landowner. In some cases, continued use of the land for livestock grazing or other activities that may be disruptive to natural ecosystem function may make control of established non-native plant species difficult or impractical. In other cases, grazing may assist in the control on non-native plant species if conducted in a prescribed manner.

B. Specific Guidelines

1. Identify negative and positive effects of habitat alterations such as non-native plantings and use this information for adaptive management in future land use decisions.
2. Promote native species production with the focus on plants used or preferred by deer.
3. Where livestock are present, use proper grazing practices such as appropriate stocking rates and rotation to favor native browse establishment to benefit mule deer. Intensive grazing may be used as a management tool on invasive species during periods of intense growth to reduce seed production, plant vigor, and storage of nutrients. However, intensive grazing must be carefully monitored and impacts measured to prevent further habitat degradation.
4. Mitigate negative effects of past pasture plantings: allow natural successional appearance of shrubs and trees to create cover for deer.
5. Use native species when possible and practice proper range management to expedite rehabilitation of deteriorated areas. Identify areas that are deteriorated but lacking invasive plant species and make these a high priority for proactively seeding native species. Use locally collected propagules of native species whenever possible to maintain genetic integrity of the ecosystem.
6. Consider potential for non-native plant invasion when deciding whether to build, improve, or maintain roads.
7. Avoid soil disturbance, particularly in sites that have not been previously disked or disturbed, as disturbance often provides the opportunity for weedy invasives to become established and outcompete native species.
8. Consider potential of unintentional transport and introduction of unwanted non-native plant species when moving livestock. Horses are particularly likely to promote spread of viable seeds because they digest plant materials less completely than do ruminants. Cattle and sheep may carry seeds in their coats and on hooves. Any hay or other forage fed to livestock should be free of seeds of non-native plants that are not already present at the site.

While native plant species are generally desired, use of non-native species can be a valid mule deer habitat management option. In some cases it is impossible to use all or only native seed, as in a year with extensive wildfires, where there is not enough native seed available. In this case not seeding some areas could result in erosion that could preclude seeding in the future or require extensive land treatments to reclaim eroded areas. If non-native plants are used, species can be selected that are palatable for mule deer and not highly competitive to the establishment of shrubs and forbs. Such an approach should include a commitment to revegetate with native plants at an appropriate time in the future. The decision of what to plant will depend on each specific case, and conditions need to

be considered prior to any vegetation management actions.

It is unlikely that non-native species will be eliminated from invaded areas, but the primary management goal should be to reduce spread of non-natives, change vegetation composition to reduce non-native dominance, and promote higher plant diversity.

C. Prevention and Control Methods

Prevention

Control of non-native invasive plants should begin with a comprehensive prevention strategy. Preventing invasions, and quickly addressing new invasions, is far less expensive both in dollars and effort than treating an already established infestation (Holloran et al. 2004). Simple precautions can be taken to avoid spread of invasive plants: washing vehicles and equipment before using them in a different area, monitoring work sites for new non-native plant species, and public education targeted at stopping spread of these species (Bossard et al. 2000). Other prevention measures include removing seed sources from dispersal routes (roads, trails, waterways); closing unnecessary travelways; minimizing soil disturbance at work sites; and limiting use of materials such as gravel, fill, straw, and seed mixes. A proactive approach to management of non-native invasive plants in normal resource management activities can assist greatly (Bossard et al. 2000, Holloran et al. 2004).

Control Methods

The following list contains methods of treatment for invasive plants taken from Bossard et al. (2000).

- Physical Control – manual hand pulling or use of power tools to uproot, girdle or cut plants.
- Prescribed Fire – particularly effective in communities that evolved with fire.
- Flooding and Draining – prolonged flooding can kill plants in areas where water levels can be controlled.
- Mulching – excludes light from weeds and prevents photosynthesis.
- Soil Solarization – a method for killing seeds by placing plastic sheeting over moist soil for a month or more.
- Biological Control – involves use of animals, fungi, or microbes to consume, kill, or weaken a target species.
- Competition and Restoration - use of native plants to outcompete alien weeds is a frequently overlooked but potentially powerful technique.
- Grazing – can be used to selectively control or suppress unwanted species, if managed carefully.
- Chemical Control – herbicides can be extremely effective in eliminating certain species.

Circumstances of each infestation will be unique and require careful consideration of site conditions. Often a combination of two or more methods works better than



(Photo by CDFG)

using any one exclusively. Be sure to consult professionals that have invasive species experience. An excellent reference with information on both native and non-native invasive plant species is *“Invasive Plants of California’s Wildlands”* (Bossard et al. 2000); available on-line at Cal-ipc.org. Detailed information on herbicides is available in the Weed Science Society of America’s *Herbicide Handbook* (Ahrens 1994) and *Supplement* (Hatzios 1998).

Additional information and training on weeds and their control can be found by contacting local universities, extension agents, county weed and pest supervisors, California Department of Food and Agriculture, and California Department of Pesticide Regulation. The California Exotic Pest Plant Council can direct readers to other local experts on weeds. The Bureau of Land Management offers an Integrated Pest Management and Pesticide Certification course in Denver, Colorado, and the Western Society of Weed Science offers a Noxious Weed Management short course in Bozeman, Montana.

Mule deer habitat in the California Woodland Chaparral Ecoregion and in other areas has not reached its current condition because of any one factor or contributing cause. Many factors are closely interrelated and most of them lead to a decrease in mule deer habitat quality or quantity. Single or combined impacts of these contributing factors either directly or indirectly alter key plant species by determining structure, composition, and function of plant communities. Natural disturbances to the system are needed to produce quality mule deer habitats. Disturbances can result in positive or negative changes in deer habitat. Unfortunately, most on-going disturbances are not positive for mule deer. Form, magnitude, and timing of disturbance are critical to achieving positive outcomes and management is required to achieve these results.

A key and often overlooked constant factor leading to deterioration in many mule deer habitats is ecological succession. During the past century, absence of fire where it originally occurred has been a major contributing factor to declines in quality mule deer habitats.

Because of its impact on plant composition and structure, grazing by both wild and domestic herbivores commonly impacts mule deer habitats. Herbivores either directly or indirectly influence the likelihood that a plant community will burn by changing the amount of volatile understory herbage. Heavy grazing by herbivores also increases the likelihood that invasive plants will take hold by removing valuable native species. Furthermore, improper grazing regimes may directly influence the hydrologic cycle of plant communities by altering moisture infiltration and runoff, as is often observed with habitat losses seen in riparian areas.

Inadequate availability of water may be a limiting factor for mule deer in some habitats. Development and maintenance of appropriately spaced artificial water sources is sometimes required and these need to be maintained even after cattle are removed from individual pastures. Often, initiation of appropriate livestock grazing regimes will result in improved hydrological conditions and natural water will return to previously dry springs or streams. Restoration of natural water sources should be a long-term goal for habitat managers. If artificial water sources are required, much experience has been gained in design and maintenance of these sources and managers should use development approaches that are proven to be successful.

Humans influence mule deer habitats directly and indirectly. Direct loss of habitat to cities, ranchettes, aqueducts, highways, roads, and agriculture is obvious, but little mitigation has been provided. The accelerated rate of development across the California landscape is a constantly growing threat to mule deer habitats. Increased use of roads and recreational vehicles negatively influence distribution of mule deer and may render otherwise suitable habitats unsuitable for mule deer, as well as leading to substantial levels of direct mortality in some locations. High levels of human activity in mule deer habitats can produce undesirable outcomes for deer populations. Recreational pursuits must be managed to provide areas free of constant human activity.

Mule deer have relatively smaller rumens than elk or livestock and thus must depend on a more diverse habitat consisting of a variety of plant species and plant structures. Diversity in forage choices provides concentrated and more digestible nutrients that are needed by mule deer. A common outcome of limiting factors discussed in this document is a tendency towards less plant diversity and, in many cases, plant monocultures dominated by less desirable or invasive plant species. These outcomes almost always mean plant communities with lower nutritional quality for mule deer.

The appropriate mix and age structure of forage species is important to high quality mule deer habitats. Contributing factors discussed in these guidelines play a large role in determining distribution and age structure of shrub communities. Shrubs and woodland vegetation provide needed cover for mule deer and must be sufficiently abundant and distributed across the landscape in a manner that provides adequate shelter from weather and predators. Old shrubs are lower in nutrition and often produce biomass that is out of reach of deer, but may provide valuable hiding and thermal cover. Too much woody cover suppresses amount and diversity of valuable understory herbaceous forage. Active management is required to maintain the appropriate balance of forage and cover requirements in shrub communities. Prescribed fire appears to be the most effective tool to achieve these needs in woodland and chaparral habitats.

Hopefully, the guidelines provided in this document will aid resource managers in creating habitat conditions in woodland and chaparral environments conducive to mule deer. These habitats can be very productive for mule deer, but active and thoughtful management is required. These guidelines were prepared to help meet that need.

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APPENDIX

APPENDIX A.

Alphabetical listing, by category, of species cited in the text.

TREES AND SHRUBS

Ceanothus (*Ceanothus* spp.)
Chamise (*Adenostoma fasciculatum*)
Cottonwood (*Populus* spp.)
Gray Pine (*Pinus sabiniana*)
Mountain mahogany (*Cercocarpus betuloides*)
Oak, Coast Live (*Quercus agrifolia*)
Oak, Valley (*Quercus lobata*)
Oak, Blue (*Quercus douglasii*)
Ponderosa pine (*Pinus ponderosa*)
Redshank (*Adenostoma sparsifolium*)
Redwood (*Sequoia sempervirens*)
Tan oak (*Lithocarpus densiflorus*)
Willow (*Salix* spp.)

FORBS AND GRASS

Broad-leaved filaree (*Erodium botrys*)
Brodiaea (*Brodiaea* spp.)
Clover (*Trifolium* spp.)
Filaree (*Erodium cicutarium* or *Erodium* spp.)
Lotus (*Lotus* spp.)
Miner's lettuce (*Claytonia perfoliata*)
Popcorn flower (*Plagiobothrys* spp.)
Red-stem filaree (*Erodium cicutarium*)
Wild oats (*Avena* spp.)

INVASIVE PLANTS

Artichoke thistle (*Cynara cardunculus*)
Arundo or giant reed (*Arundo donax*)
Barbed goatgrass (*Aegilops triuncialis*)
Barb goatgrass (*Aegilops triuncialis*)
Bindweed (*Convolvulus arvensis*)
Bull thistle (*Cirsium vulgare*)
Cape ivy (*Delairea odorata*)
Castor bean (*Ricinus communis*)
Cheatgrass, downy brome (*Bromus tectorum*)
English ivy, Algerian ivy (*Hedera helix*, *H. canariensis*)
Eucalyptus (*Eucalyptus* spp.)
Farmer's foxtail (*Hordeum murinum* ssp. *leporinum*)
Fennel (*Foeniculum vulgare*)
Fiddleneck (*Amsinkia* spp.)
Foxtail chess (*B. madritensis* ssp. *rubens*)
French broom (*Genista monspessulana*)
Gorse (*Ulex europaeus*)
Himalaya blackberry (*Rubus armeniacus* or *R. discolor*)
Hoary cress (*Cardaria draba*)
Italian thistle (*Carduus pyncephalus*)
Leafy spurge (*Euphorbia esula*)
Medusahead (*Taeniatherum caput-medusae*)
Mustards (*Brassica* spp.)
Pampasgrass (*Cortaderia selloana*) Pampas grass

(*Cortaderia* spp.)
Peppergrass (*Lepidium* spp.)
Perennial pepperweed, tall whitetop (*Lepidium latifolium*)
Poison hemlock (*Conium maculatum*)
Purple loosestrife (*Lythrum salicaria*)
Red brome (*Bromus madritensis* ssp. *rubens* or *B. rubens*)
Ripgut grass (*Bromus diandrus*)
Saharan mustard, African mustard (*Brassica tournefortii*)
Saltcedar, tamarisk (*Tamarix ramosissima*)
Scotch broom (*Cytisus scoparius*)
Scotch thistle (*Onopordum acanthium*)
Smallflower tamarisk (*Tamarix parviflora*)
Soft chess (*B. hordeaceus*)
Spanish broom (*Spartium junceum*)
Tree-of-heaven (*Ailanthus altissima*)
Yellow starthistle (*Centaurea solstitialis*)

MAMMALS

Bighorn sheep (*Ovis canadensis*)
California mule deer (*Odocoileus hemionus californicus*)
Columbian black-tailed deer (*Odocoileus hemionus columbianus*)
Coues white-tailed deer (*Odocoileus virginianus couesi*)
Desert mule deer (*Odocoileus hemionus eremicus* [= *crooki*]).
Mule deer (*Odocoileus hemionus*)
Pocket gophers (*Thomomys bottae*)
Pronghorn antelope (*Antilocapra americana*)
Rocky Mountain elk (*Cervus elaphus nelsoni*)
Rocky Mountain mule deer (*Odocoileus hemionus hemionus*)
Roosevelt elk (*Cervus elaphus roosevelti*)
Southern mule deer (*Odocoileus hemionus fuliginatus*)
Tule elk (*Cervus elaphus nannodes*)
Wild pigs (*Sus scrofa*)

AMPHIBIANS

California red-legged frog (*Rana aurora draytonii*)
California tiger salamander (*Ambystoma californiense*)

MISCELLANEOUS

Botulism (*Clostridium botulinum*)
Midges (*Culicoides* spp.)
Pathogen that causes Sudden Oak Death (*Phytophthora ramorum*)

APPENDIX B.

List of browse plants used by mule deer in the California Woodland Chaparral ecoregion. Species separated by state.

CALIFORNIA (Adapted from Sampson and Jespersen 1963, Holl et al. 1979, and Schauss and Coletto 1986, unpublished California Department of Fish and Game report).
Over-all browse ratings for deer are indicated with the plant

name. Rating symbols are: 1 = excellent; 2 = good; 3 = fair; 4 = poor; 5 = useless.

TREES AND SHRUBS

- Allscale (*Atriplex polycarpa*) 2
Arizona ash (*Fraxinus velutina*) 4-5
Arroyo willow (*Salix lasiolepis*) 3
Australian saltbush (*Atriplex semibaccata*) 3
Big sagebrush (*Artemisia tridentata*) 2-4
Big-leaf maple (*Acer macrophyllum*) 3-4
Bitter cherry (*Prunus emarginata*) 1-2
Black cottonwood (*Populus trichocarpa*) 3-4
Black sagebrush (*Artemisia nova*) 2-3
Blue elderberry (*Sambucus caerulea*) 2-4
Blue oak (*Quercus douglasii*) 1-2
Blue witch (*Solanum umbelliferum*) 1-3
Blueblossom ceanothus (*Ceanothus thyrsiflorus*) 2-3
Brewer's willow (*Salix breweri*) 3
Buckbrush or wedgeleaf ceanothus (*Ceanothus cuneatus*) 3
Budsage (*Artemisia spinescens*) 3-4
Buffalo berry (*Shepherdia argentea*) 3-4
Bush poppy (*Dendromecon rigida*) 3-4
California black oak (*Quercus kelloggii*) 1-2
California boxelder (*Acer negundo* var. *californicum*) 3-4
California buckeye (*Aesculus californica*) 1-2
California buckwheat (*Eriogonum fasciculatum*) 2-3
California coffeeberry (*Rhamnus californica*) 2-4
California hazelnut (*Corylus cornuta* var. *californica*) 3-4
California juniper (*Juniperus californica*) 3-4
California laurel (*Umbellularia californica*) 2-3
California scrub oak (*Quercus dumosa*) 1-2
California wild grape (*Vitis californica*) 3-4
California wild rose (*Rosa californica*) 3-4
California yerba santa (*Eriodictyon californicum*) 3-4
Canyon gooseberry (*Ribes menziesii*) 3-5
Canyon live oak (*Quercus chrysolepis*) 3-4
Chamise (*Adenostoma fasciculatum*) 2-3
Chaparral pea (*Pickeringia montana*) 1-2
Chaparral whitethorn (*Ceanothus leucodermis*) 1-2
Coast live oak (*Quercus agrifolia*) 3-4
Coast sagebrush (*Artemisia californica*) 4
Common snowberry (*Symphoricarpos albus*) 3-4
Coyote brush or chaparral broom (*Baccharis pilularis*) 4-5
Curlleaf mountain-mahogany (*Cercocarpus ledifolius*) 1
Deerbrush ceanothus (*Ceanothus integerrimus*) 1-2
Deerweed (*Lotus scoparius*) 3-4
Desert bitterbrush (*Purshia glandulosa*) 1-2
Desert sage (*Salvia carnososa*) 3-5
Dogwood (*Cornus sericea*) 3-4
Eastwood manzanita (*Actostaphylos glandulosa*) 4-5
Evergreen huckleberry (*Vaccinium ovatum*) 3-4
Fourwing saltbush (*Atriplex canescens*) 2-3
Fremont cottonwood (*Populus fremontii*) 3-4
Fremont silktassel (*Garrya fremontii*) 2-3
Fremontia or flannel bush (*Fremontia californica*) 1
Fuchsia flowering gooseberry (*Ribes speciosum*) 3-5
Goatnut (*Simmondsia chinensis*) 2
Gray horsebrush (*Tetradymia canescens*) 3-5
Green ephedra (*Ephedra viridis*) 3-4
Hillside gooseberry (*Ribes californicum*) 3-4
Hoary manzanita (*Arctostaphylos canescens*) 4
Hollyleaf cherry (*Prunus ilicifolia*) 2-3
Hollyleaf redberry (*Rhamnus crocea* var. *ilicifolia*) 1-2
Interior live oak (*Quercus wislizenii*) 1-2
Lemmon's willow (*Salix lemmonii*) 3
Littleleaf ceanothus (*Ceanothus parvifolius*) 2
Madrone (*Arbutus menziesii*) 3-5
Mariposa manzanita (*Arctostaphylos mariposa*) 4
Mountain pink currant (*Ribes navadense*) 3-5
Mountain whitethorn (*Ceanothus cordulatus*) 1-2
Mule fat (*Baccharis viminea*) 4-5
Narrow-leafed willow (*Salix exigua*) 3
Nevada ephedra (*Ephedra nevadensis*) 3-4
Nuttall willow (*Salix scouleriana*) 3
Oregon ash (*Fraxinus latifolia*) 4-5
Oregon oak (*Quercus garryana*) 2-3
Pacific dogwood (*Cornus nuttallii*) 3-4
Poison oak (*Toxicodendron diversilobum*) 2-3
Rabbitbrush (*Chrysothamnus viscidiflorus*) 3-4
Red flowering gooseberry (*Ribes sanguineum*) 3-5
Red shanks (*Adenostoma sparsifolium*) 4-5
Roundleaf rabbitbrush (*Chrysothamnus teretifolius*) 3-4
Rubber rabbitbrush (*Chrysothamnus nauseosus*) 3-4
Salal (*Gaultheria shallon*) 3-4
Shrubby cinquefoil (*Potentilla fruticosa*) 3
Sierra gooseberry (*Ribes roezlii*) 3-5
Silver sagebrush (*Artemisia cana* ssp. *bolanderi*) 3-4
Spiny hop-sage (*Grayia spinosa*) 2-3
Squaw bush (*Rhus trilobata*) 3-4
Tanoak (*Lithocarpus densiflorus*) 1-2
Thimbleberry (*Rubus parviflorus*) 3-4
Toyon (*Heteromeles arbutifolia*) 2-3
Twinberry (*Lonicera involucrata*) 2-3
Valley oak (*Quercus lobata*) 3-4
Valley willow (*Salix hindiana*) 3
Vine maple (*Acer circinatum*) 2-4
Wavyleaf ceanothus (*Ceanothus foliosus*) 1-2
Wax currant (*Ribes cereum*) 3-4
Western chokecherry (*Prunus virginiana* var. *demissa*) 1-2
Western hackberry (*Celtis douglasii*) 3-4
Western juniper (*Juniperus occidentalis*) 3-4
Western mountain-mahogany (*Cercocarpus betuloides*) 1
Western ninebark (*Physocarpus capitatus*) 4-5
Western redbud (*Cercis occidentalis*) 4-5
Western serviceberry (*Amelanchier alnifolia*) 2-3
White alder (*Aesculus rhombifolia*) 3-5
White-stemmed gooseberry (*Ribes inerme*) 3-5
Wild mock orange (*Philadelphus lewisii*) 3-4
Winter fat (*Eurotia lanata*) 2-3
Yellow willow (*Salix lasiandra*) 3

ARIZONA (Swank 1958, and Heffelfinger 2006)

TREES AND SHRUBS

Buckwheat (*Eriogonum* spp.)
Catclaw acacia (*Acacia greggii*)
Cliffrose (*Cowania* [= *Purshia*] *mexicana*)
Desert ceanothus (*Ceanothus greggii*)
Emory oak (*Quercus emoryi*)
Holly-leaf buckthorn (*Rhamnus crocea*)
Jojoba (*Simmondsia chinensis*)
Juniper (*Juniperus* spp.)
Kidney wood (*Eysenhardtia polystachya*)
Manzanita – point leaf manzanita (*Arctostaphylos pungens*)
Mountain mahogany (*Cercocarpus* spp.)
Rabbit brush (*Chrysothamnus* spp.)
Range ratany (*Krameria erecta*)
Sage (*Artemisia* spp.)
Skunk bush (*Rhus trilobata*)
Sugar sumac (*Rhus ovata*)
Turbinella oak (*Quercus turbinella*)
Wright’s silk-tassel (*Garrya wrightii*)

FORBS AND SUCCULENTS

Ayenia (*Ayenia filiformis*)
Barrel cactus (*Ferocactus* spp.)
Buckwheat (*Eriogonum* spp.)
Deer vetch (*Lotus* spp.)
Deer weed (*Porophyllum gracile*)
Metastelma (*Metastelma arizonicum*)
Penstemon (*Penstemon* spp.)
Prickly pear cactus (*Opuntia engelmannii*)
Spurge (*Euphorbia* spp.)

ACKNOWLEDGEMENTS

This document is the creation of the Mule Deer Working Group of the Western Association of Fish and Wildlife Agencies (WAFWA). The authors thank the members of the Mule Deer Working Group and WAFWA for their support and guidance, and especially Jim Heffelfinger for spearheading this effort. Additional thanks go out to the California Department of Fish and Game (CDFG) for allowing the authors the freedom to spend time on this document.

A number of people assisted in the development of the document in various ways. Reg Barrett (UC Berkeley), Mike Chapel and George Garcia (USFS), David Casady, Mary Meyer, Russ Mohr, Robert Schaefer, and Joel Trumbo (CDFG), Melvin George (University of California, Davis), Jason Lowe and Larry Saslaw (Bureau of Land Management), and Bill Tietje (UC Extension) provided valuable biological and editorial comments. Doug Bowman and Rod Goss (CDFG), and Bill Tietje (UC Extension) shared their literature sources, Jennapher Miller (CDFG) helped round up and copy publications, and Victoria Barr (CDFG) helped out with literature search. Their assistance is greatly appreciated.

Additional thanks to those who provided photos: Hazel Gordon (U.S. Forest Service Remote Sensing Lab); J.S. Peterson (USDA-NRCS PLANTS Database); Tulare County Public Library; Claudia Tyler (University of California, Santa Barbara); Henry Coletto (California Deer Association); and Lorna Bernard, Kim McKee, Phil Pridmore, Joel Trumbo, and Robert Vincik (CDFG).

Thanks to Steve Barton for his assistance in garnering financial support for this project. Thanks also to Rebecca Springer of Mad Dog Design and Butch Platt at Lithotech for their hard work in producing, publishing, and distributing this product.



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