# Ecological Costs of Livestock Grazing in Western North America

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**Abstract:** *Livestock grazing is the most widespread land management practice in western North America. Seventy percent of the western United States is grazed, including wilderness areas, wildlife refuges, national forests, and even some national parks. The ecological costs of this nearly ubiquitous form of land use can be dramatic. Examples of such costs include loss of biodiversity; lowering of population densities for a wide variety of taxa; disruption of ecosystem functions, including nutrient cycling and succession; change in community organization; and change in the physical characteristics of both terrestrial and aquatic habitats. Because livestock congregate in riparian ecosystems, which are among the biologically richest habitats in arid and semiarid regions, the ecological costs of grazing are magnified in these sites. Range science has traditionally been laden with economic assumptions favoring resource use. Conservation biologists are encouraged to contribute to the ongoing social and scientific dialogue on grazing issues.*

Costos ecológicos del pastoreo de ganado en el oeste de Estados Unidos

**Resumen:** *El pastoreo de ganado es la práctica de manejo de la tierra más ampliamente utilizada en el oeste de Norte América. El setenta por ciento del oeste de Estados Unidos se utiliza para pastoreo, incluyendo áreas silvestres, refugios de vida silvestre, bosques nacionales e inclusive algunos parques nacionales. El costo ecológico de esta forma ubicua de uso de la tierra puede ser dramático. Ejemplos de este costo incluyen pérdida de la biodiversidad; decrecimiento de las densidades de población para una amplia variedad de taxones; alteraciones en las funciones del ecosistem, incluyendo ciclos de nutrientes y sucesiones; cambios en la organización de la comunidad y cambios en las características físicas de hábitas terrestres y acuáticos. Dado que el ganado se congrega en ecosistemas ribereños, los cuales están entre los hábitas biológicamente más ricos dentro de las regiones áridas y semi-áridas, los costos ecológicos del pastoreo se magnifican en estos sitios. Tradicionalmente, la ciencia de pastizales, ha estado cargada de suposiciones económicas que favorecen el uso del recurso. Se alienta a los biólogos conservacionistas a contribuir al diálogo social y científico en los problemas del pastoreo.*

## Introduction

Aldo Leopold (1953) once said that to be an ecologist is to live "alone in a world of wounds." The spectacular groundswell of interest in conservation biology is heartening evidence that we no longer work alone. But what about a world of wounds? The wounding of natural processes accelerates, but some wounds are more conspicuous than others. Recognizing a clearcut forest is easy, but it often takes a trained eye to comprehend damage to rangelands. The destruction caused by livestock grazing is so pervasive and has existed for so long that it frequently goes unnoticed. Livestock grazing has re-

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ceived far less attention from conservation biologists than its widespread influence would suggest is appropriate. When I recently surveyed the first six volumes of this journal, for example, I found almost three times as many articles on deforestation as on grazing-related topics.

Livestock grazing is the most widespread influence on native ecosystems of western North America (Wagner 1978; Crumpacker 1984). Grazing by livestock, primarily cattle, is nearly ubiquitous throughout this region. Approximately 70% of the 11 western states of the United States (Montana, Wyoming, Colorado, New Mexico, and westward) is grazed by livestock (Council for Agricultural Science and Technology 1974; Longhurst et al. 1984; Crumpacker 1984), including a broad diversity of ecosystem types and virtually all types of land management designations. Grazing occurs in creosotebush deserts, blackbrush deserts, slickrock mesas, sagebrush flats, pinyon-juniper woodlands, chaparral, ponderosa pine forests, and alpine meadows above timberline.

Grazing occurs on the majority of federal lands in the West, including most of the domains of the U.S. Bureau of Land Management (BLM) and the U.S. Forest Service, as well as in many national wildlife refuges, federal wilderness areas, and even some national parks. In 16 western states, approximately 165 million acres of BLM land and 103 million acres of Forest Service land are grazed by 7 million head of livestock, primarily cattle (U.S. General Accounting Office 1988*a*). Of the BLM lands in these states, 94% is grazed. Of federal wilderness areas, 35% have active livestock grazing allotments (Reed et al. 1989; this figure is from a nationwide survey—the percentage for the West is probably higher). Urbanized areas, some dense coniferous forests, and a few rock-and-ice peaks are about all that is free from the influence of livestock. Given the ubiquity of livestock, it behooves us to understand the consequences of its presence on the Western landscape.

Understanding the influence of domestic livestock upon native ecosystems is a problematic process. Ascertaining the potential natural vegetation of most Western ecosystems is difficult because ungrazed land is extremely rare. Ecologists have gained insight into the effects of grazing primarily in three ways: (1) Historic records provide perspective on the dramatic changes that have transpired since the introduction of livestock to the West (see Cooper 1960). As Hastings (1959) pointed out, however, one must be cautious in interpreting historical records, due to the subjectivity of different observers. Historic photographs have also been used in an attempt to recreate an ecological baseline (see Hastings & Turner 1965); Bahre (1991) reviewed the necessary cautions in interpreting historic photographs. (2) Areas excluded from grazing through serendipity, such as isolated mesa tops, provide startling contrast to adjacent areas that have been continuously grazed (see Rummell 1951). (3) Areas that intentionally exclude livestock (exclosures) provide a before-grazing and after-grazing comparison. Exclosures can be monitored as they recover from the effects of grazing and can be compared with adjacent grazed sites. Almost all exclosures share two characteristics: (1) their areas are usually quite small (Bock et al. 1993*a*), often less than 50 ha; and (2) they have been grazed prior to exclosure. In other words, very few studies of truly ungrazed landscapes exist. Most recreational impact studies concur that the original impact upon a pristine site is the most severe (Cole 1981; Cole & Marion 1986); thus, exclosure studies probably underestimate the true extent of grazing effects because they cannot monitor the most drastic damage, which occurred long ago. In addition, virtually all exclosure studies examine areas too small to encompass landscape-level diversity. In summary, we lack a clear ecological benchmark for determining the effects of grazing.

Attempts to discern grazing effects are also hampered by the difficulty in distinguishing between different range management practices. Management variables include grazing intensity ("stocking rate"), livestock species, seasonality of grazing, and degree of active management, such as movement of livestock between pastures. Unfortunately, the management history of many sites is unknown. Many studies do not describe grazing intensity (see, for example, Glinski 1977; Reynolds & Trost 1980; Crouch 1982). Furthermore, standardized terminology is lacking for different grazing intensities. Relative terms, such as "heavy," "moderate," and "light" grazing, may be undefined (see Jeffries & Klopatek 1987) or qualitatively defined in very different ways. Among the criteria used are presence of livestock, presence of trails, range condition (see Jones 1981), and amount of herbage remaining after a grazing season (see Welch et al. 1991). Studies that have quantified grazing intensity have do so inconsistently. For example, two studies (Mosconi & Hutto 1982; Baker & Guthery 1990) analyzing the effect of "heavy" grazing differed in their definition by a factor of seven. The much-used term "overgrazing" is wrought with controversy and lack of clarity; even specific discussions of overgrazing fail to define it (see Menke & Bradford 1992). This rudimentary state of knowledge interferes with analysis of the role of different grazing practices on biodiversity.

Available evidence indicates that livestock grazing has profound ecological costs. Autecological, synecological, and geomorphological studies have confirmed that native ecosystems pay a steep price for the presence of livestock. Three primary attributes of ecosystems have been elucidated: composition, function, and structure (Franklin et al. 1981). Livestock grazing has a profound impact on all three. The ecological costs of livestock grazing can be summarized as follows:

(1) *Alteration of species composition of communities,* including decreases in density and biomass of individual species, reduction of species richness, and changing community organization.
(2) *Disruption of ecosystem functioning,* including interference in nutrient cycling and ecological succession.
(3) *Alteration of ecosystem structure,* including changing vegetation stratification, contributing to soil erosion, and decreasing availability of water to biotic communities.

## Alteration of Species Composition of Communities

That the introduction of a large-bodied herbivore should have dramatic effects on the species composition of plant communities in arid and semiarid regions should not be surprising. Congressional investigation into rangeland conditions on BLM and Forest Service lands showed that over 50% of public rangelands administered by these two agencies were in "poor" or "fair" condition, meaning that less than half the range was 50% similar to the presumed climax community (U.S. General Accounting Office 1988*a,* 1991*a*). Grazing affects the species composition of plant communities in essentially two ways: (1) active selection by herbivores for or against a specific plant taxon, and (2) differential vulnerability of plant taxa to grazing (Szaro 1989). Decreases in density of native plant species and diversity of native plant communities as a result of livestock grazing activity have been observed in a wide variety of western ecosystems (Table 1).

Grazing also can exert a great impact on animal populations, usually due to indirect effects on habitat structure and prey availability (Wagner 1978; Jones 1981; Mosconi & Hutto 1982; Szaro et al. 1985; Quinn & Walgenbach 1990). The deleterious effects of grazing have been observed in all vertebrate classes (Table 2). The response of native wildlife to grazing varies by habitat. Bock et al. (1993*b*) reviewed the effect of grazing on Neotropical migratory landbirds in three ecosystem types and found an increasingly negative effect on abundances of bird species in grassland, riparian woodland, and intermountain shrubsteppe (almost equal numbers of species with positive and negative responses to grazing in grassland; six times as many with negative as positive responses in shrubsteppe). Due to their mobility and visual orientation, birds may be better able to cope with grazed landscapes than mammals are (Bock et al. 1984). Platts (1979, 1981) reviewed the interaction of biological and geomorphological factors that degrade fish habitat.

The relationship of grazing to insect populations is unclear (Table 3). Studies of grasshoppers (Acrididae) on rangelands have yielded contradictory results: some report an increase in grasshopper densities on heavily grazed lands, and others report a decrease (summarized in Welch et al. 1991). Recent research has clarified that duration of grazing, seasonal differences in plant and insect communities, and plant community architecture

**Table 1. Deleterious effects of livestock grazing on plant communities in western North America.**

| *Habitat* | *Location* | *Effect* | *Authority* |
|---|---|---|---|
| Sonoran Desertscrub | Arizona | Perennial grasses and *Krameria* (palatable shrub) showed dramatic density decreases with grazing | Blydenstein et al. (1957) |
| Mojave Desertscrub | California | 60% reduction in above-ground biomass of annuals, 16–29% decrease in cover of perennial shrubs with grazing | Webb & Stielstra (1979) |
| Sagebrush Desert | Idaho | Grazed site had ⅓ species richness of ungrazed site | Reynolds & Trost (1980) |
| Desert Grassland | New Mexico | Grass density increased by 110% after 30 years of protection from grazing | Gardner (1950) |
| Semidesert Grassland | Arizona | Species richness increased, as did canopy cover for midgrass, shortgrass, shrub, and forb groups, after removal of livestock | Brady et al. (1989) |
| Semidesert Grassland | Arizona | Woody plants significantly more abundant after removal of livestock | Bock et al. (1984) |
| Ponderosa Pine Forest | Washington | Decreased species richness on grazed sites | Rummell (1951) |
| Mountain Canyon | Utah | Absence or near absence of 10 grass species on grazed sites | Cottam & Evans (1945) |
| Riparian | Oregon | Species richness increased from 17 to 45 species nine years after removal of livestock | Winegar (1977) |
| Riparian | Arizona | Herbaceous cover of grazed plot less than half that of ungrazed plot | Szaro & Pase (1983) |
| Riparian | Colorado | Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock | Schulz & Leininger (1990) |

**Table 2. Deleterious effects of livestock grazing on vertebrate animals in western North America.**

| *Organism(s)* | *Location* | *Effect* | *Authority* |
|---|---|---|---|
| Small Mammals | Idaho | Density and diversity reduced on grazed sites | Reynolds & Trost (1980) |
| Small Mammals | Nevada | Density over one-third lower, diversity almost half on grazed sites | Medin & Clary (1989) |
| Songbirds, Raptors, and Small Mammals | Utah | 350% increase in use and diversity after 8 years rest from grazing | Duff (1979) |
| Ducks and all Terrestrial Nongame Birds | Colorado | All more abundant in ungrazed habitat | Crouch (1982) |
| Upland Sandpiper (*Bartramia longicauda*) | North Dakota | Nest density reduced on grazed sites | Bowen & Kruse (1993) |
| Riparian Birds | Montana | Species composition altered by grazing; densities of ⅓ of species differed significantly between heavily and lightly grazed sites—⅔ of these were higher on lightly grazed sites | Mosconi & Hutto (1982) |
| Riparian Passerines | Southeastern Oregon | Species richness decreased on grazed sites | Taylor (1986) |
| Willow Flycatcher (*Empidonax traillii*) | Southeastern Oregon | Abundance increased from 0 to 30 when grazing intensity reduced by 4 times | Taylor & Littlefield (1986) |
| Yellow Warbler (*Dendroica petechia*) | Southeastern Oregon | Abundance increased by 8 times when grazing intensity reduced by 4 times | Taylor & Littlefield (1986) |
| Dickcissel (*Spiza americana*) and Bell's Vireo (*Vireo bellii*) | Oklahoma | Populations 50% lower on grazed sites | Overmire (1963) |
| Lizards | California | Abundance 2 times and biomass 3.7 times higher on ungrazed site | Busack & Bury (1974) |
| Lizards | Arizona | Abundance and diversity higher on ungrazed site in 4 of 5 vegetation types | Jones (1981, 1988) |
| Wandering Garter Snake (*Thamnophis elegans vagrans*) | New Mexico | 5 times more abundant in ungrazed sites | Szaro et al. (1985) |
| Desert Tortoise (*Gopherus agassizi*) | Western U.S.A. | Livestock trample young tortoises, damage burrows and shrubs used for shelter, and remove critical forage | Berry (1978); Campbell (1988) |
| Trout (Salmonidae) | Great Basin | Average increase in production of 184% when grazing reduced or eliminated | Bowers et al. (1979) |
| Trout (Salmonidae) | Idaho | More abundant, larger fish after removal of livestock | Keller & Burnham (1982) |
| Trout (Salmonidae) | Colorado | Standing crop doubled after removal of livestock | Stuber (1985) |

are important factors in determining the effect of grazing on grasshopper populations.

Grazing-induced changes in particular species translate into major conversions of community organization. Grazing is credited with transforming southern New Mexico from grassland to creosotebush (*Larrea*) desert (Whitfield & Anderson 1938; York & Dick-Peddie 1969). Kennedy (1977) noted that grazing thoroughly changed the primary plant species in most Southwest riparian zones. He referred to these changes as "com-

**Table 3. Effects of livestock grazing on insects.**

| *Location* | *Effect* | *Authority* |
|---|---|---|
| Arizona | Grasshopper density 3.7 times greater on protected site in summer, 3.8 times greater on grazed site in fall (different subfamilies, with different food preferences dominant in each season) | Jepson-Innes & Bock (1989) |
| Australia | Ant abundance increased as sheep density increased; all other groups reduced substantially at highest livestock density | Hutchinson & King (1980) |
| Colorado | Grasshoppers significantly more abundant on a lightly grazed site than on a heavily grazed site; because there was no difference between the same sites 19 years earlier, a long-term effect of grazing is indicated | Welch et al. (1991) |
| Oklahoma | Decreases in abundance of most insect groups, dramatic increase in grasshoppers | Smith (1940) |
| South Dakota | Plant community architecture changed from midgrass/tallgrass to shortgrass, which changed grasshopper species composition | Quinn & Walgenbach (1990) |

plete type conversions." Grazing can eliminate a willow stand within 30 years (Kovalchik & Elmore 1992). In Oregon, grazing delayed plant phenology two weeks (Kauffman et al. 1983*b*); such changes could have dramatic effects on communities of pollinators and dispersers. Grazing has also been observed to alter animal foraging guilds (Table 4).

Grazing destabilizes plant communities by aiding the spread and establishment of exotic species, such as tamarisk (*Tamarix*) (Ohmart & Anderson 1982; Hobbs & Huenneke 1992). Livestock help spread exotic plant species by (1) dispersing seeds in fur and dung; (2) opening up habitat for weedy species, such as cheatgrass (*Bromus tectorum;* Gould 1951; Mack 1981), which thrive in disturbed areas; and (3) reducing competition from native species by eating them. As D'Antonio and Vitousek (1992) pointed out, alien grass invasions in North America have been most severe in the arid and semiarid West, where invasion by many species (including *Bromus tectorum, B. rubens, B. mollis, B. diandrus, Taeniatherum asperum,* and *Avena* spp.) was associated with grazing.

## Disruption of Ecosystem Functioning

The deleterious effects of livestock on native ecosystems are not limited to changes in species composition. Grazing also disrupts the fundamental ecosystem functions of nutrient cycling and succession.

An often overlooked characteristic of arid and semiarid ecosystems is the presence of microbiotic (or cryptogamic) soil crusts, delicate symbioses of cyanobacteria, lichens, and mosses from a variety of taxa. The essential role of these microbiotic crusts in nutrient cycling of arid ecosystems has been increasingly appreciated. Crusts perform the major share of nitrogen fixation in desert ecosystems (Rychert et al. 1978). The availability of nitrogen in the soil is a primary limiting factor on biomass production in deserts. In the Great Basin Desert, at least, it is second in importance only to the lack of moisture (James & Jurinak 1978). Microbiotic crusts in arid ecosystems have been correlated with increased organic matter and available phosphorus (Kleiner & Harper 1977), increased soil stability (Kleiner & Harper 1972; Rychert et al. 1978), and increased soil water infiltration (Loope & Gifford 1972; Rychert et al. 1978). Crusts also play an important role in ecological succession because they provide favorable sites for the germination of vascular plants (St. Clair et al. 1984).

Given the fragile nature of microbiotic crusts, it follows that they are easily damaged by livestock grazing. In numerous studies, grazing has been correlated with the loss of microbiotic cover (Wullstein 1973; Johansen et al. 1981; Anderson et al. 1982; Jeffries & Klopatek 1987). Crusts can be severely disrupted even while they (Belnap 1993) and the more conspicuous vascular plant communities (Kleiner & Harper 1972; Cole 1990) appear healthy. Microbiotic species richness has also been shown to decrease under grazing pressure (Anderson et al. 1982). Recent studies on the Colorado Plateau have dramatically demonstrated that soil surface disturbances can virtually stop nitrogen fixation. Nitrogenase activity was reduced 80–100% in the microbiotic crust under a single human footprint, as well as under vehicle tracks (Belnap, personal communication; Belnap 1994; Belnap et al. 1994), and nitrogen content in the leaves of dominant plant species was lower in trampled than untrampled areas (Belnap, personal communication; Harper & Pendleton 1993). If a single footprint can bring a local nitrogen cycle almost to a halt, the impact of a century's work of livestock hoofprints can easily be imagined.

Grazing also can disrupt ecological succession. The cumulative impact of long-term livestock use has produced and maintained early seral vegetation throughout much of the West (Longhurst et al. 1982). Glinski (1977) demonstrated that cattle grazing of small seedlings prevented cottonwood (*Populus fremontii*) regeneration in a southern Arizona riparian zone. He concluded that long-term grazing could eliminate or reduce the upper canopy by preventing the establishment of saplings. Carothers et al. (1974) noted the lack of cottonwood regeneration in grazed areas along the Verde River, Arizona. Prevention of seedling establishment due to grazing and trampling by livestock has transformed a variety of Southwest riparian systems into even-aged,

**Table 4. Effects of livestock grazing on animal foraging guilds in western North America.**

| *Organisms* | *Location* | *Effect* | *Authority* |
|---|---|---|---|
| Riparian Birds | Montana | Flycatching guild, ground-foraging thrush guild and foliage-gleaning insectivore guild affected; bark-foraging guild unaffected | Mosconi & Hutto (1982) |
| Riparian Birds | Oregon | Grazed sites preferred by insectivores, ungrazed sites by herbivores and granivores | Kauffman et al. (1982) |
| Lizards | Arizona | More sit-and-wait lizards on grazed sites; open-space foragers and wide-ranging foragers decreased on grazed sites | Jones (1981) |
| Grasshoppers | South Dakota | Obligate grass-feeders dominated on grazed sites, mixed-forb-and-grass-feeders on ungrazed sites | Quinn & Walgenbach (1990) |

nonreproducing vegetative communities (Carothers 1977; Szaro 1989). In Oregon, grazing retarded succession in the willow-black cottonwood (*Salix-Populus trichocarpa*) community, and there was little if any regeneration of alders (*Alnus*) or cottonwoods (Kauffman et al. 1983*b*). Davis (1977) concluded that livestock grazing was "probably the major factor contributing to the failure of riparian communities to propagate themselves."

Ascertaining patterns of ecological succession in xeric rangelands is not easy; thus, the effect of livestock on successional processes is unclear. Traditionally, range management was based upon Clements' (1916) classic model of ecological succession, where seral stages lead to a stable climax. Early on, this concept of predictable, directional succession was applied to range ecosystems (Sampson 1919). This "range succession model" eventually formed the basis of range condition classification, as exemplified by government manuals and early range management textbooks (Stoddart & Smith 1943), and summarized in an extensive review by Ellison (1960). In the arid West, however, vegetation change due to grazing has not followed the prediction of this linear model. Recent evidence suggests that range ecosystems have not evolved as well-balanced communities with stable species compositions (Johnson & Mayeux 1992).

More recently, a less Clementsian view of xeric rangeland succession, referred to as the "state-and-transition model," has been proposed (Westoby et al. 1989). According to this model, relatively stable, discrete vegetation states go through transitions induced by natural episodic events such as fire or by management actions such as grazing (Laycock 1991). As Friedel (1991), Laycock (1991), and others have discussed, transitions between states sometimes cross successional "thresholds." Once certain thresholds have been crossed, as in severe soil erosion, succession may not be reversible except by strong, active management. Although this model is in its infancy, it may someday provide a means to predict if grazing can cause long-term degradation by inducing irreversible succession across thresholds.

## Alteration of Ecosystem Structure

The physical structure of ecosystems, including vegetation stratification, is often changed by livestock grazing. In central Washington, grazing was responsible for changing the physical structure of ponderosa pine forest from an open, park-like tree overstory with dense grass cover to a community characterized by dense pine reproduction and lack of grasses (Rummell 1951). Grazing was at least partially responsible for similar structural changes in ponderosa pine forests of northern Arizona (Cooper 1960). Historic records indicate that extensive willow stands once occurred throughout the rangelands of the Intermountain West, which are now almost completely absent (Kovalchik & Elmore 1992). Grazing structurally changed habitat for the wandering garter snake (*Thamnophis elegans vagrans*) through the loss of small trees and shrubs (Szaro et al. 1985). In central Arizona, lizard habitat was changed when livestock reduced low-height vegetation by totally consuming perennial grasses and severely reducing palatable shrubs (Jones 1981). In Oregon, Taylor (1986) noted that lower vegetative strata were affected by grazing. In blackbrush (*Coleogyne ramosissima*) desert habitat, ungrazed sites had significantly more shrub and herbaceous cover (Jeffries & Klopatek 1987). In a high-altitude willow riparian community in Colorado, grazing influenced the spacing of plants and the width of the riparian zone (Knopf & Cannon 1982).

Grazing removes soil litter, which can have both physical and biological effects. Schulz and Leininger (1990) observed twice as much litter in an exclosure as in surrounding grazed habitat. In Oregon, removal of soil litter was thought to be the cause of delayed plant phenology (Kauffman et al. 1983*b*), which in turn could affect communities of animal pollinators.

Researchers have long recognized that grazing contributes to the deterioration of soil stability and porosity and increases erosion and soil compaction. Seventy years ago, Aldo Leopold (1924) declared that "grazing is the prime factor in destroying watershed values" in Arizona. Grazing reduces the roughness coefficient of watersheds, resulting in more surface runoff, more soil erosion, and massive flooding (Ohmart & Anderson 1982). Grazing in the upper Rio Grande changed plant cover, thus increasing flash floods and, consequently, erosion (Cooperrider & Hendricks 1937). As grazing-induced gullying lowered the stream channel along an Oregon stream, associated plant communities changed from wet meadow to the more xeric sagebrush-rabbitbrush (*Chrysothamnos*) type (Winegar 1977). Davis (1977) concluded that removal of upland vegetation by livestock was a major factor in the increase in devastating floods. Numerous authors have noted extreme erosion and gullying when comparing heavily grazed to ungrazed sites (see Cottam & Evans 1945; Gardner 1950; Kauffman et al. 1983*a*). Ellison (1960) concluded that "as a result of some degree of denudation, accelerated soil erosion is inseparably linked with overgrazing on arid lands the world over."

Grazing has also repeatedly been shown to increase soil compaction and thus decrease water infiltration (Alderfer & Robinson 1949; Orr 1960; Rauzi & Hanson 1966; Bryant et al. 1972; Rauzi & Smith 1973; Kauffman & Krueger 1984; Abdel-Magid et al. 1987; Orodho et al. 1990). In arid and semiarid lands where water is the primary ecological limiting factor, major losses of water from ecosystems can lead to severe desertification. Some controversy exists as to whether livestock grazing was the *cause* of increased flooding and erosion or whether the synchrony of increased channel trenching and the introduction of vast livestock herds during the last century was coincidental. Episodes of channel trenching certainly occurred prior to the introduction of livestock (Bryan 1925; Karlstrom & Karlstrom 1987). Most reviewers, however, conclude that, at the least, livestock have been a contributing factor to the entrenching of stream channels in the Southwest (Bryan 1925; Leopold 1951; Hereford & Webb 1992; Betancourt 1992). This interaction of climatic, geomorphic, and biological factors has been summarized as a "trigger-pull": long-term climatic trends were already underway when cattle arrived to serve "as the trigger-pull that set off an already loaded weapon" (Hastings 1959).

## Costs of Grazing Magnified: Riparian Habitats in the Arid West

Livestock, like humans, are adapted to mesic habitats, and they select riparian areas for the same reasons we do: shade, cooler temperatures, and water. In addition, riparian areas offer an abundance of food. Many observers have noted that cattle spend a disproportionate amount of their time in riparian zones (Ames 1977; Kennedy 1977; Thomas et al. 1979; Roath & Krueger 1982; Van Vuren 1982; Gillen et al. 1984). That livestock actively select riparian habitats, however, is a cause for ecological concern because these habitats are among the biologically richest in many arid and semiarid regions and are easily damaged. Because livestock spend much of their time in riparian communities, and because the ecological stakes are highest here, many of the adverse impacts of grazing are magnified in these habitats.

Western riparian zones are the most productive habitats in North America (Johnson et al. 1977), providing essential wildlife habitat for breeding, wintering, and migration (Gaines 1977; Stevens et al. 1977; Brode & Bury 1984; Laymon 1984; Lowe 1985). Riparian habitats in the Southwest are home to the North American continent's highest density of breeding birds (Carothers et al. 1974; Carothers & Johnson 1975), rarest forest type, and more than 100 state and federally listed threatened and endangered species (Johnson 1989). Approximately three-quarters of the vertebrate species in Arizona and New Mexico depend on riparian habitat for at least a portion of their life cycles (Johnson et al. 1977; Johnson 1989). Even xeroriparian habitats—normally dry corridors that intermittently carry floodwaters through low deserts—support five to ten times the bird densities and species diversity of surrounding desert uplands (Johnson & Haight 1985).

Sadly, these biological treasures are in extreme danger. The Environmental Protection Agency concluded that riparian conditions throughout the West are now the worst in American history (Chaney et al. 1990). Over 90% of Arizona's original riparian habitat is gone (Johnson 1989). Less than 5% of the riparian habitat in California's Central Valley remains; 85% of that is in disturbed or degraded condition (Franzreb 1987). The degradation of Western riparian habitats began with severe overgrazing in the late Nineteenth Century (Chaney et al. 1990), and grazing remains "the most insidious threat to the riparian habitat type today" (Carothers 1977). An extensive survey of Southwest riparian community types concluded that "livestock may be the major cause of excessive habitat disturbance in most western riparian communities" (Szaro 1989). The Oregon-Washington Interagency Wildlife Committee (1979), composed of biologists from several government agencies, concluded that grazing is the most important factor in degrading wildlife and fisheries habitat throughout the 11 western states. Likewise, ecologists in Montana suggested that livestock grazing is the major cause of habitat disturbance in most western riparian communities (Mosconi & Hutto 1982).

Livestock affect four general component of riparian systems: (1) streamside vegetation, (2) stream channel morphology, (3) shape and quality of the water column, and (4) structure of streambank soil (Platts 1979, 1981, 1983; Kauffman & Krueger 1984; Platts & Nelson 1989). As summarized by Platts (1981), "Grazing can affect the streamside environment by changing, reducing, or eliminating vegetation bordering the stream. Channel morphology can be changed by accrual of sediment, alteration of channel substrate, disruption of the relation of pools to riffles, and widening of the channel. The water column can be altered by increasing water temperature, nutrients, suspended sediment, bacterial populations, and in the timing and volume of streamflow. Livestock can trample streambanks, causing banks to slough off, creating false setback banks, and exposing banks to accelerated soil erosion."

Riparian vegetation is altered by livestock in several ways: (1) compaction of soil, which increases runoff and decreases water availability to plants; (2) herbage removal, which allows soil temperatures to rise, thereby increasing evaporation; (3) physical damage to vegetation by rubbing, trampling, and browsing; and 4) altering the growth form of plants by removing terminal buds and stimulating lateral branching (Kauffman & Krueger 1984; Szaro 1989). Livestock grazing is one of the principal factors contributing to the decline of native trout in the West. Cattle activities especially deleterious to fish are the removal of vegetative cover and the trampling of over-hanging streambanks (Behnke & Zarn 1976). Livestock have been shown to decrease water quality of streams (Diesch 1970; Buckhouse & Gifford 1976). Changes in water chemistry (Jeffries & Klopatek 1987) and temperature (Van Velson 1979), in effect, create an entirely new aquatic ecosystem (Kennedy 1977; Kauffman & Krueger 1984). Insights such as these led the American Fisheries Society to issue a formal position statement calling for an overhaul of riparian zone management (Armour et al. 1991).

## Historical and Management Considerations

By virtually any measure, livestock grazing has serious ecological costs in western North America. Grazing has reduced the density and biomass of many plant and animal species, reduced biodiversity, aided the spread of exotic species, interrupted ecological succession, impeded the cycling of the most important limiting nutrient (nitrogen), changed habitat structure, disturbed community organization, and has been the most severe impact on one of the biologically richest habitats in the region. While undoubtedly there are exceptions to this theme of destruction, clearly much of the ecological integrity of a variety of North American habitats is at risk from this land management practice.

In addition to grazing per se, the industry of livestock production entails a number of indirect costs to native biodiversity. Livestock compete with native herbivores for forage ("usurpation") and often consume the most nutritive species ("highgrading"). Fencing, which is a fundamental livestock management tool, creates obstacles for many native wildlife species, such as the pronghorn (*Antilocapra americana*). The livestock industry has played a large role in the elimination of native predators; some of the most vehement opposition to predator reintroduction continues to come from livestock interests. Exotic species, such as crested wheatgrass (*Agropyron cristatum*), are planted as "range improvements." In addition, livestock can transmit disease to native animals (Mackie 1978; Longhurst et al. 1983; Menke & Bradford 1992).

Agency management priorities often overemphasize livestock needs at the expense of wildlife. A recent Congressional study of BLM and Forest Service management confirmed that wildlife receives only a small percentage of available staffing and funding. During fiscal years 1985–1989 the BLM directed only 3% of its total appropriation toward wildlife habitat management, while 34% of its budget went to its three consumptive programs: range, timber, and energy and minerals (U.S. General Accounting Office 1991*b*). Wildlife at national wildlife refuges also suffers from management emphasis on livestock. Cattle grazing and haying occur at 123 refuges; at any given site these activities occupy up to 50% of refuge funds and 55% of staff time. Field studies indicated that these livestock-related activities directly impeded wildlife conservation (Strassman 1987). Strong agency bias in favor of grazing often leads to contradictory management decisions. A recent Forest Service analysis of sensitive vertebrate species identified livestock grazing as one of five factors jeopardizing the northern goshawk (*Accipiter gentilis*) in the Southwest (Finch 1992). Yet the goshawk management recommendations (Reynolds et al. 1992), released by the same office in the same year, did not even mention grazing. Such predilections by agencies reflect similar biases within the range management discipline: a recent 500-page textbook on range management (Holechek et al. 1989) devotes one paragraph to nongame wildlife.

A variety of justifications are heard for grazing in the West. Because livestock has been such a prominent component of Euro-American settlement of the West, some observers see it as a traditional pastime and assume it is appropriate for the land. Some range managers maintain that livestock are actually *necessary* for ecosystem health, that "grass needs grazing" (Chase 1988; Savory 1988). Popular claims such as these are rooted in a scientific debate on the consequences of herbivory on grassland ecosystems. As the "herbivore optimization" hypothesis goes, loss of tissue to herbivores can actually increase total productivity of the grazed plant. Such a response to herbivory is referred to as "overcompensation" by the plant (Owen & Wiegart 1976; Dyer et al. 1982). When different levels of ecological hierarchy (individual, population, community; Belsky 1987) and a wide diversity of ecosystem types, geographic settings, and degrees of management intensity are lumped together into one generalized theory, clarity is lost. Much of the evidence for overcompensation comes from highly productive and intensively managed systems, not from arid rangelands (Bartolome 1993). Few studies have demonstrated overcompensation in western North America (Painter & Belsky 1993), where much of the rangeland resource is not grassland. Observations of native herbivores lend no support to the idea that compensatory growth has any relevance at the community level in western rangelands (Patten 1993). According to Vicari and Bazely (1993), "there is little evidence that the act of grazing per se increases the fitness of grasses, or any other plant species, except under highly specific circumstances."

Other scientists and range managers suggest that livestock, given their capacity for altering so many aspects of ecological organization, could be used as a wildlife management tool (Bokdam & Wallis de Vries 1992; Hobbs & Huenneke 1992). In summarizing a symposium on the topic, Severson (1990) clarified that such applications may be very limited, and that what benefits one species may prove detrimental to another. Because two species in the same community may vary in their response to grazing (Hobbs & Huenneke 1992), determination of its success or failure as a management practice depends on which species is used as a criterion. On many national wildlife refuges, grazing and haying occur with the rationale that these practices will benefit wildlife. Upon review of 123 refuges, Strassman (1987) concluded that "although in theory cattle grazing and haying can be wildlife management tools, as implemented they are tools that do more harm than good."

It is often stated that livestock have merely taken the place of large native herbivores, particularly bison (*Bison bison*). The presettlement abundance of bison on the Great Plains is legendary. West of the Rocky Mountains, however, bison were rare or absent in Holocene times. The species was present in the northern Rockies region, marginally present along the northern and western perimeter of the Great Basin (Hall 1981; Mack & Thompson 1982; Zeveloff 1988; Van Vuren & Deitz 1993) and absent altogether from Arizona (Cockrum 1960; Hoffmeister 1986), western New Mexico (Bailey 1971), as well as most of California (Jameson & Peeters 1988), and Nevada (Hall 1946). The native steppe vegetation of much of the Intermountain West, characterized by caespitose bunchgrasses and a prominent microbiotic crust, reflects the absence of large numbers of large-hooved, congregating mammals. These steppe ecosystems have been particularly susceptible to the introduction of livestock; microbiotic crusts, as mentioned earlier, are easily damaged by trampling. In contrast, the slightly wetter Great Plains grasslands, characterized by rhizomatous grasses and a lack of microbiotic crusts, were well-adapted to withstand herbivory by large ungulates (Stebbins 1981; Mack & Thompson 1982). Theoretically, then, the Great Plains should be better suited to livestock grazing than the arid and semiarid ecosystems west of the Rockies. It should also be noted that the ecological analogy between cattle and bison is incomplete. Cattle, unlike bison, spend a disproportionate amount of time in riparian habitats. In a comparative study of cattle and bison feeding ecology in the Henry Mountains, Utah, Van Vuren (1982) noted that cattle distribution was limited to gentle slopes near water, regardless of forage, while bison roamed widely, seemingly unaffected by slope or proximity to water.

The controversy about flood cycles and arroyo-cutting, discussed earlier, is but one part of a larger controversy concerning the respective roles of climate change and human land use—including livestock grazing—in changing the vegetation of western North America. The international borderlands of southern Arizona and northern Sonora, Mexico, have been the site of the most intensive study of this issue. The appearance of *The Changing Mile* (Hastings & Turner 1985) almost three decades ago promoted the then new idea that the region's dramatic vegetation change during the previous century was due to increasing aridity—to natural climate change—and not to human land-use patterns. Using pairs of photographs, one historic and one recent, *The Changing Mile* visually documented vegetation change and concluded that its cause was an increasingly arid climate. As for livestock, these authors felt the evidence was somewhat ambiguous and concluded that livestock may have contributed to vegetation change in the region "but have not been the primary agent of change" (Hastings & Turner 1965). This work has since been widely quoted by livestock interests to support the idea that historic overgrazing was overstated and, therefore, to justify the continuation of grazing in the region.

Recently vegetation change along the Arizona borderlands has received renewed scholarly attention. This new work reached a very different conclusion: "probably no single land use has had a greater effect on the vegetation of southeastern Arizona or has led to more changes in the landscape than livestock grazing range management programs. Undoubtedly, grazing since the 1870s has led to soil erosion, destruction of those plants most palatable to livestock, changes in regional fire ecology, the spread of both native and alien plants, and changes in the age structure of evergreen woodlands and riparian forests" (Bahre 1991). Moreover, the new analysis (Bahre 1991) states that "the present historic evidence ... casts serious doubt on the hypothesis that a shift toward greater aridity is the primary factor for regional vegetation changes." Bahre (1991) agrees that climatic oscillations since 1870 have resulted in short-term fluctuations in vegetation but insists that long-term directional changes, including degradation of riparian habitats and spread of exotic species, have resulted from human disturbances, including overgrazing by cattle. Bahre challenges the conclusions of *The Changing Mile* on the basis of several factors, including lack of historic evidence to support several key assumptions in the earlier work (for example, that overgrazing had been practiced since the time of the Mexican occupation), and that the majority of historic photographs were taken *after* the worst grazing damage had already occurred. In other words, *The Changing Mile* made comparisons to the wrong baseline data. For now, the best historic evidence seems to support the idea that livestock grazing, interacting with fluctuations in climatic cycles, has been a primary factor in altering ecosystems of the Southwest.

Human intervention is needed to restore the West to ecological health. According to the BLM's own definition, over 68% of its lands are in "unsatisfactory" condition (Wald & Alberswerth 1989; U.S. General Accounting Office 1991*a*). Approximately 464 million acres of American rangeland have undergone some degree of desertification (Dregne 1983). Attempts at restoration of livestock-damaged ecosystems have offered both good and bad news: riparian areas often show rapid recovery upon removal of livestock, but more xeric uplands demonstrate little inherent capacity for healing.

Riparian areas appear to be relatively resilient. At a Sonoran Desert spring, Warren and Anderson (1987) documented dramatic recovery of marsh and riparian vegetation within five years of livestock removal. All nine aspects of trout habitat studied along Summit

Creek, Idaho, improved within two years of livestock removal (Keller et al. 1979). Mahogany Creek, Nevada, also showed major improvement in fisheries habitat after only two years of exclosure (Dahlem 1979). Beaver and waterfowl returned to Camp Creek, Oregon, within nine years of cattle exclosure (Winegar 1977). However, the aquatic component of riparian systems often is the quickest to show improvement. Szaro and Pase (1983) observed extremely limited recovery of a cottonwood-ash-willow association in Arizona after four years. Knopf and Cannon (1982) noted that a willow community was slower to heal than the adjacent stream: 10–12 years was insufficient for recovery of the former.

The U.S. General Accounting Office (1988*b*) recently reviewed riparian restoration efforts on BLM and Forest Service lands in the West and concluded (1) that even severely degraded habitats can be successfully restored and (2) that successful restoration to date represents only a small fraction of the work that needs to be done. They noted that successful techniques varied considerably from site to site, and that many sites could repair themselves, given respite from livestock. Successful riparian restoration efforts are summarized by the U.S. General Accounting Office (1988*b*) and Chaney et al. (1990).

In numerous studies of riparian grazing impact, investigators concluded that total removal of livestock was necessary to restore ecosystem health. Along Mahogany Creek, Nevada, reduction in grazing had little benefit; only a complete removal brought about habitat improvement (Dahlem 1979; Chaney et al. 1990). Ames (1977) found that even short-term or seasonal use is too much and compared mere reductions in livestock numbers to letting "the milk cow get in the garden for one night." In a recent comparison of 11 grazing systems, total exclusion of livestock offered the strongest ecosystem protection (Kovalchik & Elmore 1992). As Davis (1982) put it, "If the overgrazing by livestock is one of the main factors contributing to the destruction of the habitat, then the solution would be to ... remove the cause of the problem."

The vast majority of damaged rangeland acreage is on arid and semiarid lands, where the prognosis for restoration is poor (Allen & Jackson 1992). To rehabilitate arid lands is somewhat analogous to trying to grow a garden without water. Perhaps because there is little chance of rapid success, land managers have been slow to take up the challenge of restoring arid rangelands. Cooperrider (1991) noted that "the principal purpose of most rangeland rehabilitation projects has been restoration of livestock forage. Such projects typically end up reducing plant and animal species diversity." Some dryland restoration projects touted as success stories (such as the Vale project in southeastern Oregon; Menke & Bradford 1992), actually have entailed large-scale plantings of exotic species. Such activities restore livestock forage, not native ecosystems.

Is there an ecologically sustainable future for livestock grazing in western North America? This ultimately is a question of human values, not of science. We must decide how much we really care about native diversity and ecosystem processes and what we are willing to do to sustain them. Ecological science and conservation biology have a key role to play in helping society make a wise decision. Scientific input into grazing issues has come laden with resource extraction assumptions: one of the primary goals of range management is to maximize livestock production (Stoddart & Smith 1943; Bell 1973; Menke & Bradford 1992) or to "improve the output of consumable range products" (Holechek et al. 1989). Given this economic underpinning, the ecological merit of livestock in the West has generally gone unchallenged. It is time that conservation biologists take a careful look at the most pervasive land use in western North America and scrutinize the practice described as "the single most important factor limiting wildlife production in the West" (Smith 1977) and "one of the primary threats to biological diversity" (Cooperrider 1991). Whatever decision society reaches, it will be a wiser, more informed one if the conservation biology community contributes its insights to the debate.

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