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MANAGING RANGELANDS FOR WILDLIFE

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INTRODUCTION

Rangelands are plant communities dominated by grasses, forbs, and shrubs. Their primary use by humans worldwide is for livestock grazing, but these communities also are habitat for wildlife. Traditionally, wildlife-related concerns of range managers focused on predators of livestock and on wildlife species that are hunted. Today, managers are interested in biodiversity and a wide range of species. Management of public rangelands in the United States is constrained by federal and state laws, which require managers to address the impact of management activities on all wildlife.

The majority of rangelands used by wildlife in the United States are public lands administered by the U.S. Forest Service and Bureau of Land Management, both of which have multiple-use mandates. With existing laws such as the Endangered Species Act and Clean Water Act, and ecosystem management and ecosystem health policies of the major land management agencies in the United States, there is expanding need to address the ecology of rangelands as it relates to plants, soils, water, wildlife, and livestock.

Photographs, videos, Internet web sites, agenda-driven "science," opinion pieces, the growth of advocacy groups, legal challenges (and threat of legal challenges), and society's changing sentiments about use and condition of public rangelands have generated an abundance of confusion and uncertainty about rangeland management. What formerly was a field primarily limited to understanding livestock-big game species relationships is now open to examination of livestock impacts on all native flora and fauna, and the communities and ecosystems in which they exist.

The single greatest change influencing wildlife on western rangeland management during the 1990s has been the shift of concern from competition of livestock with big game such as deer (*Odocoileus* spp.) and elk (*Cervus elaphus*), to concern for all wildlife, and biodiversity in general. For terrestrial wildlife species, the fate of species such as the willow flycatcher (*Empidonax traillii*) and sage-grouse (*Centrocercus* spp.) now dominate livestock and wildlife issues in montane meadow-riparian systems and sagebrush (*Artemisia* spp.) steppe, respectively, in many areas of the western United States. In California for example, ungulates aren't mentioned in a recent decision to amend management of >1.7 million ha on 11 national forests (U.S. Department of Agriculture 2001). Aquatic, riparian, and meadow system rangeland management would, instead, be heavily influenced by habitat needs of the willow flycatcher, mountain yellow-legged frog (*Rana muscosa*), Yosemite toad (*Bufo canorus*), and great gray owl (*Strix nebulosa*).

Effectively managing rangelands for wildlife requires achieving a specified level of habitat structure as represented by vertical and canopy cover, food items as represented by species composition, and adequate water quality and availability. Additionally, where livestock grazing is involved, there is a need to understand and manage for interspecific and social interactions between livestock and wildlife, as well as strategies to mitigate adverse effects. These interactions may be in the form of behavioral avoidance or attraction, direct mortality caused by livestock, or habitat modifications, and indirect mortality caused by disease transmission. Wildlife-livestock interactions have greater application at a broad geographic scale rather than a site-specific study area.

Because most state and federal agencies have unique missions and mandates (Salwasser et al. 1987), management philosophies and on-the-ground techniques differ markedly among agencies. Philosophical differences can be further exacerbated when adjacent tracts of land, managed by different agencies, have their own unique designations (e.g., specially designated area). Specially designated areas come in a variety of shapes and sizes, but in the United States they are typically managed by one of a few federal agencies (e.g., U.S. Forest Service, Bureau of Land Management, National Park Service, or U.S. Fish and Wildlife Service), and include such areas as wilderness, special research areas, wildlife refuges, sanctuaries, or any other site where certain activities or management tools (e.g., aircraft, mechanical equipment) may be precluded. These areas are usually small relative to the management prescriptions of adjacent properties and, thus, exist as noncontiguous islands that must be managed differently from surrounding landscapes.

Because of the varied and unique challenges confronting managers in today's world, this chapter is not intended to be an all-encompassing treatise. Rather, it presents a discussion of selected issues and techniques in an effort to provide the reader with a general understanding and appreciation for the complexities associated with managing rangelands. An extensive literature review is included and the reader is encouraged to explore the vast quantity of information that has been published on this subject, some of which is also summarized elsewhere (e.g., Krausman 1996). It is our hope this chapter adequately (1) provides an overview of rangeland management to benefit wildlife species and natural communities, with an emphasis on western North America; (2) identifies some of the topical issues and primary rangeland systems of concern; and, (3) describes some of the techniques for accommodating wildlife and wildlife issues on rangelands.

PLANT SUCCESSION AND WILDLIFE MANAGEMENT GOALS FOR RANGELANDS

Plant succession is the gradual replacement of one assemblage of plant species with others through time until a relatively stable climax community is reached (Clements 1916). As each group of plant species is replaced, the value of the community as habitat to any particular species of wildlife changes. The result is a succession of wildlife species as plant communities and populations of primary consumers undergo successional changes altering the different trophic levels (Kie et al. 1994).

Range Condition and Wildlife Habitat

Only a portion of the vegetation biomass in a rangeland will provide adequate nutrition for an herbivore. As body size decreases, diet selectivity generally increases (Van Soest 1994); consequently, many wild herbivores (which tend to be smaller than domestic livestock) consume much less of the vegetation resource than livestock, particularly cattle. Furthermore, domestic livestock may consume a greater proportion of poorer-quality bulk forages because producers supplement diets of livestock to balance nutritional requirements for growth and reproduction at least for some portion of the year. Proper estimates of carrying capacity for wildlife on rangelands assume that all nutrients will be obtained from the range (Hobbs and Swift 1985).

Rangelands exist in many different successional stages and structural conditions because of the influence of fire, mechanical disturbance, herbicide treatment, and grazing by wild and domestic herbivores. Some plant communities respond to grazing in a predictable manner, depending on the plant species present (Dyksterhuis 1949). Some plant species are dominant in climax communities because they are superior competitors in the absence of disturbance. However, they begin to decline in vigor and abundance with increased grazing pressure (Dyksterhuis 1949). As they decline, other less palatable plants present at the climax stage become more abundant as competition is reduced. If grazing intensity is sufficiently heavy and occurs over a long period of time, new plant species, well adapted to heavy grazing, appear in the community. As a result, many exotic species of plants (e.g., spurges, thistles, brome grasses) become established and overall condition of the range is reduced.

In the past, rangelands have been managed on a concept of how close existing vegetation approximates a climax community using terms such as excellent, good, fair, and poor (Dyksterhuis 1949). This procedure cannot be used on seeded rangelands, however, or those dominated by introduced, naturalized plant species such as the annual grasslands of California (Smith 1978, 1988). Also, range condition terms including excellent, good, fair, and poor are defined in terms of providing forage for livestock–habitat is species specific and differs greatly among species. A site rated as poor may provide excellent habitat for wildlife adapted to early-seral vegetation (e.g., white-tailed deer [*Odocoileus virginianus*]), whereas a site rated as excellent on this scale (e.g., grassland) may not be used at all by that species. More appropriate terms for describing the condition of rangeland vegetation as they relate to wildlife needs are climax, late seral, mid-seral, and early seral (Pieper and Beck 1990).

Additional problems may arise when changes in livestock grazing practices do not immediately produce a change in rangeland vegetation. For example, some grassland sites in southeastern Arizona that had been converted to shrublands by heavy livestock grazing failed to revert to native grasses following 20 years without livestock (Valone et al. 2002). In contrast, other sites that were protected for up to 39 years exhibited an increase in grasses, suggesting that substantial time lags following protection from grazing were necessary (Valone et al. 2002).

Since 1990, range ecologists have been developing models of change in rangeland vegetation based on the concept of multiple steady states (Laycock 1991, 1994). These states are often portrayed as state-transition models (Westoby et al. 1989), wherein "states" are recognizable assemblages of species at a particular site that are stable over time. Such models are useful in understanding why some plant communities fail to respond immediately to changes in management practices. Parameterizing state-transition models, however, often requires large data sets on composition of rangeland vegetation collected over many years. If such data are available, state-transition models can provide more precise predictions about vegetation change (Allen-Diaz and Bartolome 1998) than the classical linear succession model developed by Clements (1916) and may be useful in restoring degraded rangelands (Chambers and Linnerooth 2001).

Models of Rangelands as Wildlife Habitat

The system of classifying wildlife habitats according to potential natural vegetation and seral stage for coniferous forests (Thomas 1979) also has been applied to rangeland vegetation in southeastern Oregon (Maser et al. 1984). Habitat data were assembled for 341 species of vertebrates assessing impacts of different range management activities on those species by equating plant communities and their structural conditions with habitat values for wildlife. The structural conditions were grass-forb, low shrub, tall shrub, tree, and tree-shrub. As a plant community progresses from grass-forb to tree-shrub conditions through succession, changes occur in environmental variables important to wildlife. For example, herbage production tends to be highest in grass-forb communities; browse production highest in low-shrub and high-shrub communities; and canopy closure, canopy volume, and structural diversity highest in tree and tree-shrub communities (Maser et al. 1984). Management actions such as brush and weed control, water development, prescribed burning, seeding and planting, and grazing also can result in changes in structural conditions (Maser et al. 1984).

Accounting for needs of large numbers of wildlife species makes land-use planning difficult. To simplify the process, wildlife can be grouped into life forms based on the relationship of the species to their habitats. In southeastern Oregon, 2 characteristics of each species (where it feeds and where it reproduces) were used to distinguish 16 life forms. For example, dark-eyed juncos (*Junco hyemalis*) and mule deer (*Odocoileus hemionus*) characterize those species that feed and reproduce on the ground. Other examples of such life forms include the long-toed salamander (*Ambystoma macrodactylum*) and western toad (*Bufo boreas*), which feed on the ground, in shrubs, or in trees, and reproduce in water (Maser et al. 1984).

Beyond generalized models of wildlife habitat associations, managers occasionally estimate nutritional carrying capacity of rangelands. Most models of range supply and animal demand sum the available nutrients supplied by forage in the habitat and then divide by the animal's nutritional requirements (Robbins 1973, Hobbs et al. 1982). However, these models are simple and fail to make predictions based on varying levels of nutritional quality required by individuals (e.g., pregnant or lactating females, breeding males, migrating adults, etc.) (Hobbs and Swift 1985). To avoid overestimating the number of animals that existing plant biomass can support, carrying capacity models should consider minimum dietary nutrient concentration (Hobbs and Swift 1985, Hanley and Rogers 1989).

The influence of grazing can also affect wildlife species richness, diversity, density, and abundance. Some conclusions, for example that grazing tends to increase abundance of common species but reduces the overall diversity of species (Bronham et al. 1999, Rambo and Faeth 1999), provide a community approach that may contribute to additional generalizations when other taxonomic groups are considered.

CONTEMPORARY ISSUES IN RANGELAND MANAGEMENT

Key Rangelands of Concern

Riparian, montane meadow, and aquatic habitats continue to remain a high priority for conservation and management on western rangelands. Minimizing soil erosion and maintaining or restoring water quality are paramount in sustaining these systems for the future. Meeting these 2 umbrella objectives may accommodate the needs of some wildlife species that inhabit these systems. Increasing concern now exists for other wildlife habitats that are rangelands. This interest has arisen largely because of growing concern for biological diversity, but also for specific wildlife species that are declining and/or are being petitioned for listing under the federal Endangered Species Act. While there are numerous other plant communities and wildlife habitats that comprise rangelands throughout the world, the following systems or habitats are currently of great issue on public rangelands in the western United States.

Sagebrush Steppe

Foremost of concern among rangeland habitats at present are the expanses of sagebrush/perennial bunchgrass range that dominate much of public land in the west (e.g., Paige and Ritter 1999). From a timing perspective, just as range livestock management has been challenged in the 1990s to work toward avoiding negative impacts to the riparian zone and to more effectively use upland range, livestock use of uplands has now come under scrutiny as well. Recent research indicating that sage-grouse are declining and that they nest most successfully when there is an herbaceous understory at least 18 cm in height (Sveum et al. 1998) has created an additional challenge for livestock managers on public lands–how to avoid impacting riparian zones while ensuring adequate herbaceous cover to meet the needs of at least one nesting species in sagebrush/grass communities. Use and management of fire, herbicides, proximity to urbanization and agriculture, use of off-road vehicles, and power lines also are contributing factors affecting quality of wildlife habitat on these rangelands.

Other habitats of concern geographically associated with sagebrush steppe are browse communities dominated by antelope bitterbrush (*Purshia tridentata*), mountain mahogany (*Cercocarpus* spp.), or saltbush (*Atriplex* spp.). Often, these communities serve as a seasonal range for wildlife, such as in winter, but are grazed by livestock in summer.

Desert

Concern about potential impacts to the desert tortoise (*Gopherus agassizii*) from livestock grazing and other uses prompted the Bureau of Land Management to recently issue a grazing decision to help protect this species in California desert systems. These systems are particularly susceptible to impacts of grazing because they require a long time for recovery of vegetation growth and vigor if they are able to recover at all (e.g., Krueger et al. 2002). Additionally, concern exists for native frogs relying on the rare and often heavily impacted riparian and aquatic areas of the desert southwest (Jennings and Hayes 1993).

Aspen

Habitats dominated by quaking aspen (*Populus tremuloides*) support a high diversity of wildlife on western ranges (Debyle 1985). These habitats also serve as valuable grazing (Sampson and Malmsten 1926) areas for livestock because of the proximity of food, cover, and usually water. There is growing concern that this community is on the decline in managed forests and ranges throughout the west because of lack of stand regeneration resulting from browsing by herbivores, fire suppression, and disease (e.g., California Department of Fish and Game 1998, Knight 2001). In turn, succession to dominance by conifers or by shrub communities (e.g., sagebrush) may result, thereby decreasing the value as wildlife habitat or as rangeland for domestic livestock grazing.

Integrating Wildlife Objectives and Range Livestock Management

Livestock grazing results in impacts on rangelands and wildlife species. It can either decrease or improve the conditions for wildlife depending on the species or community attribute of interest. A goal for public land resource managers is to identify the acceptable level of livestock impact, apply appropriate standards and guidelines, and then monitor their impacts. Implementing management decisions to meet wildlife species and habitat objectives, as well as broader goals of ecosystem health on public rangelands, often are emotionally charged socioeconomic (if not sociopolitical) decisions. These decisions often involve reducing use or eliminating livestock in the area of concern for a period of time to allow recovery. Numerous case studies and demonstration areas have illustrated that these actions are effective in some rangeland habitats such as riparian and aspen communities.

Within the field of wildlife–livestock interactions, addressing competition between livestock and large native herbivores was a primary emphasis on western public lands during the 1950s–1980s; during the 1990s the emphasis shifted to developing strategies to protect and restore riparian areas from overgrazing by livestock. Preventing livestock from negatively affecting riparian areas and achieving better distribution of grazing animals throughout upland areas were desired objectives. More recently (mid 1990s to present), there is evidence demonstrating the importance of standing herbaceous vegetation for nesting sage-grouse, a vegetation component that could be difficult to meet without significant change in grazing management strategies. Thus, more encompassing ecosystem-landscape-biodiversity concepts for management of rangelands have evolved in recent years. These have caused further shifts in the directions of many interest groups, government agencies, and academicians.

On public rangelands, recent objectives go beyond achieving and maintaining good to excellent range conditions for livestock and wildlife. Instead, objectives have broadened to conserve biodiversity, improve ecosystem health, and meet habitat requirements of federally listed, or potentially listed, wildlife. These objectives could be represented in many cases by increased herbaceous cover, soil maintenance, reduction in invasive species, and clean water. A more general approach would be to define positive ecological changes through rangeland management actions. Across landscapes, achieving such positive changes likely would satisfy most concerns for wildlife simply because such large-scale changes have been needed for decades.

Examples of species receiving substantial attention at present are the willow flycatcher and great gray owl, which rely on high quality mountain meadow-willow (*Salix* spp.) riparian complexes, and sage-grouse that rely on a combined habitat structure of sagebrush and standing herbaceous vegetation. The former 2 species continue to represent the needs and concerns related to grazing impacts on montane meadow and riparian areas, while the burgeoning sage-grouse issue has been labeled the range equivalent of the spotted owl (*Strix occidentalis*) issue because desired herbaceous cover levels will be difficult to achieve on grazed rangelands.

Investigations of Wildlife–Livestock Relationships

Studies of wildlife and livestock interactions are typically conducted to increase understanding of direct and indirect effects of livestock (as the manipulated perturbation or stressor) on a native species and/or its habitat. Much of the existing work was retrospective, rather than experimental, in that it was conducted with livestock as part of the system rather than as an introduced perturbation with treatments and controls. This difference also reflects one of the fundamental social debates regarding livestock on public lands in the United States: are humans, and the impacts they bring, part of the biotic community or ecosystem (e.g., Box 2000)?

Unquestionably, the science on wildlife-livestock relationships varies in terms of its rigor, thoroughness, results, and applicability to real systems. It indicates the presence of large, non-native herbivores is beneficial to some species and detrimental to others. Some initial investigations of wildlife–livestock relationships examined how cattle and mule deer distributed themselves throughout a common range (Julander and Robinette 1950, Julander 1955, Julander and Jeffery 1964) instead of manipulating cattle to measure how deer responded with and without cattle in the same area. Unfortunately, the ability to conduct replicated experiments at appropriate spatial and temporal scales to assess livestock grazing impacts on a wildlife population is logistically difficult. Conclusions from retrospective studies, that deer or other wildlife species preferred the steeper slopes while livestock preferred the flatter areas, became dogma in range science and suggested that a harmonious coexistence occurs without objective experimental evidence.

Perhaps the most acceptable generalization that can be made is that increasing the grazing level (often termed heavy, uncontrolled, excessive, or severe grazing) above some site-based threshold results in impacts that are not desirable to any interest. Further confounding our ability to generalize among wildlife–livestock investigations is that stocking rates, number of grazing levels (ungrazed or grazed in some studies; none, light, moderate, or heavy grazing in others), time of year grazed, vegetation communities, time lags to examine the response (e.g., Dobkin et al. 1998), and wildlife species of interest are not consistently applied or comparable.

During the 1950s–1980s, the primary wildlife emphasis on public rangelands was competition among large ungulates and livestock. Kie et al. (1994) summarized much of the knowledge in this area, and large herbivores continue to be of interest (e.g., Austin 2000). Rangeland science, however, has broadened to include examinations of livestock impacts on nontraditional wildlife and biodiversity. The body of literature examining the impacts of livestock on taxonomic groups such as amphibians (Jennings and Hayes 1993, Denton and Beebee 1996, Bull and Hayes 2000), reptiles (Bock et al. 1990, Bostick 1990, Kazmaier et al. 2001), birds (Dobkin et al. 1998, Goguen and Mathews 1998, Sveum et al. 1998, Belanger and Picard 1999, Beck and Mitchell 2000), small mammals (Hayward et al. 1997), and invertebrates (Rambo and Faeth 1999, Bronham et al. 1999) continues to grow, as does the number of review papers on livestock grazing impacts on biological diversity and ecosystems (Fleischner 1994, Belsky and Blumenthal 1997, Larsen et al. 1998, Belsky et al. 1999, Jones 2000).

Using livestock as a tool to manage wildlife habitat has been advocated for many years and examples of how this benefits one or more wildlife species do exist (Severson 1990). For example, Leopold et al. (1951) described the benefits of livestock in opening up paths for deer and other wildlife throughout willow-dominated montane meadow systems. Other examples describe the benefits of livestock in helping maintain or enhance vegetation species diversity (Rambo and Faeth 1999, Humphrey and Patterson 2000) or enhancing forage quality for other large herbivores (Clark et al. 2000). Whether the mechanical benefits, or more importantly, ecological benefits are needed every year is rarely, but should be, asked in the context of the entire system affected. Have Leopold et al.'s (1951) willow meadows been opened up "enough," or do they need to be continually grazed summer-long in high mountain ecosystems, such as those in the Sierra Nevada?

Accommodating Wildlife and Habitat Objectives on Rangelands

A common link between the wildlife biologist and the range manager is the vegetation community and the wildlife habitats represented. From a wildlife perspective, perhaps an efficient technique would be to develop habitat objectives such as percent cover, desired plant species composition, and structural conditions of vegetation that are desired for a species, a suite of species, or a community as a whole, rather than a targeted species population objective. This approach leaves the range or livestock manager with the task of identifying potential strategies for managing livestock to achieve wildlife objectives. Identifying how wildlife species respond to livestock grazing might be of value in assessing whether the overall effects of the grazing level are acceptable or not; this process for wildlife would be analogous to characterizing plant species as increasers, decreasers, or invaders in response to livestock grazing (e.g., Stoddart et al. 1975).

The concept of maintaining or enhancing biodiversity on multiple use rangelands should also capitalize on interjecting management diversity in terms of grazing systems used. Interjecting unpredictable changes in habitat structure by resting habitats that normally are grazed continuously adds to this kind of diversity. Additional study and information on how individual species respond would help distinguish between desirable and undesirable trends in species responses.

Historically, land use plans prepared by the U.S. Forest Service and Bureau of Land Management, in collaboration with state wildlife agencies, often developed population objectives for species such as deer, elk, or pronghorn (*Antilocapra americana*). A more measurable approach would involve moving from a specific population target and, instead, focusing on achieving a desired habitat condition across the landscape–at the scale of allotments, resource areas, districts, or entire national forests.

Role of Monitoring and Assessment in Addressing Wildlife–Livestock Issues

"The lack of biological data is, without a doubt, one of the greatest single factors in retarding development of a larger conservation program" (California Fish and Game 1926:28)

Because of the inherent controversy and often-polarized views of wildlife and livestock relationships, difficult management decisions are often tabled in the absence of adequate data on species trends or ecological condition of the system in question. Consequently, among the most valuable activities that can be undertaken for the benefit of wildlife on rangelands is the collection of scientifically defensible data on distribution, abundance, status, trend, and habitat relationships. Ranging from basic inventory, to implementation of long-term monitoring, and experimental investigation of cause-and-effect relationships, scientific data aid management decisions. A meaningful progression of actions to examine and understand wildlife and livestock relationships might involve assessing:

a) wildlife habitat requirements and preferences,
b) livestock use of habitats preferred by wildlife,
c) livestock and wildlife effects on those habitats and vegetation communities,
d) livestock effects on wildlife species, and
e) how wildlife responds over time.

The effects studied range from direct influences of livestock on species (e.g., trampling of frogs) to numerous indirect effects (e.g., effect on prey species or hiding cover). Far more likely than experimental manipulations, however, are study and characterization of habitat conditions including structure and composition of vegetation and how it influences species productivity and abundance. An adaptive element would include mechanisms to change livestock management strategies as information is gained or to test specific hypotheses with an experimental or manipulative approach.

MANAGING LIVESTOCK ON RANGELANDS

Heavy livestock grazing has been detrimental to many wildlife species in western North America (Smith 1977, Gallizioli 1979, Peek and Krausman 1996). Uncontrolled grazing clearly can affect the structure and composition of wildlife habitats. When adverse impacts occur, elimination of livestock can improve habitat conditions, although in many situations changes in livestock management practices can result in similar benefits. When properly managed, livestock grazing can be used to improve habitat for wildlife dependent on early-seral stage plant communities (Longhurst et al. 1976; Urness 1976, 1990; Kie and Loft 1990; Ohmart 1996). Information on relationships between livestock and wildlife is available in a variety of books, symposium proceedings, and review papers (Smith 1975, Townsend and Smith 1977, Schmidt and Gilbert 1978, DeGraaf 1980, Wallmo 1981, Peek and Dalke 1982, Thomas and Toweill 1982, Menke 1983, Severson and Medina 1983, Halls 1984, Severson 1990, Krausman 1996).

The relationship between grazing and wildlife habitat is complex. Livestock influence wildlife habitat by modifying plant biomass, species composition, and structural components such as vegetation height and cover. The impact of livestock grazing on wild ungulates can be classified as direct negative, indirect negative, operational, or beneficial (Mackie 1978, Wagner 1978). An example of a direct negative impact is competition between cattle and deer for a resource such as food or cover (Mackie 1978, Wagner 1978). Competition occurs when 2 organisms use a resource in short supply, or when one organism harms another in the process of seeking the resource (Birch 1957, Wagner 1978). Factors affecting impacts of livestock on wildlife include diet similarity, forage availability, animal distribution patterns, season of use, and behavioral interactions (Nelson and Burnell 1975, Severson and Medina 1983).

Indirect negative impacts of cattle grazing include: (1) gradual reductions in vigor of some plants and in amount and quality of forage produced; (2) elimination or reduction of the ability of forage plants to reproduce; (3) reduction or elimination of locally important cover types and replacement by less favorable types or communities, by direct actions over time or by changing the rate of natural succession; and (4) general alterations and reduction in the kinds, qualities, and amounts of preferred or otherwise important plants through selective grazing, browsing, or other activities (Mackie 1978).

Operational impacts are associated with livestock management (Mackie 1978) and include fence construction, water development (Evans and Kerbs 1977, Wilson 1977, Yoakum 1980), brush control (Holechek 1981), and disturbance associated with handling of livestock. For example, deer may temporarily move from pastures when cattle roundups occur (Hood and Inglis 1974, Rodgers et al. 1978).

Small mammals also influence rangeland vegetation (Moore and Reid 1951, Wood 1969, Batzli and Pitelka 1970, Turner et al. 1973, Borchert and Jain 1978) and compete with livestock for forage (Fitch and Bentley 1949, Howard et al. 1959). Because of their size and susceptibility to predation, rodents, lagomorphs, and other small mammals are highly dependent on the structure of vegetation in their habitats (Grant et al. 1982, Parmenter and MacMahon 1983, Bock et al. 1984). Grazing by livestock influences vegetation structure in those habitats and can significantly affect small mammal populations (Reynolds and Trost 1980).

Livestock grazing adversely affects many grassland birds, although moderate grazing can be neutral or beneficial to some species (Buttery and Shields 1975). Livestock management practices also can affect birds indirectly. For example, an organophosphate insecticide externally applied to cattle to control warbles may kill American magpies (*Pica hudsonia*) and cause secondary mortality among red-tailed hawks (*Buteo jamaicensis*) eating carcasses of the poisoned magpies (Henny et al. 1985).

Livestock management practices that can affect wildlife habitats and populations include livestock numbers, timing and duration of grazing, animal distribution, livestock types, and specialized grazing systems. These practices can be modified to reduce or eliminate adverse effects on wildlife and, at times, to enhance wildlife habitats (Severson 1990).

Livestock Numbers

Livestock numbers, or stocking rates, usually are specified by animal unit months (AUMs). One AUM is one animal unit (one mature cow with a calf, or equivalent) grazed for one month (Heady 1975:117). Livestock effects on wildlife become more pronounced with increasing stocking rates. A few cattle in a pasture may have no discernible effect on wildlife, but beyond some threshold wildlife response may increase rapidly. A range manager's traditional definition of proper grazing is based on maintaining a mix of plant species valuable as livestock forage and preventing soil erosion. Optimum livestock densities for wildlife may occur at different, and often lower, stocking rates. Thus, as with most effects of livestock on wildlife, responses can be difficult to interpret because of inherent site differences (Johnson 1982), and differences in grazing intensity, timing, and duration.

Timing and Duration of Grazing

Moderate cattle grazing of riparian areas in late fall in Colorado had no detectable impact on 6 species of birds dependent on the grass-herb-shrub layer for foraging, nesting, or both (Sedgwick and Knopf 1987). However, summer grazing can eliminate habitat specialists such as willow flycatchers, Lincoln's sparrows (*Melospiza lincolnii*), and white-crowned sparrows (*Zonotrichia leucophrys*) (Knopf et al. 1988).

The time of year that livestock are present can alter the composition of plant communities. Heavy grazing during a period of rapid growth of one plant species will favor other species that grow more rapidly at other times. For

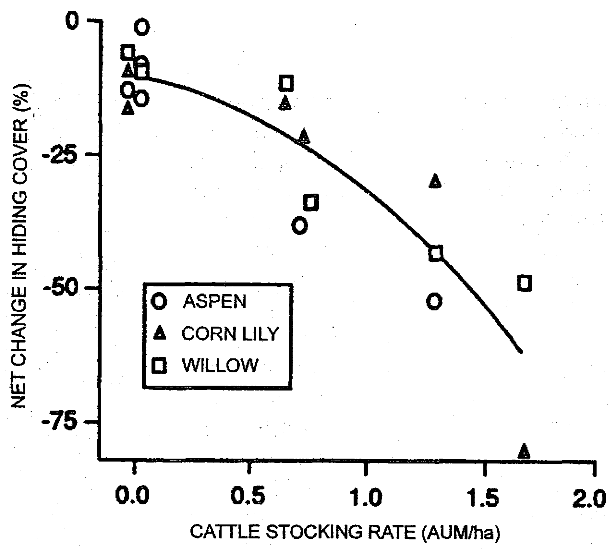

Fig. 1. Net change in mule deer hiding cover between 0 and 1 m in height from beginning of summer until mid-August as a function of cattle stocking rate (AUM/ha = animal unit months per hectare) (after Loft et al. 1987).

example, spring grazing of annual grasslands in California reduces grass cover and encourages growth of summer-maturing forbs such as turkey-mullein (*Eremocarpus setigerus*), the seeds of which are readily eaten by mourning doves (*Zenaida macroura*) (Kie 1988). Conversely, many wildlife species are most susceptible to livestock-induced changes in habitat during their reproductive seasons. Birds that nest on the ground or in shrubs can experience reproductive losses if their nests are trampled or otherwise destroyed by cattle. For example, willow flycatchers in California breed exclusively in riparian deciduous woodlands, and prefer willows as nesting substrate (Valentine et al. 1988). Flycatchers prefer to nest near the edges of willow clumps or along livestock trails (Valentine et al. 1988, Sanders and Flett 1989), where they are susceptible to physical disturbance. In one study, 4 of 20 willow flycatcher nests in a 4-year period were destroyed by cattle before young fledged, and 4 other nests were destroyed after young fledged (Valentine et al. 1988). When cattle stocking levels were reduced and 75% of the remaining cattle were confined to a fenced pasture away from willow flycatcher nest sites until 15 July, no willow flycatcher nests were lost (Valentine et al. 1988).

Excessive grazing can accelerate loss of hiding cover early in summer when mule deer fawns are young (Loft et al. 1987) (Fig. 1). These conflicts can be minimized or eliminated by delaying grazing until later in the year (Kie 1991).

Livestock Distribution

Livestock congregate around sources of water, supplemental feed, and mineral blocks; their impacts are most pronounced in those areas. Riparian zones, because of their abundant forage and water, are good examples of livestock concentration areas. Cross-fencing, developing alternative water sources, and providing feeding supplements on upland sites away from riparian areas more evenly distribute livestock. However, in certain situations, wildlife can benefit from patchy livestock distribution because some areas are lightly grazed. For example, many species of wildlife inhabit ecotonal areas ("edges"), and patchy distribution of livestock across home ranges of those species enables selection of grazed versus ungrazed patches to serve as foraging areas or refugia.

Types of Livestock

Effects of grazing on wildlife depend on the species of livestock. Differences in diet between cattle and domestic sheep dictate the effects they have on plant species composition. Also, cattle usually range within the confines of a fenced allotment, but sheep often are herded. Herded bands of sheep may have enhanced some habitats for mule deer in California (Longhurst et al. 1976) by repeated grazing and browsing that stimulated regrowth of more palatable shrubs and herbaceous vegetation. However, transmission of diseases from domestic sheep to mountain sheep (*Ovis canadensis*) may have eliminated many populations of the latter from California (Wehausen et al. 1987). As a result, professional organizations (e.g., Desert Bighorn Council Technical Staff 1990) and federal agencies have adopted management policies that reduce the probability of contact between domestic sheep and mountain sheep (U.S. Department of Interior 1992, U.S. Department of Interior and California Department of Fish and Game 2002).

Competition between pronghorn and domestic sheep is greater than between pronghorn and cattle because of increased overlap in forage preferences. On overgrazed sheep ranges, insufficient forb growth was available for pronghorn during the critical mid-winter period, and pronghorn die-offs were common (Buechner 1950). In general, domestic sheep are more likely than cattle to affect pronghorn adversely (Autenrieth 1978, Salwasser 1980, Yoakum 1980, Kindschy et al. 1982), and even moderate use by sheep during the winter dormant period can leave range units unsuitable for pronghorn until plant regrowth in spring (Clary and Beale 1983). Cows with calves often exhibit grazing patterns different from those of steers, and differences among breeds of cattle and sheep may occur.

Specialized Grazing Systems

Many specialized grazing systems exist, although most can be classified into 3 types (Heady 1975, Stoddart et al. 1975). *Continuous grazing* allows livestock to graze season-long or year-long. *Deferred grazing* refers to delaying or deferring grazing until after most of the range plants have set seed. Deferred grazing allows plants to grow, store carbohydrates, and reproduce at high rates. *Rotational grazing* involves dividing a range unit and rotating livestock through different pastures.

Combinations of periodic deferment and rotational grazing are called *deferred-rotation* grazing systems. A common one of these is the *4-pasture deferred-rotation* system, in which 4 range units or pastures are used, with 3 being grazed year-long and the fourth being deferred for 4 months. The pastures are then rotated each year.

Rest-rotation grazing is similar to a deferred-rotation system, but the period of rest consists of a full year or more. *Short-duration* grazing systems are similar to deferred-rotation systems, except that ≥8 small pastures are used, stocking rates are high in each pasture as it is used, but livestock are present for only short periods of time. Because timing of livestock grazing is critically important to most rangeland wildlife species, rotational grazing systems designed to consider wildlife have the greatest potential to reduce adverse effects.

Rest-rotation grazing may have the most potential to provide benefits to wildlife. This system often is economically disruptive because it foregoes livestock forage, but such losses may be compensated by benefits derived from wildlife-related recreation on public lands. For example, development of a rest-rotation grazing system in a single deer-hunting zone in California might specify that each range unit would be grazed only 1 of 3 years. The value of unused livestock forage, calculated on the basis of net economic value at $12.82 per AUM, would equal about $71,000 over each 3-year grazing cycle. However, increased deer populations and additional hunting opportunities would be valued at $6.5 million over the same period (Loomis et al. 1991).

Using Livestock to Manage Wildlife Habitat

In some situations, livestock grazing can be used to manage wildlife habitat (Longhurst et al. 1976, 1982; Holechek 1980, 1982; Urness 1982, 1990; Severson 1990). Livestock grazing has been applied to the management of habitat for species as diverse as mule deer (Smith et al. 1979, Willms et al. 1979, Reiner and Urness 1982), northern bobwhites (*Colinus virginianus*) (Moore and Terry 1979), and Canada geese (*Branta canadensis*) (Glass 1988). For example, cattle grazing in late winter and spring on foothill, annual grasslands in California encourages growth of forbs that are valuable to many wildlife species.

In other situations, application of prescribed grazing has met with mixed results. Too often, the intent of using livestock grazing has been to manage habitat for a single species, whereas entire communities actually are affected. Using livestock to maintain a plant community in an early seral stage often will benefit those wildlife species dependent on such habitat, while simultaneously impacting species associated with climax communities (Kie and Loft 1990).

The prescription, or strategy, for grazing is important. Maximizing benefits to wildlife from changes in grazing will involve reductions in livestock numbers and shortening grazing seasons compared to management plans designed to maximize livestock production. Livestock grazing by itself is neither good nor bad for wildlife, but depends on a variety of factors, including wildlife species of concern, livestock numbers, timing and duration of livestock grazing, livestock distribution, and kinds of livestock (Kie and Loft 1990). Wildlife and range managers might consider avoiding generalizations and evaluate the role of livestock on wildlife and their habitats independently for each species, grazing plan, and management situation.

MANAGING RANGELAND BY ANTHROPOGENIC MANIPULATION

Fire

Rangeland species evolved under the influence of fire and, hence, many are fire adapted. The natural occurrence of fire varies among regions as a result of fuels, topography, climate, and ignition source. The effect that fires have on landscapes is further dependent upon fire size, intensity, frequency, time of year during which they occur, and resulting burn patterns (Riggs et al. 1996). The interval at which fire occurs on a landscape varies as a function of active fire suppression, prior fire regime, plant community, and geographic location (Wright and Bailey 1982).

Effects of fire on wildlife populations may be positive or negative depending upon the temporal scale under consideration (short- vs. long-term), species involved, and characteristics of the burn. Fire effects on wildlife may be characterized as those directly affecting diet and those relating to habitat structure. Although effects on forage quality tend to be rather short-lived following a fire (Hobbs and Spowart 1984), structural changes may persist for decades, as is the case when forested and shrub stands are eliminated (Bunting 1986, Everett 1986, West and Yorks 2002). Effects of fire on bird and small mammal populations tend to be related to modifications of vegetation structure (Blake 1982, Bock and Bock 1983, Niemi and Probst 1990, Riggs et al. 1996).

Diet quality may be altered by fire as a result of alterations to floristic composition of plant communities, chemical composition of plant tissues, and structure of the plant canopy (Riggs et al. 1996). Although investigators have observed increases in both crude protein (Hobbs and Spowart 1984, Cook et al. 1994) and in vitro digestibility (Hobbs and Spowart 1984) in forages following fire, some of the greatest nutritional benefits may be derived through increases in foraging efficiency (Hobbs and Spowart 1984, Canon et al. 1987). Fire removes litter and dead standing herbage of low nutritional value (Van Soest 1994) enabling herbivores to more efficiently select nutritious plant material (Hobbs and Spowart 1984). The effects of burning on forage quality and stand composition and canopy among graminoids and herbaceous species persist for 1–3 years (Hobbs and Spowart 1984). Ultimately, effects on animal condition and productivity are most definitive; Svejcar (1989) noted increases in cattle performance when feeding on burned tallgass prairie.

Grazing prior to burning proportionately reduces nitrogen losses in forage (Hobbs et al. 1991), and grazing that precedes fire in tallgrass prairie reduces spatial variability of patches and improves animal performance (Hobbs et al. 1991). However, grazing of dry prairies following fire can inhibit forage recovery, and preference for burns by cattle may require adjustments to stocking rate (Erichsen-Arychuk et al. 2002).

Riggs et al. (1996) discussed the economics of prescribed fire and reported the larger the prescribed fire, the more cost effective, because fixed costs are applied over a greater area. They cautioned, however, that beneficial effects of fire treatments on wildlife habitats and populations should outweigh issues focusing too heavily on the amount of area burned. The role of fire varies from region to region and by ecosystem. Thus, prescriptions should be tailored to specific project areas.

Other Methods of Vegetation Manipulation

In addition to burning and grazing, vegetation manipulation of rangelands may occur through use of hand tools, mechanical equipment, and chemical spraying. The goals, as well as logistic and financial constraints, will affect which method is most suitable for any given area. Mechanical treatments are used to remove undesirable overstory species that inhibit growth of understory forage species (Bleich and Holl 1981, Fulbright and Guthery 1996, Holechek et al. 1998, Stephenson et al. 1998). Herbicide application may be used to control either unwanted brush or herbaceous species.

Although there may be social and legal constraints that affect use of herbicides, their application may be appropriate in some cases. In contrast to mechanical removal of vegetation, application of herbicides over large areas is typically less expensive and time consuming. Herbicides may be applied by hand, or with sprayers mounted to tractors or aircraft (Koerth 1996). The Herbicide Handbook Committee (1994, 1998) provides a thorough review of the types of chemicals available and their known effects.

Mechanical removal of brush from rangelands for the benefit of wildlife tends to be most successful when applied to patches intermixed in a landscape mosaic (Fulbright and Guthery 1996). In contrast, extensive clearing is detrimental to species dependent on woody plants. Major techniques for large scale brush removal include use of roller choppers, shredders (e.g., rotary axe), and crushers for top growth removal or, conversely, whole plant removal by root plowing, chaining and cabling, disking, and bulldozing and power grubbing (Bleich and Holl 1981, Fulbright and Guthery 1996). Additional considerations when selecting mechanical methods include topography, extent of resprouting, soil type, and size of the area to be treated (Holechek et al. 1998).

MANAGING RANGELAND RIPARIAN AREAS

Riparian areas are important habitats for terrestrial and aquatic wildlife (Carothers and Johnson 1975; Thomas et al. 1979*a,c*; Platts and Raleigh 1984; Skovlin 1984; Platts 1990). Their importance is a result of being obligate habitat for many aquatic species, of the uniqueness of their soil and vegetation complexes that produce diverse vegetation structure and concomitant diverse biological communities, and of their limited extent across a diversity of landscapes. Their value for a given species of wildlife is a function of water availability (for example, mule deer in the Sonoran Desert vs. wildlife in the Prairie Pothole Region of North America), life stages, animal movements, weather, and other factors.

Riparian vegetation and its structural arrangement are important for wildlife. Many vertebrate and invertebrate species depend directly or indirectly on riparian vegetation for food, cover, or other life requisites. Some wildlife use riparian zones disproportionately more than any other habitat. For example, of 363 terrestrial species in the Great Basin of southeastern Oregon, 288 depend directly on riparian zones or use them more than other habitats (Thomas et al. 1979*a*). Herpetofaunas also are strongly associated with riparian areas (Jones 1988). Riparian soils and substrates are important to amphibians, reptiles, and small mammals because these wildlife forms often inhabit subsurface environments. The temperate microclimate, availability of moisture, and greater biomass production of these areas provide for complex food webs.

The value of riparian areas to wildlife is only generally described, owing to the difficulty of long-term observations. Mule (Thomas et al. 1979*b*) and white-tailed (Compton et al. 1988) deer select woody riparian vegetation for cover and forage. Selected bird species have demonstrated an affinity for distinct layers of vegetation (Gutzwiller and Anderson 1986). Riparian zones provide migration routes for birds, bats, deer, and elk (Wauer 1977) and are frequently used by deer and elk as travel corridors between high-elevation summer ranges and low-elevation winter ranges. Moreover, riparian habitats are strongly selected by mountain lions (*Puma concolor*) in some areas (Dickson and Beier 2002).

Riparian habitats are of further importance because they comprise only about 1% of the landscape in the United States (Knopf 1988). Further, >70% of the original riparian habitats in the United States have been lost through a variety of land use practices (Megahan and King 1985). Barclay (1978) reported that natural riparian habitats within the Oklahoma grasslands have nearly vanished, and channelization was responsible for conversion of 86% of bottomland forests to other land uses. In the southwestern United States, many historically perennial streams are largely ephemeral watercourses today (Johnson et al. 1989).

Central to development of management strategies for riparian areas are: (1) an understanding of what constitutes a riparian area, (2) their internal functions and processes, (3) the influences on riparian ecosystems, and (4) their importance to wildlife. Elmore (1989) argued that a fundamental understanding of the functioning of riparian ecosystems was initially necessary to evaluate benefits and incorporate management actions into land use plans.

Rivers and streams transport water and sediments (Jensen and Platts 1987). Thus, riparian habitats are unique products derived from the dynamic processes that a given stream produces and are influenced by the interactions of climate, geology, geomorphology, hydrology, pedogenesis, and chemical and biological processes. Little information is available, however, on wildlife/riparian interactions. As a result, wildlife management considerations frequently are excluded from land use plans (Dwyer et al. 1984, Dickson and Huntley 1987). Substantial work has been done on riverine/riparian dynamics (reviewed by Curtis and Ripley 1975; Thomas et al. 1979*a,b*; Brinson et al. 1981, Kauffman and Krueger 1984; Platts and Raleigh 1984; Skovlin 1984; Warner and Hendrix 1984; DeBano and Schmidt 1989; Platts 1990).

Value, Structure, and Function of Riparian Areas

Several authors have proposed riparian terminology; both Swanson et al. (1982) and Johnson and Lowe (1985) suggest that disparity exists among users. They defined riparian areas as the sum of the terrestrial and aquatic components characterized by: (1) presence of permanent or ephemeral surface or subsurface water, (2) water flowing through channels defined by the local physiography, and (3) the presence of obligate, occasionally facultative, plants requiring readily available water and rooted in aquatic soils derived from alluvium. Riparian ecosystems usually occur as an ecotone between aquatic and upland ecosystems, and have distinct and variable vegetation, soil, and water characteristics. Typically, riparian areas are viewed as riverine habitats with perennial surface flows and associated plants and soils. However, surface flows may be ephemeral or periodic, as in desert washes or arroyos of the southwestern United States.

Riparian vegetation typically functions to allow necessary sediment transport and natural erosional processes. It also effectively reduces accelerated erosion that could result in loss of riparian habitats (Miller 1987). Riparian trees supply large organic debris and function to influence the physical (morphology), chemical (nutrient cycling), and biological (flora and fauna) components of the system (Bisson et al. 1987). Changes in stream channel structure and habitat diversity can occur when large organic debris is removed (Bilby 1984). Structural diversity, an important feature of riparian vegetation (Jain 1976, Anderson and Ohmart 1977), is affected by consequences of natural or human-caused habitat disruption.

Management Problems and Strategies

Management of riparian habitats is important because of the role of these ecosystems in water quality and nutrient recycling (Stednick 1988), and because riparian vegetation is considered to be the most sensitive and productive North American wildlife habitat (Carothers and Johnson 1975). Indeed, no other habitat in North America is as important to noncolonial nesting birds; riparian areas are equally important to other terrestrial vertebrates (Szaro et al. 1985).

Riparian zones are easily affected by natural or induced changes on their watersheds, including grazing (Kauffman and Krueger 1984, Skovlin 1984, Chaney et al. 1990). Moreover, problems seemingly related to riparian habitats alone cannot be resolved by considering only that habitat. As a result, management of riparian areas should be considered both onsite (within the riparian zone) and offsite (outside the riparian zone), which accounts for all adjacent uplands that exert influence over the watershed. Onsite activities such as grazing management and vegetation treatments are performed within riparian habitats; offsite activities include logging, road construction, and slash burning. Management activities outside the riparian zone may change the quantity and quality of water entering the riparian area (Stednick 1988). A variety of range management options are available for sustaining health of riparian habitats including complete protection (Stromberg and Patten 1988), multiple-use approaches, and exclusive use.

Livestock grazing is perhaps the greatest biological threat to riparian habitats in the western United States, given that about 91% of the total rangeland is grazed (Chaney et al. 1990). Improper livestock grazing affects all 4 components of the riverine/riparian system–channel, stream banks, water column, and vegetation (Platts 1990). Livestock grazing problems usually are the result of improper distribution of cows and not simply too many (Severson and Medina 1983). Concentrated livestock use results in sparse tree or shrub stands of low vigor, generally with substantial dead material on the ground, a tight, sod-bound soil, and lack of tree or shrub reproduction. Damage occurs in several ways. One is compaction of soil, which reduces moisture infiltration and increases runoff. Another is constant removal of herbage, which allows soil temperatures to rise and increases evaporation from the soil surface. A third is physical damage to the trees or shrubs by rubbing, trampling, and browsing (Severson and Boldt 1978). The primary method for resolving overuse of riparian areas has been modified grazing strategies, which have met with mixed results (Dwyer et al. 1984, Skovlin 1984, Chaney et al. 1990).

Isolated case studies have demonstrated that revised grazing management improved conditions, but also that condition of riparian habitats continues to decline (U.S. General Accounting Office 1988). Myers (1989) reported 74% of the grazing systems evaluated failed to positively improve rangeland health within 20 years; however, riparian vegetation usually improves from grazing relief within 4–6 years, depending on severity of use (Platts and Nelson 1989). Areas with severe overuse require greater periods of time (>15 years) for native species such as sedges (Cyperaceae) to displace species adapted to overuse (Elmore and Beschta 1987).

Conventional grazing systems (Heady 1975) were developed with consideration only for production and maintenance of forage plants, primarily graminoids. Application of these systems to maintain woody streamside vegetation and stream bank integrity likely will not be satisfactory, given the ecophysiology of shrubs and trees. Platts (1990: 6) provided an excellent description of grazing strategies designed to complement restoration objectives with livestock management, and suggested that, "the solution is to identify and develop compatible grazing methods," given our state of knowledge of the functions of riparian systems. Indeed, at least one grazing strategy is available that would provide riparian areas with the necessary rest or protection needed to restore, maintain, or enhance their productivity. The least acceptable option is "no use" by ungulates and this option may be attractive in situations where restoration is a major objective of overall riparian management. Another recommendation is to fence critical reaches of riparian habitats in an effort to maintain the integrity of the streamside zone (Platts 1990).

A good management strategy for sustaining rangeland riparian areas will: (1) maintain the productivity of the vegetation (e.g., structure, species composition), (2) maintain the integrity of stream dynamics (e.g., channel and bank stability), and (3) recognize that several factors (e.g., soils, vegetation, hydrology, and animals) interact to maintain a dynamic equilibrium within the riparian zone. Successful management of riparian areas is dependent on application of knowledge from the physical sciences, such as hydrology and geomorphology, combined with an aggressive program that provides adequate protection to the structure, composition, and diversity of vegetation in such areas.

DEVELOPING RANGELAND WATER SOURCES

Increasing the amount of water available to wildlife has been used to enhance habitat for species inhabiting arid rangelands (Kie et al. 1994). Techniques include improvement of natural springs, seeps, and waterholes, and construction of artificial devices to capture and store rainfall (Tsukamoto and Stiver 1990, Young et al. 1995, Arizona State University College of Law 1997). Recently, development of rangeland water sources has been questioned (Broyles 1995) and become controversial (e.g., Broyles and Cutler 1999, Rosenstock et al. 2001) and will require substantial effort to resolve (Rosenstock et al. 1999).

Many methods have been used to make subsurface water available to wildlife including manual techniques, explosives, prescribed fire, and chemicals. Recently, horizontal well technology has been applied to development of springs and seeps for wildlife (Kie et al. 1994). Handwork, although time consuming and costly, may be the most practical way to accomplish some types of developments (Weaver et al. 1959). Helicopters can be used to transport personnel and hand tools into remote sites, thereby allowing development of those sites (Bleich 1983).

Water sources can be improved with explosives (Weaver et al. 1959), but caution is necessary to ensure that water-yielding subsurface formations are not altered drastically and water flow is not interrupted. When such damage does occur, it is usually the result of a heavy charge opening a crack that allows water to escape. Explosives should be used only on marginal seeps where sufficient water is not immediately available and where it can be used safely. Explosives also are useful in clearing channels to allow storm flows to bypass a spring, or to lay pipe to be used for gravity flow of water to a basin (Weaver et al. 1959).

Prescribed fire can be used to remove phreatophytic vegetation, resulting in a decrease in the transpiration of subsurface water and increased surface flows (Biswell and Schultz 1958, Weaver et al. 1959). Use of prescribed fire requires extreme caution and periodic reburning may be necessary to maintain surface flows. However, the importance of small patches of desert riparian vegetation to a multitude of species makes any substantial reduction in the occurrence of such vegetation undesirable (Bleich 1992). Where prescribed fire can be used to temporarily clear a spring site or seep so that other development may proceed, its use may be desirable, but its role is limited.

Herbicides increase surface flows by eliminating vegetation responsible for evapotranspiration of subsurface water. They can be particularly useful where water is limited; loss of cover or shade may be more than offset by making a permanent water supply available to wildlife (Weaver et al. 1959). The limited distribution of native, riparian vegetation in arid areas makes widespread use of herbicides undesirable. Herbicides can, however, be used to control saltcedar or tamarisk (*Tamarix* spp.) at desert water sources (Sanchez 1975). Control of this exotic species can be successfully accomplished on a small scale by hand cutting and herbicide application (Sanchez 1975, Neill 1990).

Development of Springs

Development of springs should: (1) provide at least one escape route for wildlife to and from the site that takes advantage of the natural terrain and vegetation; (2) provide an alternate escape route where feasible; (3) protect water developments from livestock while allowing access for wildlife; (4) reduce the possibility of wildlife drowning by providing gentle basin slopes or ramps in tanks; (5) maintain or provide adequate natural cover, plantings, or brush piles around the watering area; (6) provide, where applicable, a sign to inform the public of the purpose of the development; (7) provide for development of sufficient capacity to supply water whenever it is needed for wild animals; and (8) provide livestock and public access to water outside the protected water development (Yoakum et al. 1980, Bleich 1992, Kie et al. 1994). If shy animals are involved, water for human consumption can be piped some distance from the wildlife water source. For example, sustained camping should be discouraged within a 1-km radius of water used by mountain sheep.

Ramps or walk-in wells offer a simple and inexpensive method of making water available to wildlife (Weaver et al. 1959). Unless the ramp is cut through rock, however, the sides must be boarded to keep material from sloughing into the excavation. Ramps should be a minimum of 1 m wide to allow large animals to enter and exit easily. Ramps are also important for escape in other types of water developments such as livestock troughs (Wilson 1977) and guzzlers (Andrew et al. 2001).

Construction of small basins or pools at a water source is an effective way to conserve water and make it readily available to wildlife. Basins may be constructed with rock, cement, or masonry, or they may be gouged from solid rock near the source when small seeps originate in a rock stratum. A simple basin, constructed with hand tools, can be chiseled into solid rock and will effectively store water for years. Where appropriate, power tools and explosives may be used to create larger storage basins. When explosives are used, care must be taken not to damage the source of the water, or the rock face so that it cannot be modified to store water. A major advantage of this type of development is that they are nearly indestructible.

Rock basins can be enlarged with cement and rocks or masonry materials. Similarly, these materials may be used to construct diversions to protect a basin from debris caused by storm flows, or to create an artificial basin at a location where the development of a solid rock basin is impractical. Special masonry techniques may be necessary to ensure a bond between the mortar and rock (Gray 1974).

Many springs and seeps occur in canyon bottoms. Even when developed, such springs are subject to damage by water from storms. A method of development that often is satisfactory is to bury a length of perforated asphalt or plastic pipe packed in gravel, at the spring source, and pipe the water to a basin or trough away from the canyon bottom and danger of flooding. Placing large rocks over a source after it has been developed and capping the development with concrete increases protection. Alternatively, a redwood spring box may be installed at a water source allowing access for maintenance with water piped to a trough in a safe location.

Plastic pipe is a good choice for use because it is lightweight, durable, and not subject to rust or corrosion; further, repairs are easily accomplished. Any type of pipe should be buried sufficiently deep to prevent freezing, trampling by livestock and wild ungulates, or damage from floods. A continuous downhill grade will help prevent air locks from developing in the pipe and ensure constant flow of water. When water is to be piped away from excavated springs, a trough constructed of concrete or masonry is preferred because it will not rust. If the trough poses a potential hazard for small animals and birds, a ramp should be installed to facilitate access to the water (Bond 1947).

Horizontal Wells

Traditional techniques used to develop springs and seeps have several disadvantages: (1) flow of water from the source cannot be controlled, (2) variable flow may be inadequate to generate enough water to create a surface source, and (3) exposed spring water and the source may be susceptible to contamination (Welchert and Freeman 1973). Horizontal well technology can overcome some of these disadvantages (Coombes and Bleich 1979; Bleich 1982, 1990; Bleich et al. 1982*a*).

Horizontal wells have several advantages: (1) success rate, particularly in arid regions where historical sources may have failed, is high; (2) amount of water can be readily controlled, thus reducing waste; (3) the area is not readily subject to contamination; (4) they are relatively inexpensive to develop; and (5) maintenance requirements are low. Horizontal wells also have disadvantages: (1) the initial cost of the equipment necessary to construct them can be high (although private contractors can do the work with their own equipment), (2) transporting the necessary equipment to remote sites can be difficult, and (3) some horizontal wells require a vacuum relief valve to prevent air locks from interrupting the flow.

Site selection is the most important and difficult step in development of a horizontal well. Several factors, including presence of historical springs and seeps, distribution of phreatophytes, and presence of an appropriate geological formation, must be evaluated (Welchert and Freeman 1973). Dike formations (a tilted, impervious formation that forms a natural barrier to an aquifer) and the contact formation (a perched water table over an impervious material) are both suitable for horizontal well development. Developing a dike formation requires the impervious barrier be penetrated to tap the stored water (Fig. 2). A contact formation is developed by penetrating at or above a seep area at the boundary of an impervious layer (Fig. 2).

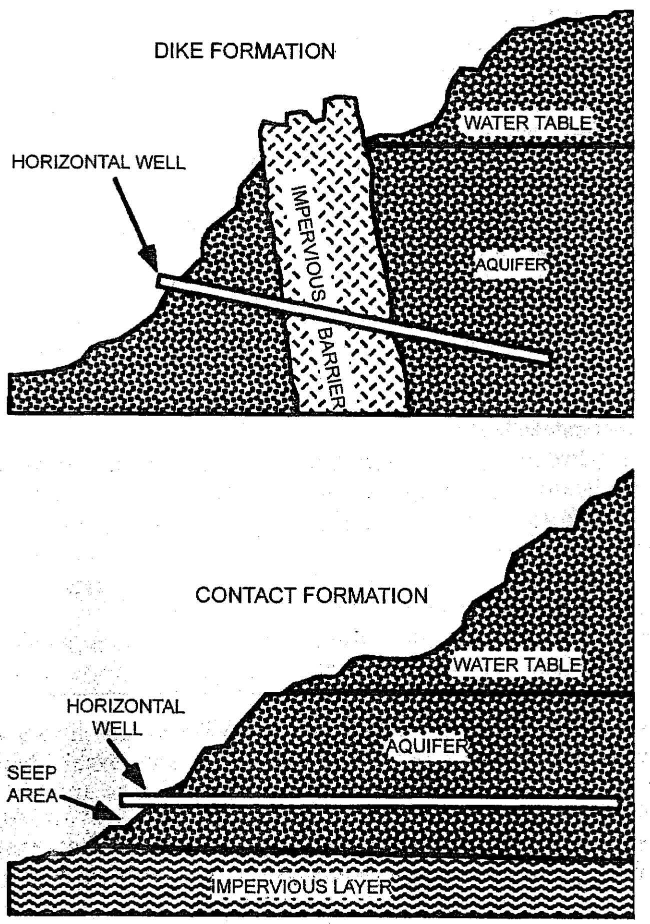

Fig. 2. Horizontal wells can be developed in dike or contact formations. The position of the well relative to the aquifer and impervious barrier is critically important to the success of the well (after Welchert and Freeman 1973).

Tinajas

Tinajas are rock tanks created by erosion that hold water. In some desert mountain ranges, tinajas may provide the only sources of water for wildlife. The capacity of tinajas can range from a few liters to more than 100,000 L of water.

Several techniques are available to increase storage capacity of tinajas. Sunshades can be used to reduce evaporation of water (Halloran 1949; Halloran and Deming 1956, 1958; Weaver et al. 1959). Shades can be constructed by anchoring eyebolts into the canyon walls, installing cables, and attaching shading material such as sheet metal to the cables (Weaver et al. 1959). In Arizona, sunshades have been built with a framework of 5-cm pipe placed into holes drilled into bedrock, with shading material then attached to the framework (Werner 1984).

Some tinajas can be deepened or enlarged with explosives (Halloran 1949, Weaver et al. 1959), but use of this method risks damage to the tinaja. A safer, and potentially more effective, method involves constructing an impervious dam on the downstream side, combined with a pervious structure to divert debris around the tinajas but allowing water to flow into them (Werner 1984). Deep, steep-sided tinajas often pose special problems for wildlife, because individuals can become trapped when water levels are low. Pneumatic equipment or explosives can be used to chisel or blast access ramps in such situations (Halloran 1949). Mensch (1969) used explosives to create an escape ramp at a natural tinaja in which 34 mountain sheep had died within a 2-year period.

Sand Dams

Some of the earliest techniques designed to increase water availability in arid regions involved construction of sand dams or sand tanks (Sykes 1937; Halloran 1949; Halloran and Deming 1956, 1958). These devices originally were constructed by placing a concrete dam across a narrow canyon. One or more pipes that could be capped to prevent water from draining penetrated the dam. The dammed area then filled with sand and gravel washed in by floods. Water soaks into the sand and gravel, and is stored, protected from excessive evaporation (National Academy of Sciences 1974).

Sand dams must be securely anchored in bedrock, and the design and construction of the dam may be the most important aspect of the entire system (Bleich and Weaver 1983). Because seepage at the bedrock interface could be a significant source of water loss, Bleich and Weaver (1983) emphasized that techniques used must result in an efficient bond between cement and bedrock (Gray 1974).

Storage volume of sand dams can be increased in a variety of ways (Sivils and Brock 1981, Bleich and Weaver 1983), but dams should not be too large. Compounds such as calcium aluminate can be added to the concrete to decrease set-up time (Gray 1974); however, sand dams should be no more than 12 m long and 3 m high (Halloran and Deming 1956, 1958). Water stored behind sand dams can be piped to a trough some distance from the dam (Sivils and Brock 1981), or used to flood natural or constructed potholes downstream.

Because precipitation in arid regions often occurs as violent thunderstorms, washes and canyons often flow large amounts of water over a short period of time. These brief flows may not allow sufficient time for storm water to saturate areas behind sand dams, especially if the underground storage capability has been enhanced (Sivils and Brock 1981, Bleich and Weaver 1983). Rock-filled baskets or gabions anchored into bedrock can be placed across a wash or canyon perpendicular to the direction of flow to slow water velocity. Such structures also raise and widen the wash.

Reservoirs and Small Ponds

A reservoir consists of open water impounded behind a dam. Reservoirs can be constructed by building a dam directly across a drainage or by enclosing a depression on one side of a drainage and constructing a ditch to divert water into the resulting basin (Yoakum et al. 1980). They also recommended that reservoirs be designed to provide maximum storage with minimum surface area to reduce evaporation. Major points to consider in selection of reservoir sites include: (1) suitability of soils for dams (clays with a fair proportion of sand and gravel, i.e., 1 part clay to 2–3 parts grit); (2) the watershed area above the dam should be sufficiently large to provide water to fill the reservoir, but not so large that excessive flows will damage the spillway or wash out the dam; (3) channel width and depth with a bottom easily made watertight and channel grade immediately above the dam as flat as possible; (4) easy access for wildlife to the water; and (5) an adequate spillway naturally incorporated into the development.

The base thickness of the dam must be equal to or greater than 4.5 times the height plus the crest thickness. Slopes of the dam should be 2.5:1 on the upstream face and 2:1 on the downstream face. Minimum width of the top of all dams should be 3 m. Fill of the dam should be at least 10% higher than the required height to allow for settling. Freeboard (depth from the top of the dam to the high-water mark when the spillway is carrying the estimated peak runoff) should not be less than 60 cm, and the spillway should be designed to handle double the largest expected volume of runoff. A natural spillway is preferred and it should have a broad, relatively flat cross section. Water should be taken out through the spillway well above the fill, and then re-enter the main channel some distance downstream. Spillways should be wide, flat-bottomed, and protected by riprapping, or by facing with rocks. The entrance should be wide and smooth, and the grade of the spillway channel should be low so the water will flow through without cutting (Hamilton and Jepson 1940).

New reservoirs usually do not hold water satisfactorily for several months. Bentonite spread over the bottom and sides of the basin and face of the dam will help seal the impoundment. The basin also can be lined with polyethylene or another appropriate material, with 15–30 cm of dirt rolled evenly over the top (U.S. Department of Interior 1966). Other artificial materials such as Hypalon® (Water Saver Company, Denver, Colorado, USA) are superior to polyethylene, because of their strength and resistance to ultraviolet radiation. These liners can be custom made for reservoirs of different sizes.

Dugouts

Large earthen catchment basins built to collect water for livestock were commonly called charcos by early settlers along the Mexican border, and dugouts by pioneers in other areas (Yoakum et al. 1980). Dugouts can be placed in almost any type of topography, but are most common in areas of comparatively flat, well-drained terrain. Such areas facilitate maximum storage with minimum excavation. Dugouts can be small, rectangular excavations (Fig. 3). All sides should be sloped sufficiently to prevent sloughing (usually $\leq 2:1$) and one or more relatively flat side slopes ($\leq 4:1$) should be provided to facilitate access for large mammals (U.S. Department of Interior 1964).

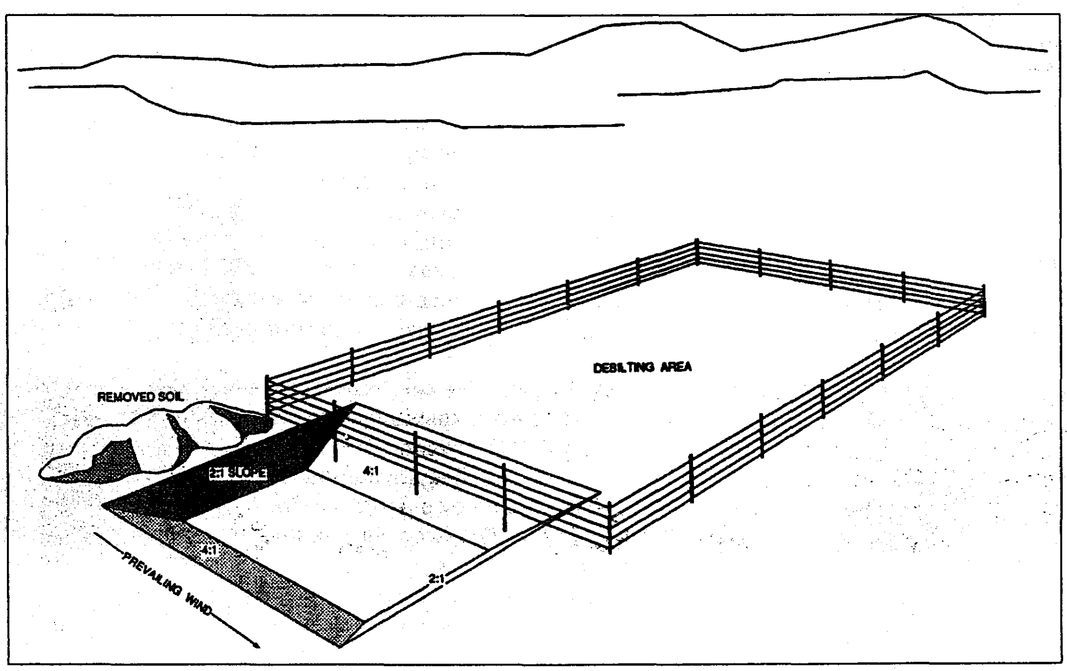

Fig. 3. Dugouts, also known as charcos, can be constructed to provide water for wildlife on rangelands (after Yoakum et al. 1980, Kindschy et al. 1982).

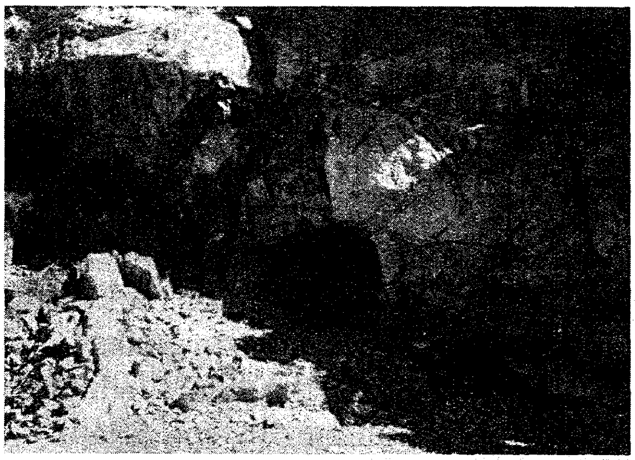

Fig. 4. An adit is a short tunnel that has been blasted into solid rock to store water for wildlife. The entrance to the adit must be at the same elevation as the bottom of the wash in which it is located.

Adits

Adits (Fig. 4) are short, dead end tunnels that extend into solid rock constructed with a downward sloping floor to allow access by wildlife (Halloran and Deming 1956, 1958). Adits have been constructed in Arizona and other western states, primarily to benefit mountain sheep (Parry 1972, Weaver 1973).

Personnel skilled in hard rock blasting techniques should be used to construct adits. These water storage depots should have openings at least 2×3 m and be at least 4–5 m in length. The water storage depth should be at least 4 m to ensure a dependable water supply (Halloran and Deming 1956, 1958). Commercial masonry sealers should be used to prevent seepage of water through rock fractures (Halloran and Deming 1956, 1958; Gray 1974; Werner 1984).

Because the opening of an adit must be approximately the same elevation of the wash in which it is placed, it may be necessary to construct a diversion that allows flood waters to enter, yet causes debris, sand, and boulders to bypass the adit. Boulders placed on the upstream sides of adits can be used for this purpose (Halloran and Deming 1956, 1958). Another effective, but simple, technique involves construction of a rock gabion (Werner 1984).

Adits also can be designed to store water from a natural source, such as a seasonal or permanent spring (Werner 1984), and water sometimes can be diverted into adits from natural slick-rock aprons above the site. Adits also can be used to store water that normally would be unavailable, and water can be pumped from the adit into a nearby tinaja (Werner 1984). In such instances, the adit should be covered to reduce evaporation. Shade structures have been used to reduce evaporation at adits in which stored water is directly available to wildlife (Halloran and Deming 1956, 1958).

Guzzlers

Guzzlers are permanent, self-filling, structures that collect and store rainwater and make it directly available to wildlife. Guzzlers can be constructed to provide water for small animals only, or for animals of all sizes.

Fig. 5. Contemporary underground guzzlers (Lesicka and Hervert 1995) store up to 40,000 L of water and have no moving parts. Wildlife walk down a ramp to reach stored water.

Several techniques can be used to collect water for guzzlers. Aprons that collect rainfall can be of manufactured or natural materials, including concrete or sheet metal, but asphalted, oiled, waxed, or otherwise treated soil aprons can be used (Glading 1947, Fink et al. 1973, Rauzi et al. 1973, Myers and Frasier 1974, Frasier et al. 1979, Johnson and Jacobs 1986, Rice 1990, Lesicka and Hervert 1995).

Guzzlers useful for wildlife generally store water in underground tanks, and wildlife walk a ramp to enter the guzzler to drink (Halloran and Deming 1956, 1958; Lesicka and Hervert 1995) (Fig. 5). However, water can also be stored in underground or aboveground concrete, plastic, metal, or fiberglass tanks (Garton 1956*a*,*b*; Roberts 1977; Bleich et al. 1982*b*; Remington et al. 1984; Werner 1984; Bardwell 1990; Bleich and Pauli 1990; deVos and Clarkson 1990; Gunn 1990; Lesicka and Hervert 1995). Aboveground tanks (Fig. 6) usually have a float-valve to regulate water at a drinking trough away from the water storage tanks (Roberts 1977, Werner 1984, Bleich and Pauli 1990). Underground tanks generally have no moving parts (Lesicka and Hervert 1995) and are not as subject to mechanical failures as are designs that incorporate a float valve. Moreover, guzzlers that store water for large mammals below the surface of the ground are nearly undetectable by humans more than a few meters from them (Fig. 7); current designs (Lesicka and Hervert 1995) present little risk of drowning to native vertebrates, including desert tortoise (Andrew et al. 2001).

The most important step in installation of a guzzler is locating a suitable site. A guzzler should not be placed in a wash or gully where it may collect silt or sand or be damaged by floodwaters; many guzzlers have been installed in areas lacking critical habitat components (Lewis 1973). When constructing a guzzler for small animals, Yoakum et al. (1980) recommended that: (1) size of the water-collecting apron be proportioned so the storage tank will need no water source other than rainfall to fill it, (2) a site should be chosen where digging is comparatively easy, and (3) the tank should be placed with its open end away from the

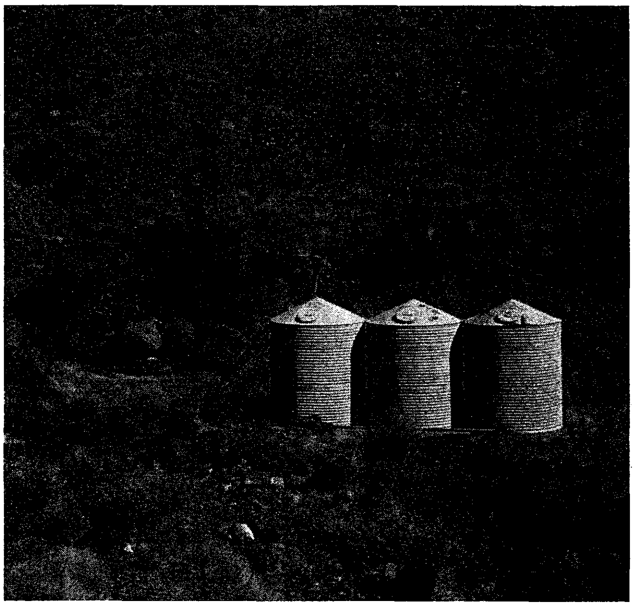

Fig. 6. Guzzlers constructed with above ground storage tanks generally have a float valve to control the water level in the drinking trough. Guzzlers of this type store up to 10,000 L of water for use by large mammals in the Mojave Desert, California.

prevailing wind and, if possible, facing in a northerly direction to reduce water temperature, evaporation, and growth of algae.

Tanks usually are made of concrete or plastic. Occasionally, steel tanks are used as are used heavy equipment tires (Elderkin and Morris 1989, Morris and Elderkin 1990). The plastic guzzler is a prefabricated tank constructed of fiberglass impregnated with plastic resin. Only washed gravel aggregates should be used for construction of concrete tanks, or the concrete may disintegrate in several years. Tanks made of steel are used for guzzlers in some areas and give satisfactory service. Use of tanks constructed of other artificial materials is relatively new.

Concrete sealed with bitumul, galvanized metal sheet roofing, glass mat and bitumul, rubber or plastic sheets,

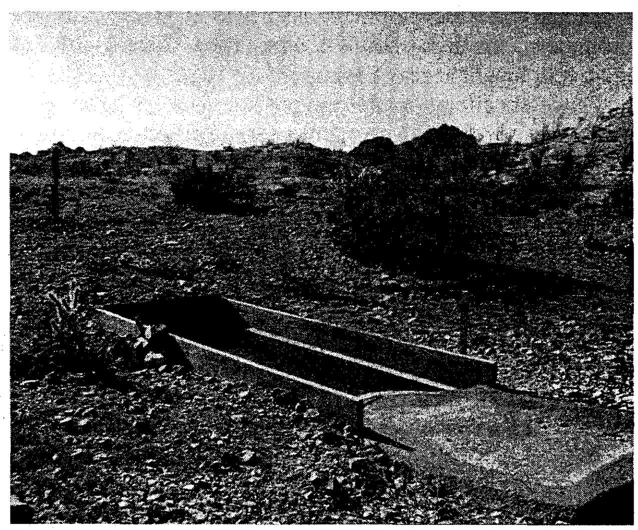

Fig. 7. Underground guzzlers of the design by Lesicka and Hervert (1995) are nearly invisible to humans more than a few meters away, making them especially useful in designated wilderness.

asphalt, and plywood have been used successfully for water collecting surfaces. Durable materials such as concrete or metal are least expensive to maintain, although soil cement appears to be a promising material; (Rice 1990) and Lesicka and Hervert (1995) successfully used areas of native desert soil. Efficiency (percent of water collected) and life-spans (years) vary among materials: steel (98%, 25 years) is best, followed by asphalt roofing (86–92%, 8 years), plastic covered with 2.5 cm of gravel (66–87%, 8–15 years), butyl rubber (98%, 15–20 years), asphalt paving (95%, 15 years) and liquid asphalt soil water (90%, 5 years) (Fairbourn et al. 1972).

The area of the water-collecting surface needed to fill a guzzler (Fig. 8) depends on the storage capacity of the guzzler, minimum annual rainfall at the site, and type of collecting surface. Each 10 m^2 in apron surface area will result in collection of about 1 liter of water for each centimeter of rainfall. Calculations should be based on minimum precipitation expected, rather than the average or maximum, to prevent guzzler failure during drought years. When different types of aprons are used, required surface area can be calculated from the harvest efficiencies (Fairbourn et al. 1972). Leakage, evaporation, and heavy use by wildlife may also dictate a larger apron.

Big-game guzzlers are designed to collect water from either artificial (Gunn 1990) or natural aprons (Stevenson 1990, Lesicka and Hervert 1995). Using slick-rock catchments to collect runoff from bare rock areas is a common technique (Bleich et al. 1982*b*, deVos and Clarkson 1990, Stevenson 1990). These guzzlers take advantage of the fact that rock surfaces yield nearly 100% of the precipitation falling on them as runoff. Several authors (Bardwell 1990, Gunn 1990, Stevenson 1990, Lesicka and Hervert 1995) provide design specifications and other recommendations for construction of these catchments. Bardwell (1990), Bleich and Pauli (1990), deVos and Clarkson (1990), and Gunn (1990) provide information regarding performance of these units over time. These investigators

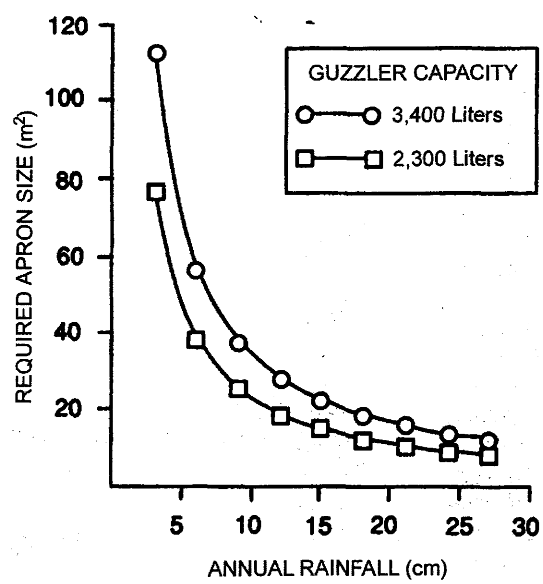

Fig. 8. Size of an apron necessary to fill a guzzler is dependent upon total annual rainfall and storage capacity of the guzzler. The relationship portrayed is based on the assumption the apron yields 100% of rainfall as runoff (after Yoakum et al. 1980).

also evaluated techniques used in the construction of big-game guzzlers and evaluated the reliability of materials.

One of the most important considerations when constructing guzzlers is that all anthropogenic devices are subject to failure; regular monitoring is an essential aspect of any maintenance program. Recently, methods of monitoring the status of water sources that incorporate remote sensing have been developed (Hill and Bleich 1999) for use in areas that are difficult to reach, or that have otherwise restricted access, such as wilderness areas. This technology does not replace biannual visits, which are necessary to detect potential failures, or correct those that already may have occurred (Bleich and Pauli 1990, Hill and Bleich 1999).

The effectiveness and performance of some big-game guzzlers depends on plumbing components. For example, Bleich and Pauli (1990) reported that frozen pipes and fittings accounted for 35 of 98 failures among 22 guzzlers over an 11-year period. Furthermore, of the 98 failures, float-valve malfunction accounted for 31, design and construction flaws for 9, and natural disasters for 6. Other problems, including rusted tanks, rusted drinker boxes, and vandalism, accounted for 17. Overall, each of the 22 guzzlers evaluated averaged 4.4 mechanical failures over an 11-year period, but each was in service an average of 87% of that time. Mechanical failures did not necessarily lead to an inoperative guzzler, but did require effort to repair them.

The most complete guide for construction of guzzlers currently available was prepared by Brigham and Stevenson (1997) and is available on request from the U.S. Department of Interior, Bureau of Land Management, National Applied Resources Sciences Center, P.O. Box 25047, Denver, Colorado, USA.

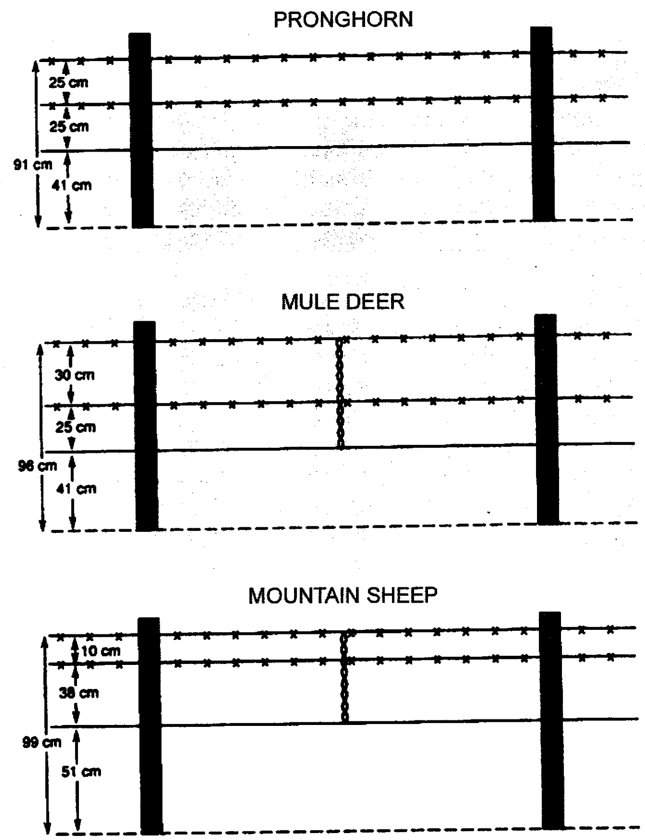

Fig. 9. Recommended specifications for wire fences constructed on ranges used by pronghorn (after Yoakum 1980, Kindschy et al. 1982, U.S. Department of Interior 1985), mule deer (after Jepson et al. 1983, U.S. Department of Interior 1985), and mountain sheep (after Hall 1985, Brigham 1990). Note the use of a smooth bottom wire on all designs and the lack of stays on fences for use on pronghorn ranges.

CONSTRUCTING RANGELAND FENCES

The relationship of fences and wildlife on rangelands in the western United States has been a point of contention for the past century. Fences constructed to control domestic livestock can adversely impact some wildlife species. For example, fences can be major obstacles or traps to pronghorn (Martinka 1967, Spillett et al. 1967, Oakley 1973) and mule deer (Yoakum et al. 1980, Mackie 1981). Proper fence design and use of appropriate construction materials can reduce adverse effects. Details of fence construction on rangelands used by pronghorn, mule deer, elk, bison (*Bison bison*), and collared peccary (*Pecari tajacu*) are available elsewhere (U.S. Department of Interior 1985, Karsky 1988). Preventing the movement of some wildlife species may be desirable, and specific fence designs can accomplish that goal (Longhurst et al. 1962, Messner et al. 1973, deCalesta and Cropsey 1978, Jepson et al. 1983, Karsky 1988).

Fences and Pronghorn

The severity of pronghorn-fence problems varies among areas. Fences are primarily a problem for herds moving seasonally to and from wintering areas on northern rangelands (Oakley 1973). However, seasonal movement problems also were reported in New Mexico (Russell 1964, Howard et al. 1983) and Texas (Buechner 1950, Hailey 1979), especially during drought years.

If fencing is necessary, only that required to provide proper livestock control and minimize hindrance to pronghorn and other wildlife should be used. Unrestricted passage for all age classes during all seasons and all weather conditions should be provided (Yoakum et al. 1980). Fencing watering areas on dry summer rangelands may be as detrimental to pronghorn as fencing migration routes. If a fenced water development is provided specifically for pronghorn, the area should encompass at least 1–2 ha of relatively level terrain (Yoakum et al. 1980).

Fence specifications to control livestock on pronghorn range have evolved over many years (Spillett et al. 1967, Autenrieth 1978, Salwasser 1980, Yoakum 1980, Kindschy et al. 1982, U.S. Department of Interior 1985). Fences should consist of 3 strands of wire, the bottom strand being smooth (Fig. 9). Four- to 6-strand barbed-wire fences limit pronghorn movements and should not be used. The bottom wire should be at least 40 cm above ground. Absence of stays between posts will facilitate the occasional movement of pronghorn through the fence (Yoakum et al. 1980, Kindschy et al. 1982, Hall 1985).

New fences should be flagged with white cloth so pronghorn can become familiar with their locations. By the time a white rag tied to the top of each fence post deteriorates, pronghorn will have become accustomed to the fence (Kindschy et al. 1982). Painting the top of steel fence posts white also helps make the fence more visible to pronghorn (Hall 1985).

Where snow accumulation restricts pronghorn movements, let-down or adjustable fences should be used (Yoakum et al. 1980). A let-down fence can consist of a wooden stay at each fence post to which the wires are attached. The stay is secured to the fence post with a wire loop at the top and either a second loop or a pivot bolt at the bottom.

Let-down fence sections may be designed to permit pulling the let-down sections back against sections of permanently standing fence. Let-down fences should provide for adjustments in wire tension. When the wire is so taut that it does not lie flat on the ground or is so loose that wire loops are formed, a hazard is created for people and animals (U.S. Department of Interior 1985). Adjustable fences (Fig. 10) that allow the movement of one or more wires can allow pronghorn passage during periods when livestock are not present (Anderson and Denton 1980). Adjustable fences are particularly useful when winter snow depths exceed 30 cm (Yoakum et al. 1980).

Pronghorn passes are structures that resemble cattle guards intersecting a fence (Spillett et al. 1967, Mapston and ZoBell 1972, Yoakum et al. 1980, Howard et al. 1983). Suitable locations for pronghorn passes make use of the tendency of individuals to parallel a fence, looking for a way to cross. The pass capitalizes on the ability of pronghorn to jump laterally over obstacles. Pronghorn passes have been built and tested under a variety of conditions (Spillett et al. 1967, Howard et al. 1983). Some adult pronghorn quickly learn to use the facilities, but others do not. Pronghorn fawns often were unable to negotiate the passes. Pronghorn passes are of limited value and should not be used as a panacea for pronghorn access problems (U.S. Department of Interior 1985).

Net-wire fences prevent the movement of pronghorn fawns in particular, and should not be used on public rangelands where pronghorn occur (Autenrieth 1978, Yoakum 1980). However, some adults may become adept at jumping a net-wire fence up to 80 cm high. Higher net-wire fences can be used where the goal is to restrict the movement of animals, such as in live-trapping, control of animals in research projects, decreasing crop depredations, or restricting access to hazardous areas such as highways.

Fences and Mule Deer

The relationship between livestock fences and mule deer has not raised the political furor that it has for pronghorn. However, throughout North America where fences have been built, they likely have caused far greater mortality to deer than to pronghorn. Deer are more apt to be trapped as individuals, whereas large numbers of pronghorn may be restricted. Also, deer frequently are caught in fences in isolated areas not readily witnessed, whereas pronghorn mortalities in open country are easy to observe.

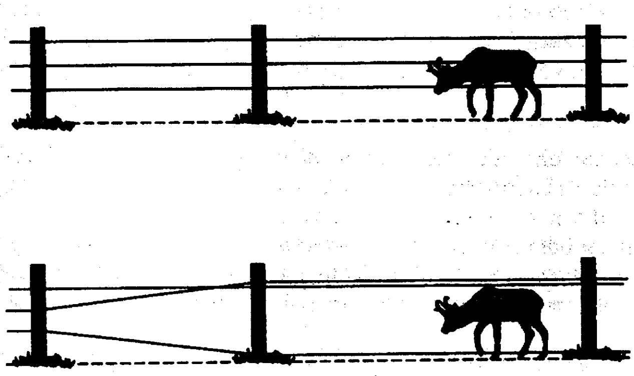

Fig. 10. Adjustable fence modifications to facilitate movement of pronghorn and other ungulates (after Anderson and Denton 1980).

Deer often crawl under fences when not hurried, but jump them when startled or chased (Mackie 1981). When a deer jumps a fence, its feet can become entangled between the top 2 wires, resulting in death. Limiting total fence height to 96 cm can reduce this problem (U.S. Department of Interior 1985) (Fig. 9). If the top wire is barbed, it should be separated from the next wire by 30 cm; otherwise, it should be a smooth wire (Jepson et al. 1983). Unlike fences used on pronghorn ranges, wire stays should be placed every 2.5 m between posts to keep the top wires from twisting around the leg of a deer (Yoakum et al. 1980, U.S. Department of Interior 1985).

The effective height of a fence as a barrier to deer moving uphill is increased on steep slopes. For example, a 110-cm fence on a 20% slope is equivalent to a 140-cm fence on level ground. On a 50% slope, it is equivalent to a 190-cm fence on level ground (Kerr 1979, Anderson and Denton 1980). Thus, height adjustments should be made accordingly.

Let-down fences along seasonal travel routes for deer help ensure free movement. The let-down feature of the fence also helps prevent damage from snow loading during winter. Movements of mule deer also can be aided with an adjustable fence. Net-wire fences no higher than 90 cm allow movement of adult deer but prevent passage of fawns. They should not be placed on summer and autumn migration routes used by deer.

Fences and Mountain Sheep

The construction of wire fences on ranges used by mountain sheep (for example, to exclude livestock from water developments) presents particular problems. Mountain sheep are likely to become entangled in a fence when placing their head through the top 2 wires. This problem is minimized if the 2 top wires are no more than 10 cm apart (Brigham 1990). A 3-wire fence should be used with wires spaced at 51, 38, and 10 cm intervals (Fig. 9), allowing mountain sheep movement under the bottom wire and between it and the middle wire (U.S. Department of Interior 1985, Brigham 1990). Six-wire fence designs (U.S. Department of Interior 1985) are dangerous to mountain sheep and should not be used (Brigham 1990). To minimize the probability of mountain sheep becoming entangled, fences consisting of uprights and 2 parallel rails easily can be constructed (Andrew et al. 1997) (Fig. 11).

Electric Fences

Electric fences often are used to control livestock or feral hoof stock such as burros, and some designs pose little hindrance to movement of wildlife. Electric fences are most effective on moist sites, where 2 wires may be sufficient to control cattle. On sites with at least 60 cm of rain annually, an electric fence can be made of 2 smooth wires at heights of 60 and 90 cm above ground (U.S. Department of Interior 1985, Karsky 1988). The top wire is electrified and the bottom wire serves as the ground. The wires are free running at all posts, and pose little danger of entrap-

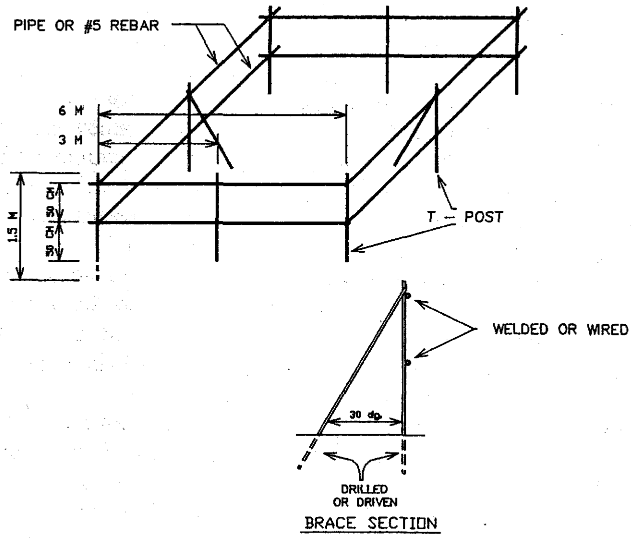

Fig. 11. A simple fence, constructed of metal t-posts and rebar spaced at appropriate intervals effectively excludes feral asses from water sources in desert ecosystems, yet allows passage by native ungulates (after Andrew et al. 1997).

ping mule deer. On drier sites, electric fences require more wires to function effectively (Karsky 1988), and the added wires can adversely affect movements by wildlife.

Wood and Steel Fences

Fences can be constructed entirely from wood posts and rails in a variety of designs with raw materials obtained at the site or manufactured materials (U.S. Department of Interior 1985, Karsky 1988, Andrew et al. 1997). Wood fences are usually expensive but can be attractive and may require less maintenance than wire fences. Construction options include post and pole, log worm, log and block, and buck and pole designs (Karsky 1988). The same principles apply to wood fences as to wire fences in minimizing hindrance to wildlife movements. The top rail or pole of a wooden fence should be kept low to allow mule deer to jump over and the bottom rail or post kept sufficiently high to allow movement of fawns. Andrew et al. (1997) designed an inexpensive rail fence using t-posts and rebar, which was totally effective in reducing access to water sources by feral asses and yet provided unimpeded access by mountain sheep and mule deer.

Rock Jacks

In many areas, soils are too shallow and rocky to allow steel fence posts to be easily driven into the ground (Hall 1985). At such sites, rock jacks are often constructed in the form of wood-rail cribs or wire baskets. The cribs or baskets are filled with rocks and serve as anchors to which wire fences can be secured. Cover and dens for small mammals are provided if the bottom rail of a rock jack is kept 10–15 cm above the ground (Hall 1985). Use of rocks at least 30 cm in diameter will also provide crevasses suitable for use by small mammals (Maser et al. 1979, Hall 1985).

Fences To Exclude Wildlife

Excluding selected wildlife species from certain areas may be desirable. Elk, mule deer, and other species often heavily depredate orchards, vineyards, and other crops; appropriate fence designs can help alleviate such problems. Highways can be hazardous to mule deer and other ungulates that need to reach critically important seasonal ranges. Fences can be used to channel their movement to suitable underpasses and minimize collisions with vehicles. Experimental plots used in research often require exclusion of one or more species of wildlife. Finally, fencing can be used as an alternative to other control measures in reducing predation on livestock.

A 1.8-m upright net-wire fence, or one slanted at 45 degrees to a total height of about 1.3 m, can be used to exclude mule deer (Longhurst et al. 1962, Messner et al. 1973, Karsky 1988). Electric fences with 4–6 wires also discourage deer movements (Karsky 1988).

Fences can be used to reduce or eliminate the need for lethal control of coyotes (*Canis latrans*), which can be excluded from pastures by either woven wire (Thompson 1979, deCalesta and Cropsey 1978, Jepson et al. 1983) or electric fences (Gates et al. 1978, Dorrance and Bourne 1980, Karsky 1988, Nass and Theade 1988). To be effective, a woven wire fence must be at least 170 cm high, have mesh openings no larger than 10 × 15 cm, and have an overhang to prevent jumping and an apron to prevent digging, each at least 40 cm wide (Thompson 1979). A 7-wire electric fence (4 hot wires alternating with 3 ground wires) totaling 130 cm in height also can be used (Dorrance and Bourne 1980). Other electric fence designs are available to deter coyotes (Karsky 1988). In general, fencing to control coyotes is expensive, and probably justified only to protect small areas of high production capacity, such as irrigated pastures.

SUMMARY

Management of livestock on public rangelands has become a divisive and contentious issue. Land management agencies increasingly are criticized for failing to give appropriate consideration to grazing issues that affect wildlife, or wildlife habitat, on public lands. The single greatest change influencing conservation of wildlife on western rangelands during the 1990s has been the shift from an emphasis on competition of livestock with big game to concern for biodiversity in general.

We chose to not criticize current grazing practices but to present a reasonable review of contemporary issues related to livestock management on public lands. Further, we have attempted to: (1) provide an overview of rangeland management to benefit wildlife species and natural communities, with an emphasis on western North America; (2) identify some of the topical issues and primary rangeland systems of particular concern; and (3) describe some of the methods for accommodating wildlife and wildlife-related issues, including habitat enhancement techniques, on rangelands. Students and others making use of information in this chapter are encouraged to further explore the vast literature on management of rangelands and livestock, and to use that information to ensure the persistence of healthy and productive rangeland ecosystems, particularly as they relate to the issue of wildlife conservation.

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