THE EFFECTS OF FIRE AND GRAZING ON VEGETATION AND AQUATIC HABITATS IN SEMIDESERT GRASSLANDS OF THE SOUTHWEST

by

CLAUDIA E. KELLEY

A THESIS

Presented to the Interdisciplinary Studies Program:
Environmental Studies
and the Graduate School of the University of Oregon
in partial fulfillment of the requirements
for the degree of
Master of Science

March 1991

APPROVED: [redacted]
Dr. Stanton Cook, Co-chair

[redacted]
Dr. William Liss, Co-chair

An Abstract of the Thesis of

Claudia E. Kelley for the degree of Master of Science
in the Interdisciplinary Studies Program: Environmental Studies
to be taken March 1991

Title: THE EFFECTS OF FIRE AND GRAZING ON VEGETATION AND AQUATIC HABITATS IN SEMIDESERT GRASSLANDS OF THE SOUTHWEST

Aproved: [redacted]
Dr. Stanton Cook, Co-chair

[redacted]
Dr. William Liss, Co-chair

This paper reviewed and consolidated the current literature pertaining to the effects of fire and grazing on upland vegetation and the correlated effects on riparian zones and aquatic habitats in the Southwest. Lists of the major species of the semidesert grasslands biome have been assembled and specific information gathered on the ecological attributes of the biome. Restoration procedures and management strategies were presented.

VITA

NAME OF AUTHOR: Claudia E. Kelley

PLACE OF BIRTH: Wahiawa, Oahu, Hawaii

DATE OF BIRTH: July 8, 1959

GRADUATE AND UNDERGRADUATE SCHOOLS ATTENDED:

University of Oregon
Central Washington University

DEGREES AWARDED:

Bachelor of Science, 1982, Central Washington University

Bachelor of Arts, 1982, Central Washington University

AREAS OF SPECIAL INTEREST:

Environmental Education and Wildlife Conservation

PROFESSIONAL EXPERIENCE:

Teacher, Biology, Hawaii Preparatory Academy, Kamuela Hawaii. Summer, 1986

Hydroacoustic Technician, Grant County Public Utility District, Ephrata, Washington, April-September 1984, April-September, 1985

Technician, Environmental Department, Battelle Pacific Northwest Laboratories, Richland, Washington, April-September 1981

VOLUNTEER EXPERIENCE:

Wildlife Rehabilitation and Education, Cougar Country, Eugene, Oregon, 1988-1991

AWARDS AND HONORS:

University of Oregon Joseph Kinsman Starr Scholarship, 1988-1989

Northwest Association of Environmental Studies Scholarship, 1989-1990

ACKNOWLEDGEMENTS

I wish to express my gratitude to Dr. Stanton Cook and Dr. William Liss, my committee members, for their guidance and patience and for generously sharing their knowledge. I would also like to thank Dr. David Gori of The Arizona Nature Conservancy, who is always ready and willing to share his expertise, and for his endless encouragement.

Additional thanks go to Dr. Phillip Dumas and Dr. Darwin Goodey for their support.

DEDICATION

This paper is dedicated to my mother for her generous care, love, and support.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION AND OVERVIEW

Distributed throughout the American Southwest are numerous ephemeral, intermittent, and perennial streams and wetlands (Platts & Nelson, 1989). Although they comprise only a small percentage of the land, these riparian areas are among the most productive, valuable, and unique ecosystems in the semidesert Southwest and their effect and influence upon other ecosystems is extensive (Meyers, 1989). Relative to the limited geographic area they encompass, riparian ecosystems possess a disproportionate importance (Brown, Brinson, & Lugo, 1978), particularly in arid regions. Typically these areas exhibit a nearly classic example of edge effect (Odum, 1978) and are uniquely characterized by a high density and diversity of species and high productivity. Because riparian lands interface aquatic and upland ecosystems, a close association and dependency exists between these lands (Brown et al., 1978).

Throughout the past decade, research has shown that riparian areas provide a complex of habitats for a large diversity of plants and animals (Brinson, Swift, Plantico, & Barclay, 1981). Since small elevational changes in the floodplain produce differences in the soils, a large variety of habitats are available to species living within the area. The survival of a large number of small-bodied species that are restricted or endemic to riparian habitats, and that of most livestock and game animals, depends upon the resources provided by stream margins, even though much of the latter species' time is spent elsewhere. Riparian lands, in addition to providing critical habitats, also provide natural highways that facilitate safe movement between areas (Odum, 1978).

Many of the attributes of riparian areas which make them attractive to wildlife, such as flowing water, high plant productivity, shade, and use as travel corridors, also enhance their value to humans. As outdoor recreational opportunities have become limited, increased pressure has been placed upon riparian lands to provide for hiking, fishing, wildlife watching, and camping. Because of their value to diverse special interest groups, conflicts have arisen over what is appropriate riparian management. All too frequently, these areas become battlegrounds between environmental and conservation groups, that wish to see riparian areas protected, and exploiting groups that wish to further their extractive use of these productive areas (Brown, 1982).

Misuse and exploitation of riparian areas has led to their degradation. Fortunately, land managers have recognized the need to improve deteriorating riparian areas and efforts and studies are underway to determine effective methods of restoration.

Much research has been conducted and numerous articles published on the techniques available for enhancing and rehabilitating riparian ecosystems in Southwest environments (DeBano & Schmidt, 1989a). Unfortunately, the information is located in disparate sources and there have been few systematic attempts to pull it all together. The compilation of such information will make it more readily available to individuals and organizations attempting to develop management strategies and facilitate conservation and rehabilitation.

For example, riparian plant communities and native fish are among The Nature Conservancy's highest conservation priorities in Arizona. On two of their reserves they manage either the entire or a significant proportion of the upland watershed that sustains these threatened communities and species. However, these lands can be degraded by overgrazing. Thus, one of the major concerns of The Nature Conservancy and similar organizations is how such watersheds should be managed in order to return them to a more pristine condition and how the chosen management activities will affect riparian communities and species.

In this paper the author will review and consolidate the current literature pertaining to the effects of fire and grazing on upland vegetation and the correlated effects on riparian zones and aquatic habitats in the Southwest. This study will be limited to research and literature that focuses on (a) the effects of grazing and fire on the composition and abundances of plant species in semidesert grasslands and shrublands, and (b) how vegetation cover and composition in semidesert grasslands affects the hydrology of the riparian area.

Methodology

The information has been obtained from books, journals, research papers, technical reports, published and unpublished manuscripts, conference proceedings, and personal communications.

For the purposes of this paper, the author has used The Arizona Nature Conservancy's definition of riparian area:

> A riparian area is defined as an aquatic or terrestrial ecosystem that is associated with bodies of water such as streams, lakes, or wetlands or is dependent upon the existence of perennial, intermittent or ephemeral surface or subsurface water drainage (Gori, personal communication, Feb. 20, 1991).

The Region and Its Habitats

Upland Vegetation

The geographic area of this study comprises the semidesert grasslands that adjoin and largely surround the Chihuahuan desert. In the Southwest, extensive areas of this grassland occur in southeast Arizona, the southern half of New Mexico, Trans-Pecos Texas, extreme northeast Sonora, Chihuahua, and Western Coahuila. According to Brown (1982), "At its western limits in Mohave, Yavapai, Gila, Pinal, Pima, and Graham counties of Arizona, semidesert grassland may be encountered above and rarely even within the Sonoran Desert (p. 123). (See Figure 1.)

Originally, southwestern semidesert grasslands were mapped and described by Shreve (1917) as desert-grassland transition. Since that time these communities have most frequently been mapped or described as desert grass or desert grassland. Use of the term desert grassland has been replaced by the more accurate term semidesert grassland (Brown, 1982). Brown describes the semidesert grassland as a perennial grass-scrub dominated landscape located between desertscrub below and evergreen woodland, Plains grassland, or, less frequently, chaparral above.

The lower elevational range for the grasslands is between 1,100 and 1,400 meters, whereas the upper elevation usually ranges between 1,400 and 1,700 meters, occasionally to 1,900 meters. The average annual precipitation for most semidesert grassland is between 250 and 450 millimeters. Except for in extreme west-central Arizona, over 50% of this total comes between April and September when rainfall averages 150 millimeters or more (Brown, 1982).

These grasslands, which formerly extended over considerable areas of North America, are characterized by drying winds, seasonal drought, mild winters, and warm to hot summers, with several days over 100° F. It is drought and mild winter temperatures that make this area particularly susceptible to alteration by livestock and people. Such alterations have resulted in much of the original perennial

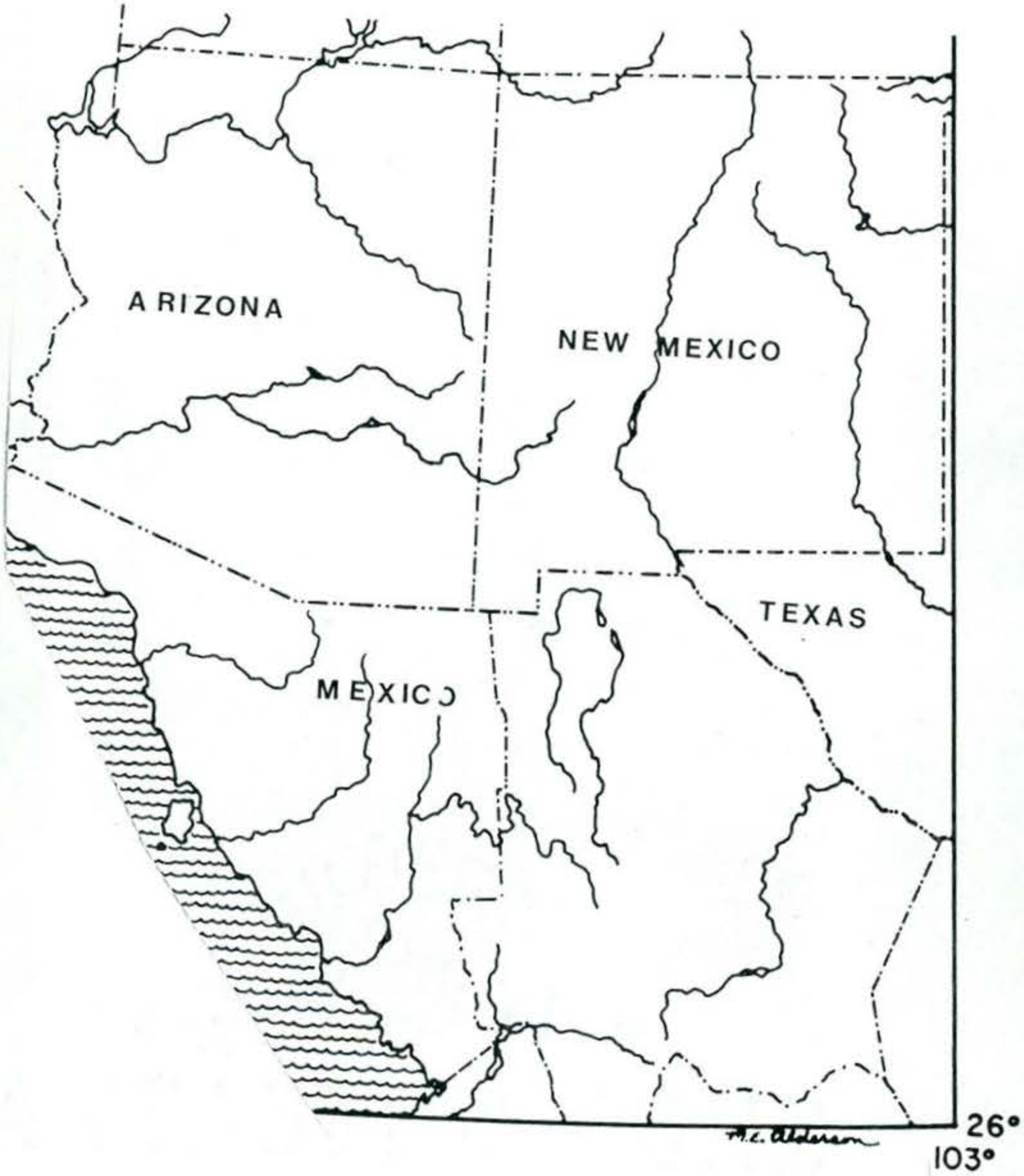

Figure 1: Semidesert grasslands of the American Southwest.
Source: Adapted from Brown (1982).

bunch grass being replaced by annual grasses, low-growing sod grasses, shrubs, woody plants, leafy succulents, and cacti. Frequently, the grassy landscape is broken up by large, diverse, well-spaced scrub (Brown, 1982).

Although semidesert grassland is transitional in that it is located between Plains grassland to the northeast and Chihuahuan desert scrub to the south, and shares some of the floral and faunal characteristics of both, Brown (1982) considered it a separate biome. Its distinctness is most clearly evident when the elevational gradient is considered:

> Much greater separation is seen by viewing all of these areas along vertical gradients. The complex gradient of elevation involves changes in climate and soils that largely control vegetation. However, since climates and soils are harder to measure directly, and vegetation reflects these differences and makes up 99% of the standing crop biomass in terrestrial ecosystems, it seems logical to use vegetation as the primary discriminator of ecosystem boundaries. (West, 1983, p. 327)

The dominant factor affecting desert vegetation is the uncertain or inadequate water supply. The harsh environmental conditions of the region have influenced the vegetative structures of plants and "shaped the processes of evolutionary development, the origins, survivals, changes, and movements by which the present flora and its distribution have been determined" (Shreve & Wiggins, 1964, p. 28). The conditions to which desert plants are exposed modifies the:

> . . . number of plants per hectare, the size and height of the plants, the number of species per square kilometer, the number of growing days and total growth per year, the size of the seed crop, the opportunities for germination, and the chances

> for survival of seedlings, as well as the operation of all the processes concerned in evolution. (Shreve & Wiggins, 1964, p. 26)

Shreve and Wiggins (1964) further point out that these climactic limitations also:

> . . . serve to prevent the full degree of development that would enable plants to form a closed covering, attain a considerable size, maintain vegetative activity throughout the year, and meet the environmental conditions without structural features or types of physiological behavior that tend to reduce their maximum performance. (p. 26)

Before extensive human habitation the grasses of the semidesert grassland were perennial bunch grasses; and even today in locations sparsely covered by shrubs or cacti and with deep soils, well protected from erosion, perennial grasses may cover relatively vast stretches of landscape. "Such purely grass landscapes, however, stand in marked contrast to most semidesert grassland cover" (Brown, 1982, p. 124).

In areas of heavy grazing and heavy to moderate rainfall, these once vast stands of bunch grasses have been significantly reduced (Kearney & Peebles, 1960), often having been replaced by such low-growing sod grasses as Curly Mesquite Grass (*Hilaria belangeri*). Where summer rainfall averages are low, bunch grasses are replaced by annual grasses. Presently there are extensive landscapes where shrubs, cacti, and forbs greatly outnumber or completely replace the grasses (Brown, 1982). (For a discussion of causes, see Chapter IV.)

Of the dominant grasses presently found in this biome, species such as fescue (Festuca), wheat-grass (Agropyron), muhly (Muhlenbergia), drop-seed (Sporobolus), and most importantly, grama (Bouteloua), which are the mainstay of the grazing industry, are considered by many to be the most valuable (Kearney & Peebles, 1960). Black Grama (Bouteloua eriopoda), together with Tobosa Grass (Hilaria mutica) are also the most characteristic. Generally, Black Grama is located on gravelly upland sites whereas Tobosa Grass usually occurs on heavier soils subject to flooding (Brown, 1982).

Both summer- and winter-active species are among these dominant species. The summer-active perennial grasses that are commonly found in the area are listed in Table 1. In some areas, particularly at higher elevations, grasses from the Plains grasslands (Table 1) are mixed with, and at times even dominate semidesert grasses. Frequently just the tougher, less palatable grasses (Table 1) are present. Lehmann Lovegrass (Eragrostis lehmanniana), a grass introduced from South Africa, is now prevalent in some areas of the biome and appears to be spreading at the expense of more palatable native grasses (Brown, 1982). Several additional species of plants, which are also replacing native grasses, are Tarbush (Flourensia cernua), Whitehorn (Acacia neovernicosa), Creosotebush (Larrea tridentata), and Mesquite (Prosopis glandulosa) (Brown, 1982).

A small assortment of seasonally abundant forbs and weeds are also abundant in either the spring or summer

TABLE 1. Southwest Semidesert Grass Species

Common Name	Latin Name	Annual/ Perennial	Elevation (in feet)	Habitat
Threeawn	Aristida divaricata	P	5,000-7,000	rocky hills
	Aristida longieseta	P	3,000 & up	dry hills, plains
	Aristida purpurea	P	up to 5,000	dry rocky hills, plains
	Aristida wrightii	P	1,000-5,000	dry plains, rocky slopes
Chino Grama	Bouteloua breviseta	P		
Spruce-top Grama	Bouteloua chondrosioides	P	2,000-5,000	rocky hills
Sideoats Grama	Bouteloua curtipendula	P		rocky hills
Black Grama	Bouteloua eriopoda	P	3,000-5,500	dry hills, mesas open ground
Slender Grama	Bouteloua filiformis	P	below 5,000	mesas, rocky foothills
Blue Grama	Bouteloua gracilis	P	1,500-2,300	
Hairy Grama	Bouteloua hirsuta	P	1,500-2,300	
Buffalo-grass	Buchloë dactyloides	P	1,500-2,300	sandy soil, rocky areas
Plains Lovegrass	Eragrostis intermedia	P	3,500-6,000	sandy prairies, rocky hills, canyons
Lehmann Lovegrass	Eragrostis Lehmanniana	P		

TABLE 1. (Continued)

Tanglehead Grass	Heteropogon contortus	P	1,000-5,500	sandy plains, rocky slopes, canyons
Tobosa Grass	Hilaria mutica	P		dry mesas, hills
Wolftail	Lycurus phleoides	P	4,000-6,500	dry rocky hills, plains
Bush Muhly or Hoe Grass	Muhlenbergia porteri	P	2,000-5,500	mesas, rocky slopes
Vine Mesquite Grass	Panicum obtusum	P	1,000-6,000	low open ground
Pappus Grass	Pappophorum vaginatum	P		
Little Bluestem	Schizachyrium scoparium			
Burrograss	Scleropogon brevifolius	P	5,000	mesas, open slopes, valleys
Plains Bristlegrass	Setaria macrostachya	A or P	2,000-7,000	dry rocky soil
Arizona Cottontop	Trichachne californica	P	1,000-6,000	mesas & rocky hills in open ground
Slim Tridens	Tridens muticus	P	up to 5,500	dry plains, gravelly slopes, rocky plains
Hairy Tridens	Tridens pilosus	P	up to 5,500	plains, rocky hills

TABLE 1. (Continued)

Fluffgrass	*Tridens pulchellus*	P	up to 5,500	mesas, rocky hills

Source: Adapted from Brown (1982), Kearney and Peebles (1960), and Shreve and Wiggins (1964).

seasons (Table 2). While many shrubs (Table 3) may be found in Southwest semidesert grasslands, their abundance varies depending on location, edaphic conditions, and grazing history. Two particularly significant native shrubs, Burroweed (*Isocoma tenuisecta*) and Snakeweed (*Gutierrezia sarothrae*), "have now replaced the grass understory over millions of acres and are indicators of former grass areas" (Brown, 1982, p. 131).

Another well-represented group, dry-tropic stem and leaf succulents and cacti (Table 4) are important in the structure of many grassland communities (Brown, 1982).

Except for One-seed Juniper (*Juniperus monosperma*) and mesquite (*Prosopis* spp.), which are found in open, well-drained soils, trees (Table 3) are not commonly found within semidesert grasslands. Those that are present are usually located in drainages.

The semidesert grassland is also home to a variety of mammals and birds. In addition to the number of endemic taxa centered in the semidesert grassland, the area also provides important resources to numerous species from adjacent scrublands and desertlands. In general, grassland and other open landscape-adapted species have fared less well than scrub-adapted species. For example, Antelope (*Antilocapra americana*) have disappeared from large areas of their former semidesert grassland range, whereas Javelina (*Dicotyles tajacu*) and Mule Deer (*Odocoileus hemionus crooki*) have increased their density and extended their range during this

TABLE 2. Seasonally Abundant Forbs and Weeds

Common Name	Latin Name	Season	Annual/ Perennial	Elevation (in feet)	Habitat
Amaranths	*Amaranthus* spp.	summer	A	1,000-8,000	sandy washes, river bottoms, irrigated land
Spiderlings	*Boerhaavia* spp.	summer	A or P	500-7,000	open sunny areas
Buckwheats	*Eriogonum* spp.	spring	A or P	500-8,500	roadsides, sandy soils, washes
Filarees	*Erodium* spp.	spring	A	up to 7,000	plains, mesas
Lupines	*Lupinus* spp.	spring	A or P	up to 9,500	dry slopes, sandy soils, mesas
Devils-claws	*Martynia* spp.	summer	A	up to 5,000	plains, mesas, roadsides
Mallows	*Sphaeralcea* spp.	spring	usually P	below 500 & up to 3,000	lower dry rocky slopes, edges of sandy washes, sandy gravelly mesas and slopes, often in timber breaks
White-mats	*Tidestromia* spp.	summer	A or P	up to 5,500	sandy soils

Source: Adapted from Brown (1982) and Kearney and Peebles (1960).

TABLE 3. Southwest Semidesert Grassland
Shrub and Tree Species

Common Name	Latin Name	Elevation (in feet)	Habitat
Catclaw Acacia	*Acacia greggii*	up to 4,500	alongside streams and washes
Wright's Lippia	*Aloysia wrightii*	1,500-6,000	common on dry rocky slopes, northern slopes, and low altitudes
Barberry	*Berberis trifoliata*	3,000	
False Mesquite	*Calliandra eriophylla*	up to 5,000	common to dry gravelly slopes and mesas
Desert Hackberry	*Celtis pallida*	1,500-3,500	foothills and meadows
Desert Willow	*Chilopsis lineans*	up to 4,000	along washes in deserts and foothills
Javelina-bush	*Condalia ericoides*		
Lotebush	*Condalia spathulata*	2,500-5,000	dry mesas and bajadas
Mormon or Mexican Tea	*Ephedra trifurca*	up to 4,500	desert or grasslands
	Ephedra antisyphilitica		arid hillsides and plains

TABLE 3. (Continued)

Turpentine Bush	*Ericamerica laricifolia*		
False Buckwheats	*Eriogonum* spp.	1,000-9,000	common to sandy soil, open pine forests, and dry locations
Snakeweed	*Gutierrezia sarothrae*	3,000-8,000	
Jimmyweed	*Isocoma heterophylla*		
Burroweed	*Isocoma tenuisecta*		
One-seed Juniper	*Juniperus monosperma*	3,000-5,000	
Allthorn	*Koeberlinia spinosa*	2,500-5,000	hillsides and mesas
Mimosa	*Mimosa biuncifera*	3,000-6,000	dry soil on mesas and rocky slopes
	Mimosa dysocarpa	3,500-6,500	arroyos and washes
Mariola	*Parthenium incanum*	2,500-6,000	dry plains and mesas, usually in caliche soil
Mesquite	*Prosopis glandulosa*		valleys and desert slopes
	Prosopis juliflora	up to 5,000 (6,000 along streams with high water tables)	

TABLE 3. (Continued)

Oaks	*Quercus chihuahuensis*	3,500-7,000	foothills
	Quercus emoryi	3,000-8,000	dry foothills and moist canyons
	Quercus grisea	3,500-7,000	foothills
	Quercus oblongifolia	up to 7,000	common in foothills of desert mountain range
	Quercus toumeyi	3,500-7,000	foothills
Littleleaf Sumac	*Rhus microphylla*	3,500-6,000	dry mesas and slopes
Threadleaf Groundsel	*Senecio longilobus*	2,500-7,500	dry plains, mesas, and slopes and along washes
Desert Zinnia	*Zinnia acerosa*		
Lotebush	*Zizyphus obtusfolia*		

Source: Adapted from Brown (1982), Kearney and Peebles (1960), and Shreve and Wiggins (1964).

TABLE 4. Dry-tropic Stem and Leaf Succulents and Cacti Common to Semidesert Grasslands

Common Name	Latin Name	Elevation (in feet)	Habitat
	Dry-tropic Stem and Leaf Succulents		
Agaves	*Agave lechuguilla*		
	Agave parryi	4,500-8,000	
	Agave parviflora	4,500	
	Agave scabra		
	Agave schotti	4,000-7,000	
Sotols	*Dasylirion wheeleri*	4,000-6,000	gravelly and rocky hillsides and mesas
	Dasylirion leiophyllum		
Ocotillo	*Fouquieria splendens*		
Bear-grasses	*Nolina erumpens*		
	Nolina microcarpa	3,000-6,500	exposed mountainsides
	Nolina texana		
Yuccas	*Yucca baccata*	3,000-8,000	usually found with pinon and juniper
	Yucca carnerosana		
	Yucca elata	1,500-6,000	grasslands and desert

TABLE 4. (Continued)

	Yucca macrocarpa		
	Yucca rostrata		
	Yucca torreyi		
	Cacti		
Turk's Head	*Echinocactus horizonthalonius*	2,500	limestone ledges and hillsides
Hedgehogs	*Echinocereus* spp.		
Rainbow cactus	*Echinocereus pectinatus* var. *rigidissimus*		
Barrel Cactus	*Ferocactus wislizenii*	up to 4,500	
Pincushions	*Mammillaria grahami*		sandy plains, gravelly slopes, rocky ridges
	Mammillaria gummifera		
	Mammillaria mainae		
	Mammillaria wrightii		
	Neolloydia erectocentra		
	Neolloydia intertexta		
Prickly Pears	*Opuntia chlorotica*	2,000-6,000	
	Opuntia phaeacantha	1,000-7,500	

TABLE 4. (Continued)

	Opuntia violacea		
	Opuntia violacea var. *macrocentra*		foothill slopes, along arroyos
Cane Cholla	*Opuntia imbricata*		
	Opuntia spinosior	1,000-5,000	
Desert Christmas Cactus	*Opuntia kleiniae*		

Source: Adapted from Brown (1982), Kearney and Peebles (1960), and Shreve and Wiggins (1964).

century. This change is due primarily to the actions of livestock, whose reduction and elimination of grasses has facilitated the invasion of woody and shrubby species by opening the stands of grass, thereby reducing, and in time eliminating, the incidence of fire (Brown, 1982).

A more detailed description of mammals associated with this habitat can be found in Donald F. Hoffmeister's (1986) Mammals of Arizona (see Bibliography).

Riparian Communities

Within the semidesert biome we need to distinguish between upland, arid communities, and wetland communities. Riparian vegetation contrasts strongly with the drought-stressed semidesert uplands. Riparian ecosystems differ from others in a number of significant features. Riparian areas are wet and frequently subjected to catastrophic water flow. They provide "corridors for the transport of water and erodible material derived from the landscape" (Brinson et al., 1981, p. 17).

The convergence of energy and material from the surrounding landscape produces the nutrient-rich soil and lush vegetation characteristic of riparian ecosystems. Their association with streams produces their characteristic linear and reticulate form. The abundance or riparian ecosystems depends on the drainage density of streams (kilometers of stream length per square kilometer of land area). Their function as corridors of material transport and their property of linearity "combine to assure that riparian ecosystems are profoundly connected to other ecosystems upstream and downstream from them" (Brinson et al., 1981, p. 17). Riparian ecosystems are also unique in the relatively large transition zones they possess in relation to their overall size. "These transition zones are the boundaries at which terrestrial and aquatic ecosystems interface and the sites of important exchanges of material and energy in the landscape" (Brinson et al., 1981, p. 17).

Riparian areas often support a variety of plant communities, ranging from mature hardwood forests to swamps and marshes (Brinson et al., 1981, p. 69). These communities may be quite distinct but frequently they are highly interspersed or occur as discrete stands within other communities (Brown et al., 1978). Most riparian environments are dominated by woody vegetation, whereas herbaceous riparian communities are more limited in extent (Brinson et al., 1981). Streamside habitats in the Southwest frequently support broad-leaved, deciduous riparian forests of Sycamore (*Platanus wrightii*), Cottonwood (*Populus fremontii*), Ash (*Fraxinus velutina*), Willow (*Salix goodingii*, *Salix bonplandiana*), and occasionally Mexican Locust (*Robinia neomexicana*), Mulberry (*Morus microcarpa*), and Choke Cherry (*Prunus virginianus*). Other vegetation common to Southwest riparian areas includes Alder (*Alnus oblongifolia*), Seepwillow (*Baccharis salicifolia*), Deer Grass (*Muhlenbergia rigens*), Mesquite (*Prosopis* spp.), Netleaf Hackberry (*Celtis* reticulata), Elderberry (Sambucus mexicana), and Burrobush (Hymenoclua monogyra) (Gori, personal communication, February 20, 1991).

Riparian communities are highly dynamic; they are subject to frequent change, replacement, and succession (Brown et al., 1978).

Because plant and animal species are sensitive to the edaphic conditions under which they develop, soil moisture is an extremely important variable and small topographic variations "can mean the difference between a waterlogged, anaerobic environment and a well drained, aerated substrate" (Brinson et al., 1981, p. 17). Only a limited number of plant species are able to withstand even brief periods of inundation while even fewer species are capable of surviving in perpetually waterlogged soil (Brinson et al., 1981).

In floodplains dramatic differences in species composition may exist with elevational variations of only a few centimeters. Additionally, "the differentiation between valley floor and upland vegetation increases with increasing drainage area" (Brinson et al., 1981, pp. 18, 56).

The species composition of floodplains includes those that are confined to wetter areas and those capable of tolerating the drier conditions of uplands (Brinson et al., 1981). The composition of vegetation in riparian soil is derived from a variety of sources and parent rock, these soils vary considerably in their depth, degree of wetness, texture, and rock fragment content (Myers, 1989). Their alkalinity varies too. Yet factors related to water have the greatest influence: regional climate, bed surface characteristics, stream hydrology, and frequency and intensity of flooding (Brown et al., 1978).

Hydroperiod is widely recognized as a key external function that influences vegetative composition and productivity. What is sometimes overlooked is that the intensity of flooding is as important as the frequency. Depending on the intensity, frequency, and timing, flooding can either enhance or stress the ecosystem. Either extreme of stagnant water or abrasive flooding can stress the biotic community and reduce productivity. In contrast, gentle seasonal flooding typically increases productivity relative to the surrounding uplands. "Over the annual cycle, benefits seem to exceed metabolic costs in natural floodplains where organisms are well adapted to the flooding regime" (Odum, 1978, p. 2).

Due to the interspersion of the many physical and biological features, riparian ecosystems are very complex habitats which are used by a relatively large number of fish and wildlife species. Without the water and vegetation provided by these areas many species would be unable to survive, or would do so at lower densities. Of all the terrestrial wildlife species that occur in a region, 10 to 80% prefer or require riparian ecosystems (Brinson et al., 1981).

In western riparian areas as many as 50% of bird species are found primarily in riparian vegetation. Riparian bird communities generally support a number of passerine species, several birds of prey, several upland game birds, and a variety of birds associated with aquatic feeding areas (Brinson et al., 1981).

Mammals are important in most riparian ecosystems, as part of food chains and for their ability to modify riparian communities. The number of mammal species present in riparian areas usually ranges from five to thirty with population densities varying greatly. A riparian community typically includes several furbearers, a few small and medium sized mammals, and at least one large mammal. While some of these species are abundant in nonriparian areas, many prefer or require riparian ecosystems. Furbearers and other water-oriented mammals are nearly completely restricted to riparian areas. Without healthy riparian ecosystems, the survival of many mammal species would be threatened (Brinson et al., 1981).

In general, most research conducted on riparian species has overlooked amphibians and reptiles. However, as these groups have become recognized as valuable indicators of environmental quality their importance in the structure of riparian ecosystems has been recognized. "Nearly all amphibians (salamanders, toads, frogs) depend on aquatic habitats for reproduction and overwintering, and many species are specifically adapted and restricted to riparian environments" (Brinson et al., 1981, p. 86). Although most reptiles are less restricted to water, many species, such as turtles and snakes, display a clear preference for riparian ecosystems (Brinson et al., 1981).

Because riparian ecosystems are suitable for riparian as well as upland species, riparian areas support a greater diversity of wildlife than nearly all non-water related habitats. Unfortunately, the extensive alteration of riparian ecosystems throughout the United States has been accompanied by declines in the populations of many plant, fish, and wildlife species. Approximately 29% of the species or subspecies listed as threatened or endangered by the U.S. Fish and Wildlife Service depend upon riparian ecosystems for their survival. Many other riparian vertebrate species and an undetermined number of invertebrate and plant species are candidates for future federal listing as threatened or endangered (Brinson et al., 1981).

CHAPTER II

HISTORICAL OBSERVATIONS

Fire

Historically, fire has played a significant role in the evolution of vegetation in the American West. It is now widely accepted that prior to the evolution of man, lightning was the major source of natural fires. Evidence indicates that in more or less regular periods, throughout the summer months, the North American continent was literally swept with waves of electrical energy (Komarek, 1965). Over long periods of time these lightning fires produced a natural mosaic of vegetation types or, as it is commonly referred to, a fire environment (Komarek, 1967; Pyne, 1982).

Lightning fires have occurred in nearly every terrestrial environment on the globe and ignited everything from Alaskan tundra to Southwest grasslands, and although "the frequency and intensity of these fires varies by region, their existence is undeniable" (Pyne, 1982, p. 10). Komarek (1967) contends that lightning fires, not humans, are the major agents responsible for the development of American grasslands. He has also observed that the highest concentration of genes that are responsible for resistance and adjustment to a fire environment are found in both the oldest tree family, the conifers, and the oldest genera of grasses, such as Aristida, Stipa, and Andropogon. He has also theorized that intensive lightning bombardment, and the resulting fire, may act as a mutagenic agent, accelerating fire adaptability in areas with frequent lightning fires (Komarek, 1964).

Dry lightning storms, thunderheads from which little precipitation reaches the ground, are the most effective fire starters. These commonly occur after dry seasons or drought, and are nearly annual spring events in the Southwest (Pyne, 1982).

If present day fire records are any indication of the frequency with which lightning fires occurred in earlier times, then they certainly were of sufficient frequency to have significantly influenced the flora and fauna of the Southwest. From 1960 to 1971, Forest Service officials in this region witnessed six separate 10-day outbreaks of lightning fire ranging from 511 to 799 per observational period (Pyne, 1982).

There are reports of fires starting from other natural ignition sources as well. These include volcanic discharges, spontaneous combustion, branches rubbing together, and even stones striking against each other. However, these sources cannot account for the widespread adaptations of natural communities to fire nor for the universal acquisition of fire by humans. Thus lightning fires have become the tool which explains the evolution of fire in both human culture and natural ecosystems (Pyne, 1982).

Lightning and fire are both natural and cultural phenomena. Not only have they been topics of science, they have long been topics of folklore and mythology.

Although fire was once perceived as a "paroxysm of a nature gone mad or as the lethal frenzy of a capricious god" (Pyne, 1982, p. 13), eventually humans began to see it as an integral part of nature and discovered the benefits that could arise from its use as a tool.

In North America early Northern Athapascan peoples were observed using fire to improve their hunting areas by burning sub-artic coniferous forests on the lower MacKenzie River (Stefansson, 1908). The use of fire as an aid to hunting was most likely brought with the Athapascan people who arrived in the Southwest sometime between the eleventh (Terrell, 1972) and fifteenth century (Worcester, 1979; Forbes, 1980; Haley, 1981) and moved on into the Sonoran Desert region in the 1600s (Dobyns, 1981; Worcester, 1979). Pyne (1982) notes that prior to white contact, fire practices in the Southwest were "controlled by Athabascan Indians and tribes related to them linguistically" (p. 515). He concludes that the fire history of this area was principally determined by the fire practices of these natives who originally migrated from the Alaskan Interior.

Pre-contact Native Americans used fire for a variety of purposes. For example, some used fire as an aid to agriculture (Komarek, 1965), some to repel the enemy (Pyne, 1982), and some in the belief that range fires would bring rain (Hough, 1926). While the use of fire was widespread among Native Americans, the Apache and Sioux used fire far more extensively than any other tribes (Pyne, 1982). As an aide to hunting, Apaches used fires and smoke to attract deer, who sought the smoke as a means of dispersing flies and mosquitoes (Pyne, 1982; Komarek, 1967). They also set snares in areas that had been intentionally burned-over, in the knowledge that the newly sprouted grass would attract small animals (Pyne, 1982).

Archeological evidence indicated that Apaches customarily employed fire-drives in hunting (Dobyns, 1981). University of California ethnographers, in a 1930 survey, documented Apaches who still remembered hunting with fire-drives. Additional evidence of the use of fires in the early history of this region is found in the journals of early expedition leaders such as Mexican Captain Antonio Comaduran. Comaduran reports an eye-witness account of Western Apaches using fire to drive game in the Sonoran Desert Margin. He records that his command waited in ambush for over an hour while the fire passed. This illustrates the expanse of territory that was burned by just a single small Apache hunting party (Dobyns, 1981).

Stewart (1956) states that "the American grasslands were regularly and frequently burned over at the time of European discovery and exploration" (p. 129). According to Dobyns (1981), "Fire constituted the principle technology that Indoamericans possessed for modifying natural environments in order to augment their food supplies" (p. 28). However, it should be noted that Hastings and Turner (1965) characterized the use of fire by Native Americans as infrequent. They state, "About the Indian's use of fire in hunting, the evidence is scanty, but the safest assumption is that he added little to the natural incidence of burning" (Hastings & Turner, 1965, p. 26). Dobyns (1981) observed that "Hastings and Turner based their conclusions on the very kind of traveler's accounts least likely to define the true frequency of Indoamerican fires" (p. 28). He therefore concludes, "Hastings and Turner consulted a class of travel journal left by participants in traveling parties which Apaches easily and systematically avoided" (Dobyns, 1981, p. 31).

This brings added importance to Captain Comaduran's 1830 records of a fire in Arivaipa Canyon that his Native American scouts said had already been burning for five days when Comaduran first saw it. It is important to note that the longest-burning human-ignited fire recorded by Comaduran was in the canyon praised frequently in recent decades as one of the most beautifully vegetated in the American Southwest. Thus the Arivaipa Canyon vegetation is not the pristine wilderness it has often been claimed to be but rather the consequence of a major 1830 burn ignited by Native Americans.

Undoubtedly the Arivaipa Canyon vegetation was also affected by Native American fires in later years (Dobyns, 1981).

The Native Americans' periodic burning of grasses removed the bulk of sun-dried biomass and encouraged the growth of new, more nutritional and palatable grass (Humphrey, 1963; Dobyns, 1981; Pyne, 1982). These fires most likely had the additional affect of suppressing the growth of shrubs and trees such as has been produced in twentieth century experiments (Humphrey, 1963; Komarek, 1965; Dobyns, 1982).

One fact frequently overlooked in discussion of the historical changes of Sonoran Desert vegetation is that it and the grasses of the Sonoran Desert and its borders have evolved over many millenia in close association with the use of fire by Native American hunters (Stewart, 1956).

"Fire has been discovered to be a regulatory mechanism in the kind of chaparral vegetation that characterized much of the mountain terrain in the headwaters of Sonoran Desert streams" (Dobyns, 1981, p. 33). Although it takes at least five years for a burned-over area to return to a climax state, the effects of fire can result in an increase of important food for deer as well as other wildlife. Following a chaparral fire numerous species of oaks sprout vigorously from the stump, and numerous seeds of many forbs and grasses, dormant since the last burn, germinate (Dobyns, 1981; Humphrey, 1963).

The most important big game animal to native hunters, deer, showed better health and an increase in size and numbers following a chaparral fire. Moreover, quail populations have been known to increase by 250%, while jackrabbit populations have increased by as much as 450% following a chaparral fire (Dobyns, 1981).

The maximum influence of fire upon the vegetation in the region east of the San Pedro River probably reached its height during the period of Apache-Northern Piman military conflict, which began about 1690 (Dobyns, 1981). As the Apaches moved into Piman territory, the Pimans engaged them in battle, at the urging of, and at times with the help of the Mexican government. The Apache's hit and run attacks caused considerable loss of life and property to the sedentary Piman Bands (Worchester, 1979). After years of debilitating battles with the raiding Apaches, the Mexican government forced the Northern Pima to evacuate the San Pedro River Valley (Dobyns, 1981). This led to immediate occupation of this area by Western Apaches, giving them a greater foothold for economic raiding.

One may infer that the Northern Piman inhabitants of the Arivaipa Canyon and San Pedro River continued the practice of setting fires as an aid to hunting well into the eighteenth century. The extensive studies of Dobyn (1981) led him to conclude that rabbit hunts with fire near riverine horticulture settlements had considerable effects on Arivaipa Canyon vegetation. He suggests that the use of fires in the taking of big game in the mountains to the east "would have favored grasses at the expense of shrubs and trees, especially when compounded by fires set during intertribal battles" (Dobyns, 1981, p. 36).

The expansion of Apache territory after 1762 subjected the region west of the continental divide, at least to the San Pedro River, to intentional Apache burning such as Comaduran's reports document (Dobyns, 1981).

With the colonial removal of Northern Pimans, the Apache-Piman conflict moved from the mountains east of the San Pedro River valley to the ranges between that stream and the Santa Cruz River. Therefore, from 1762 until the termination of Apache hostilities in 1886, most of the fires started by either Northern Pimans or Apaches, to influence the outcome of intertribal battles, were likely set in the Santa Rita, Ricon, or Santa Catalina Mountains (Dobyns, 1981).

As these intertribal battles came to a close, most if not all southern Apache tribes settled close to Spanish frontier military posts. Here they continued hunting to some extent to supplement the rations received from the Spanish. However, the location where they sought game by use of fire-drives shifted to areas much closer to the frontier military posts. Since the majority of the Apache populations hunted relatively near these outposts, it is likely that grass flourished at the expense of trees and brush in these areas. In areas such as Arivaipa Valley and Canyon and the upper Gila River tributaries, brush and trees likely gained at the expense of grasses (Dobyns, 1981).

The confinement of the Indians to restricted areas in the Southwest generally eliminated their practice of burning grasslands. At the same time, the increasing number of non-Indian settlers in the Southwest, with their large herds of livestock, resulted in considerable overgrazing. The herding and grazing practices of the Anglo ranchers left large tracts of rangeland almost devoid of grasscover (Humphrey, 1963). In the event of a lightning-caused fire, or an accidental fire, there was seldom enough grassland biomass to sustain the fire, and therefore there were few rangefires of significant magnitude. These factors, along with the Euroamerican attitude that all rangefires were destructive and therefore to be suppressed, ushered in an era of fire controls that contributed greatly to changes in rangeland vegetation.

Extensive publicity which dramatized only the harmful effects of fire perpetuated a widespread attitude that fire was "an insidious enemy rather than a useful friend" (Kozlowski and Ahlgren, 1974, p. xi). Increased population, land use, and demands for forest products, coupled with the damaging effects of man-caused fires, made it more necessary to suppress uncontrolled fires. Thus a campaign was initiated to inform the public about the need to prevent dangerous and damaging man-caused fires. However, the originators of fire prevention campaigns never intended to imply that fire had no place in nature. "Public enthusiasm for such campaigns led to overapplication, a fear of any fire in the ecosystem, and a hesitancy to recognize the dependence of some ecosystems on fire for renewal" (Ahlgren, 1974, p. 2).

In recent years a slowly increasing number of researchers have focused on the positive, as well as the negative, aspects of fire in relationship to ecology (Komarek, 1967). Their observations have led to a more accepting attitude toward burning as a method of rangeland management (Komarek, 1965; Pyne, 1982). The results of recent and current research are presented in Chapter III.

Grazing

> Now there were days when the red peoples of northern Sonora and Arizona heard a new kind of thunder, a rhythmic beating on the earth in tempos they had never known, the sound of horses' hooves. Now there were nights when the lights of campfires caught the sleek sides of the wondrous beasts tethered about the mud and mat houses, and there were new sounds in the shadows. . . . A new way of life had been revealed to the Indians of the West. (Terrell, 1972, p. 31)

As Comaduran had been impressed by the Native American's possession and widespread use of fire throughout the Southwest, so too did he find their possession of large numbers of horses impressive (Dobyns, 1981).

The horse had been reintroduced to the Southwest in the 1500s by the Spanish (West, 1983). In fact, when Terrell (1972) wrote of the historical significance of Melchior Diaz's 1539 journey along the trail to Cibola, he was not referring to the peoples' daring expedition or to the success of their journey, but rather to the fact that Diaz and his men took the first horses into the immense territory that would later become known as the American Southwest.

Many of the Southwest tribes, including the Western Apaches, soon came to recognize the value of horses and came to count them among their most treasured possessions. The importance of horses in the lives of these native people led to the first stage in a cycle of degradation that has plagued the Southwest grasslands.

Not only did the reintroduction of the horse bring about changes in the daily lives of many Native Americans, it also brought about significant changes in the vegetation of the Southwest. In fact, it became a relatively common practice for individuals to store up their wealth in the form of horses, even to the detriment of their land (Ryden, 1970; Dobyns, 1981). This practice would have far reaching effects on vast areas of Southwest rangeland.

Other Old World domesticated animals were introduced to this region as well. As the sixteenth century drew to a close, cattle, sheep, and mules could also be found in the upper Rio Grande Valley as well as other parts of the Southwest (Dobyns, 1981).

Initially the number of horses and other livestock possessed by Native Americans was relatively small. Horses, as well as cattle and mules, were frequently obtained through economic raids on pueblos, towns, and frontier outposts, probably for the purpose of immediate consumption. However, with time it appears that they began to breed horses and mules, and this resulted in a slow build-up of the population (Dobyns, 1981).

As the populations increased, the combined effects of grazing and the sharp hooves, which cut and trampled the grass roots, physically destroyed the plants and initiated the processes of erosion and vegetation change in the region (Dobyns, 1981).

Cultural differences between Mexicans and Native American tribes were responsible for the differences in the effect the livestock had upon the region. Mexicans and certain Native American tribes such as the Gila River Pima, Maricopa, and Navajos, who had learned these herding practices from the Mexicans, close-herded their livestock and corralled them at night. Although labor-intensive, this practice did have the advantage of protecting their herds and flocks from loss to raids (Dobyns, 1981).

Unfortunately, corralling concentrated the grazing, browsing, and hoof-puddling rather than to disperse it over the range. Where livestock ranged over sufficiently steep slopes, close-herding caused excessive trailing which initiated the gullying process. From his interpretation of Comaduran's journals and the writings of Hough, of Bennett, and of Spicer, and the studies conducted by McNaughton, Dobyns (1981) concluded that stock-trail gullying emerged as one of the greatest consequences of the

Western Apaches' build-up of horse populations in the early 1800s. By 1830 the number of horses kept by the Apaches surpassed that critical point at which grass-roots were trampled faster than they could recover, contributing to localized damage (Dobyns, 1981).

It is important to point out that the crucial factor in the environmental modification of arid zones is not just the consumption of biomass but the physical destruction of vegetation and the modification of crucial soil surface-bonding. Despite this important distinction, the deterioration caused by trampling was not widely recognized in the region's erosion literature until the twentieth century (Dobyns, 1981).

The journals of early American explorers have become valuable tools in the study of vegetation changes in the Southwest. Unlike early Spanish explorers, the American explorers were able to correlate their descriptions of land with precise locations, thus providing useful information for documenting such changes (Cox, Morton, LaBaume, & Renard, 1983).

The second stage of environmental degradation in the Southwest occurred with the advent of the railroad and settlement by non-Indians. Historical records indicate that grass flourished, with some exceptions, in southeast Arizona prior to 1880 (Cox et al., 1983) and by 1910, nearly all of the Southwest lands reached their maximum capacity of sheep (Stoddart & Smith, 1955). The grazing of livestock, the plowing of the floodplains, and channelization brought about significant changes in the composition of vegetation, especially in areas adjacent to major waterways. Prior to channelization of the lowlands, when the sacaton bunchgrasses were prevalent, the water entering major channels was spread over large areas and the floodwaters slowed. Sediment trapped by the sacaton enhanced the soil's fertility. Additionally, because the water moved slowly, it percolated downward, and so replenished the shallow aquifers on a yearly cycle. Discharge was thus spread over months rather than being concentrated into mere hours (Cox et al., 1983).

The third stage of environmental degradation in the Southwest began in the 1930s with the advent of Federal and private land management practices.

Although gradual, the changes in vegetation on upland ranges between 1930 and 1980 were just as extensive and harmful. With the implementation of the Civilian Conservation Corps and the Work Progress Administration and passage of the National Industrial Recovery Act in the 1930s, new grazing lands were made available. Land was divided by fences into grazing units and water was developed on upland ranges by private and Federal groups (Cox et al., 1983). Range livestock that were once confined to lowland areas were now moved to upland range areas (Cox et al., 1983).

The devestating cycle which occurred during this era is described by Cox et al. (1983) who wrote:

> From 1890 to 1980, wet periods with abundant forage were followed by overstocking, and drought periods were followed by livestock reduction. With each successive cycle, perennial grass productivity declined and the rangeland supported fewer livestock. Excessive and continuous use of perennial grass slowed plant recovery and favored invasion of woody plants. (p. 343)

The result has been that today the region is dominated by shrubby plants.

CHAPTER III

THE INFLUENCE OF FIRE ON VEGETATION

Throughout much of the history of North America, fire has had a significant influence on life and landscape. This influence has directly affected vegetation and soils and indirectly affected water resources and animals (Kirby, Lewis, & Sexson, 1988).

Early ecologists overlooked the role fire played in creating North American grassland, except for noting its detrimental effects (Vogl, 1974). In time, individuals such as Humphrey (1963) and Komarek (1965) pointed out that prior to the arrival of Euroamericans, grassland fires were not infrequent occurrences but a natural and integral part of grassland environments in North America.

Although the immediate effects of fire are common knowledge, the long-term impact upon ecosystems is not as well understood, and the scientific use of fire to ensure the perpetuation of viable plant communities and wildlife populations is in many respects a relatively recent practice (Kirby et al., 1988).

Fire is still regarded by many as a completely destructive phenomenon. Not until recent years has it become evident that complete fire exclusion, as well as uncontrolled fire, can be detrimental to certain ecosystems (Kirby et al., 1988). Research indicates that some plant and animal species even require fire for their survival and optimal health. Land managers now realize that complete fire exclusion may cause dangerous fuel accumulations which contribute to catastrophic fires, disease and insect problems, deterioration of range, decreased watershed yield, and a reduced carrying capacity for wildlife (Vogl, 1974).

As recognition of the importance of fire increases, some ecologists and conservations contend that fire could be used as a cure-all for many, if not all, of the problems relating to grassland regeneration. These individuals believe that fire should be allowed to play out its natural role in the environment with minimum interference by humans. However, this position overlooks the fact that not all ecological niches within an environment are fire-adapted (Ahlgren, 1974).

The uses of fire must be evaluated not only in terms of past effects, but also in relationship to present man-altered ecosystems and must include careful evaluation of present conditions and influences. As a result of man's use and suppression of fire, use of hergbicides and pesticides, and introduction of diseases, along with the alteration of vegetation and changes in animal populations, fire in present environments acts in a different context than it did in previous times. Therefore, the lessons learned from fires of the past may not be applicable in predicting the effects of fire on modern ecosystems. Each situation, area, and species must be considered separately, and carefully evaluated (Ahlgren, 1974).

Fire is a profound biological event and as it moves across the land it can dramatically alter the land and the living conditions of plants and animals (Komarek, 1967; Pyne, 1982). Fire is actually just a violent form of decomposition, and its role in maintaining a biochemical equilibrium is an important one (Pyne, 1982). As described by Komarek (cited in Pyne, 1982), "The earth, born in fire, baptized by lightning, since before life's beginning has been and is a fire environment" (p. 34). And as Pyne (1982) pointed out, "Fire and life . . . form a necessary kind of symbiosis" (p. 34). Fire causes chemical degradation, while the heat it releases leads to thermal degradation. Wildland fire is therefore "part of a dynamic equilibrium between the production and decomposition of biomass. Its function is to recycle, both on the microscale of nutrients and on the macroscale of the community" (Pyne, 1982, pp. 34-35). The nutrients within a biota are stored in various parts of its system--in the standing vegetative cover, in the layer of dead organic matter known as litter and duff, and in the soil. The layer of litter is the primary source of nutrients and it is this layer that is affected by fire (Pyne, 1982). Fire changes organic material to ash, thereby increasing the solubility of nutrients, making them more readily available to plants. This process increases the concentration of minerals and proteins in the plants, making them more beneficial to animals. The soluble nutrients also facilitate the decomposition of the remaining organic matter (Komarek, 1967).

The activity and the amount of nitrogen-fixing bacteria generally found in the soil also increases after a fire (Komarek, 1967). Thus legumes, which are known for their root-containing nitrogen-fixing bacteria, are a common component of many fire plant communities (Komarek, 1967; Vogl, 1974). And although a significant amount of nitrogen is released into the air by burning, burned soils rarely show a decrease in this important element (Komarek, 1967).

The dark color of the ash and other residues of incomplete combustion creates an important physical characteristic of the burned over area. A beneficial speeding up of the chemical and physical processes in the soil results from the increased absorption of radiation (Komarek, 1967).

The unusual soil parent materials characteristic of many grasslands can produce growth limitations which are frequently intensified by climatic conditions. Fortunately, the extensive range of tolerances possessed by grasses enables them to dominate very diverse habitats. These tolerances have enabled grasses to flourish in acidic to alkaline substrates, waterlogged to xeric soils, stable bedrock to shifting sands, and fertile valleys to sterile slopes. These environmental extremes often impede the growth of woody plants, thereby facilitating the establishment and growth of the better-adapted grassland species (Vogl, 1974).

The establishment and growth of grasslands is also promoted by fire. As was discussed previously, fire yields energy and imparts organic and inorganic chemicals into the components of an ecosystem--its water, soil, and air. Ecosystems have developed mechanisms by which they may cope with this flow of energy and release of chemicals. "Some organisms have adapted defensively to protect against the energy released by fire, but others have adapted so as to seize on the nutrients released by fire--and have consequently even encouraged properties promoting combustion" (Pyne, 1982, p. 34).

Plants possessing characteristics which protect them from the fire survive whereas many others do not. Thus, recurring fire can act as a natural selective agent, eliminating certain species from an area in which others thrive (Komarek, 1967). In areas otherwise capable of supporting brush and trees, recurring disturbances by fire promote the growth of grasses at the expense of woody species, even though some are quite fire tolerant and even fire dependent. Whether an area subjected to frequent fires becomes a grassland depends upon factors such as climate, soil, and fire intensity and frequency, as well as biotic factors (Vogl, 1974).

In nature, the frequency and intensity of fire vary depending upon the work required of it. "The greater the litter, the more intense the fire; the more frequently litter is built up, the more frequent the fire" (Pyne, 1982, p. 35). According to Pyne:

> Fire environments tend to exhibit certain regional and historic uniformities and to generate distictive patterns of fire. Thus different regions at different times show a characteristic mode of fire behavior. When this behavior pattern is combined with a regular or systematic source of ignition, the result is a fire regime. (p. 28)

For example, fires tend to be frequent but of low intensity in southern temperate forests, while they are episodic and of high intensity in northern forests. Within a given regime, fires are usually contagiously distributed in both time and space and do not occur in geographic isolation. Historical records indicate that most fires were actually fire complexes that were a response to regional weather conditions (Pyne, 1982).

Man can deliberately alter fire behavior by changing the quantity and quality of fuel that is available to wildfires. Humans can initiate fires under weather conditions which would not be conducive to natural ignition. They can modify burning intensities and can create ignition patterns that would not normally occur in nature. Man has the capability of encouraging extensive fires when natural conditions would have prevented the transition to large fires and of containing fires that would have escalated to large ones under natural circumstances (Pyne, 1982).

As man's willingness to tolerate small, low-intensity fires has decreased and his fire-suppression efforts have become more effective, new environmental problems have arisen. Increased suppression has disrupted nutrient recycling, thereby slowing down the growth of ecosystems, and increasing the likelihood of future fires of catastrophic intensity. Because of the increased rate and quantity of energy released, the larger fires behave more erratically. Thus fire suppression has introduced an element of instability into a system it had intended to regulate and protection efforts have become somewhat self-defeating (Pyne, 1982).

There are important differences between fires caused by lightning, which usually occur during the reproductive season of plants, and those caused by humans. Nearly all catastrophic fires have resulted from unnaturally large accumulations of highly flammable plant material. Before the fire exclusion practices of man, nature rarely allowed a build-up of such large amounts of fuel. While fires of high intensity do occur in nature, they are the exception, not the general rule (Komarek, 1967). Records for the Southwest indicate "that 78 percent of lightning fires do not exceed one-quarter acre in size and that only 3 percent exceed 10 acres" (Pyne, 1982, p. 27). Because of their size and intensity, large fires present unique problems and are responsible for the greatest damage incurred by fires. Although large fires are relatively rare, the potential exists in nearly all fires to make the transition to large ones. Moreover, the transition to a large fire may be nearly instantaneous or long-delayed. This fact has dictated much of the present-day fire-control strategy, which is to prevent ignition or suppress the fires while they are still small, and to modify the fire environment so that fires will burn with less intensity. Because low-intensity, long-burning fires are no longer tolerated, the amount of burned land has been greatly reduced over the last century (Pyne, 1982).

When evaluating the impact fire has upon vegetation, it is important to note that soil, especially damp soil, is a very good insulator, and that the greatest amount of heat from a fire is above the burning material. Thus, important parts of plants are well protected by only a small amount of soil. Only in particularly intense fires, where a large mass of fuel is burned, is the soil significantly heated. As a result, the living portions of most grasses remain undamaged while the tops of most woody species are injured or killed (Komarek, 1967).

However, the effect fire has upon vegetation is not entirely detrimental. The regrowth of fire-selected shrubs and grasses contains greater amounts of such elements as calcium, potash, protein, and phosphorus, which are necessary for the development and maintenance of life. When burned at the correct time, many grasses also produce better quality and increased quantities of seed. Several theories have been proposed to account for the mechanism by which grasses produce improved seed. The theories range from increased nutrient supply, to a reduction in diseases and insects, to release from competition with other plants, to the removal of the dead grass material as a mulch (Komarek, 1967).

Because fire generally harms woody plants while leaving the living parts of grasses undamaged, fire stimulates and sustains healthy herbaceous growth which successfully competes with the woody species for space, moisture, and light. Once the grassland is established, woody plants have difficulty invading the area, especially if the grassland is healthy and subject to recurring fires. In areas capable of supporting both grass and trees, the abundance of woody plants usually depend upon fire intensity and frequency, with the most intense and frequent fires resulting in the least number of woody species. Thus in areas with frequent fires, species composition and stand structure of trees are usually simplified, often resulting in monotypes of a fire-resistant tree or shrub. However, the reduction in the number of woody species present is frequently offset by a corresponding increase in herbs. In fact, it may even increase it by encouraging the additional growth of grasses, legumes, and other forbs, including annual plants. On occasion, though, in grasslands already low in species due to extreme conditions, fires may create monotypes. Under these conditions fire "may stimulate vegetative reproduction of the prevailing dominants to such an extent that they physically compete with and ultimately eliminate any incidental species present" (Vogl, 1974, p. 163).

Several abundant, summer-growing annual grasses in Southern Arizona are Needle Grama (Bouteloua aristidoides) and Sixweeks Threeawn (Aristida adscensionis). The herbage production of these grasses is highly positively correlated with summer rainfall, and highly negatively correlated with the abundance of competing perennial grasses and shrubs. Following a burn, annual grass production shows little change in drier years but in years of average or above-average summer rainfall, production has been higher on burned than on unburned areas (Cable, 1972).

Within semidesert grasslands, annual grasses are at a competitive disadvantage to perennial grasses and shrubs. The roots of perennial grasses are more extensive and are ready to begin using moisture with the first summer rains, while annual grasses must develop and grow their root systems each year. Annual grasses originate from seed each year. Since seeds on the ground are not damaged by most fires, burning has little direct effect on herbage production the following growing season. Indirectly, however, fire may significantly change soil moisture by eliminating or reducing the density of competing perennial species (Cable, 1972).

Vogl (1974) points out that most studies conducted on vegetational changes have not examined the effects of repeated burning on the same grassland. It is believed, though, that after a given number of fires a semi-equilibrium is reached whereby the species composition remains relatively constant; fire primarily influences the number of individuals per species. Research has also shown that for many species the number of grass stems per plant and the number of grass plants per area can be increased by burning (Vogl, 1974).

The susceptibility of vegetation to burning ranges from relatively resistant to highly susceptible, with results ranging from a drastic reduction in the density of some species to increases in the density of others. Several factors and their interrelationships can significantly influence the effect burning has on vegetation: (a) the season the fire occurs, (b) the growth habits of individual species, (c) species patterns of distribution and dispersion, (d) climatic conditions immediately prior to and following a fire, and (e) fire intensity as a reflection of available herbaceous fuel (Cable, 1972).

Approximately 40 years of research on the effects of burning on semidesert grasses and shrubs in southern Arizona has provided information useful to land managers. Plants such as Burroweed (_Aplopappus tenuisecta_) and Larchleaf Goldenweed (_Aplopappus laricifolius_), which develop from seed, can be adequately controlled by periodic burning. Sufficient herbaceous fuel is available in years of high grass production since neither Burroweed nor likely Larchleaf Goldenweed compete seriously with perennial grasses for moisture (Cable, 1972).

When sufficient fuel is present, cacti are moderately susceptible to fire. Since the competition for soil moisture between cacti and perennial grasses is minimal, adequate fuel is usually present following wet years (Cable, 1972).

The response of shrubs to fire differs according to (a) the nature of the fire, which in turn depends on the abundance of fuel, (B) the tolerance to fire of their young and mature shoots, and (c) the adaptive behaviors of their seeds. Small-sized Velvet Mesquite plants (_Prosopis juliflora_ var. _velutina_) can be kept under control through periodic burning to top-kill the trees. However, few if any mature trees can be killed in this fashion. Within dense mesquite stands, the high natural resistance to burning of mature trees and the lack of sufficient herbaceous fuel usually causes low mortality from fire (Cable, 1972).

Although a component of the Sonoran Desert vegetation, the presence of Velvet Mesquite on upland sites as the dominant woody species, very often with an understory of Burroweed (_Haplopappus tenuisectus_), is a reliable indication of former grassland (Humphrey, 1963). The lack of herbaceous fuel in dense mature stands of creosotebushes also results in their low mortality from burning. Fortunately, small creosotebushes which are beginning to invade grasslands can be controlled by fire. Because both False-mesquite (_Calliandra eriophylla_) and Velvet-pod Mimosa (_Mimosa dysocarpa_) sprout readily following a fire, neither are susceptible to burning other than top-killing (Cable, 1972).

When fuel is adequate, fire can thus be used to limit the encroachment of many shrub species and prevent their attaining seed-bearing size (Cable, 1972).

Some perennial grass species survive burning very well, because of their distribution with respect to fuel concentrations (Santa Rita Threeawn [*Aristida glabrata*]) or their ability to reseed themselves following a fire (Lehmann Lovegrass [*Eragrostic lehmanniana*]). Because of low resistance to burning and poor natural reseeding, others such as Black Grama (*Bouteloua brevisecta*) and Tall Threeawns (*Aristida hamulosa*, *Aristida ternipes*) are easily damaged by burning. Other species such as Arizona Cottontop (*Trichachne californica*), Rothrock Grama (*Bouteloua rothrockii*), and Tanglehead Grass (*Heteropogon contortus*) are only intermediately affected by fire (Cable, 1972).

In general, within the semidesert Southwest prescribed burning may reduce perennial grass production temporarily and will likely even change the relative abundance of the several species present, but it will seldom increase total production (Cable, 1972).

Throughout the past century, within given areas of the semidesert Southwest, the original vegetation has largely been eliminated. Were it not for early photographs, there would be little documentation of many of the changes that have occurred. However, knowledge of the physiological responses of grasses and woody plants to fire, early photographs, and the historical records leave little doubt of the invasion of the area by ligniphytes and that this can be attributed largely to too many grazers and too few fires (Humphrey, 1963).

CHAPTER IV

THE EFFECTS OF GRAZING ON VEGETATION

One of the most complex and encompassing issues Western rangeland managers face today is the effective management of livestock (Kinch, 1989). In recent decades, as habitat destruction has become a leading cause of plant and animal extinctions (Simmons, 1989), increased concern and attention has been focused on the role livestock grazing plays in destroying habitat. Grazing is of particular concern in the arid and semi-arid West where impacts are concentrated in fragile riparian areas (Kinch, 1989). Although the number of animals allowed in a given grazing area is carefully determined in terms of the allowable animal unit months (A.M.U.s), it is based on the assumption that cattle will evenly distribute themselves throughout the total area of each unit (Wagner, 1978). However, this is not true, especially in the semidesert Southwest, where because of the tendency of livestock to seek water, food, and shade during hot weather months, cattle tend to concentrate in and near riparian areas (Bureau of Land Management, no date) and in areas of moderate topography. This results in the trampling and overuse of vegetation along the banks and bottoms of streams and increased erosion (Wagner, 1978).

Grazing in these areas has induced both subtle and dramatic changes in vegetation, altering both the structure and composition of riparian plant communities (Medin & Clary, 1989). According to Martin and Turner (1977) human activities during the past 100 years have been so pervasive that few areas can be found which are free from impact.

Stoddart and Smith (1955) believe that had plants been appreciated as living organisms, and their requirements for proper development recognized, much of the past injury to range land might have been avoided. In their view, any grazing which inhibits such development constitutes misuse of the range.

While livestock use can be damaging, it is important to remember that vegetation and herbivorous animals coevolved throughout much of the world and that vegetation has provided sustenance to animals for millions of years. Since plants are physiologically capable of sustaining some grazing without significant damage, some use does not cause undue injury. This belief is supported by the climax state of Western ecosystems in pre-Columbian times which carried some 20-30 million wild grazing animals. Although North American vegetation did not coevolve with cattle, there is still considerable evidence that range vegetation may even be enhanced by cattle grazing (Wagner, 1978). It is only through abnormal or excessive use of vegetation that widespread destruction occurs. According to Stoddart and Smith (1955) "nature has given most plants a margin of safety which permits them to function at near-normal levels with occasional grazing and generally to survive for long periods despite severe mutilation" (p. 98).

It is important then to be able to determine the impact that grazing has on plants and the extent to which these plants may be disturbed without risking their permanent injury (Stoddart & Smith, 1955). The role grazing plays in altering vegetation ranges from subtle to dramatic. Minimal impact is not readily apparent to the untrained observer and involves the suppression of a few highly palatable species from an area. Intermediate effects are those in which the species composition is noticeably changed but the general life form of the vegetation remains unaltered. The most dramatic impacts occur when the major life form of the vegetation is altered, as when grassland is replaced by woody and shrubby species. Unfortunately, such significant vegetation change is frequently accompanied by a decline in the soil-holding capacity of the vegetation, which is followed by erosion (Wagner, 1978).

Some of the most profound vegetation changes during the past century have occurred in Arizona, New Mexico, and Texas, where the replacement of semidesert grasslands by shrublands has been accompanied in some areas by severe gullying and the loss of up to three inches of topsoil from sheet erosion. There is disagreement among authorities as to the cause of this vegetation change (Wagner, 1978). Humphrey (cited in Wagner, 1978) attributes the change to the discontinuance of prehistoric fires. Hastings and Turner (1965) believe that a combination of grazing and climate change are responsible. They explain that throughout the past century in Arizona, temperatures have increased and winter rainfall has declined. However, in response to this, several other authors point out what they believe to be a flaw in the theory when it is applied generally. They contend that Hastings and Turner are overlooking the rainfall records for New Mexico and western Texas which were part of the original Southwestern grasslands. They argue that the 100-year summer rainfall record for New Mexico, which is a summer rainfall area, shows more years of greater than average rainfall between 1882 and 1943. Thus climatic change cannot account for the changes in vegetation. Still other authors (York & Dick-Peddie, 1969; Leopold, 1924) believe strongly that the changes in vegetation can be attributed solely to livestock grazing.

Any influence, such as disease, weather, or fire, which harms certain plant species more than others will place those most damaged at a competitive disadvantage in the community. The less affected plant species generally increase at the expense of the more affected species and the plant community composition changes in ways that it would not have otherwise. Grazing by wild or domestic herbivores is a particularly effective influence in this regard. Because each herbivore is specialized to a certain extent in its feeding preferences, it suppresses the growth of its preferred plant species (Wagner, 1978).

The effect of vegetation alteration is to produce an environment for an entire fauna adapted to the new conditions. An important implication of this is that range degradation for some species may be range improvement for others. Because of centuries of selective breeding, domestic livestock seem to be more capable of shifting their diets without as much detrimental effect on their nutrition as their wild counterparts. As a consequence, when grazing alters vegetation composition, even slight or subtle changes may have a detrimental impact upon wild species whereas such changes may leave the range in quite favorable condition for domestic animals. Additionally, when a species overuses and reduces the abundance of its preferred food plant, it is forced to feed on less preferred plants. Thus, two herbivorous species which would not materially compete in a system with a variety of healthy vegetation might converge in their feeding patterns and be driven to competition when vegetation is degraded. Therefore, when excessive use by cattle converts grasslands to shrublands, the carrying capacity for their wild grazing competitors may be reduced. Thus, animals may compete directly for annually produced forage, and indirectly by altering the species makeup. However, competitive effects are not limited to competition for food. If a given animal eats vegetation required for cover by another, the effect is just as detrimental as food shortage (Wagner, 1978).

Grazing profoundly changes the structure and composition of plant communities. Heavy livestock grazing may reduce vegetation density and restrict regeneration of woody riparian species including willow (<u>Salix</u> spp.) and cottonwood (<u>Populus</u> spp.). The presence of recurring disturbances such as those caused by grazing on young aspen shoots (<u>Populus</u> spp.) is an excellent example of the changes that may occur in a community (Chaney, Elmore, & Platts, 1990).

Any grazing, whether moderate or extreme, has a measurable influence upon the metabolism of plants. The influence is complex and depends upon a number of variables: (1) the time of year in which grazing begins and ends, (b) the climate, and (c) the cyclic functions of the plants. Stockmen have long believed that lightly grazed plants are more vigorous and healthy than ungrazed plants. As early as 1955, Stoddart and Smith (1955) brought to attention research that suggested properly grazed plants are equally, or more productive, than ungrazed plants. They equated a grazed plant to one which had been carefully pruned by a skilled nurseryman.

Since the mid 1970s a debate has ensued over whether herbivory is beneficial or detrimental to plants. Researchers who support grazing's benefits theorize that through removal or damage to the productive, absorbtive, or reproductive tissue of plants, herbivory increases the net productivity, longevity, or seed production of certain species, and that such increases have resulted in an increased fitness of grazed populations and a coevolved mutualistic relationship between consumers and plants (Belsky, 1986).

Several studies can be cited in support of this contention. Belsky (1986) referred to a number of authors who have together cited over 40 different studies that support the contention that grazing or clipping may benefit the affected plant. Some of these studies have reported a partial compensation in total biomass as a result of herbivory. Although much less frequent, increases in the dry weight of aboveground grass biomass after tissue removal have also been reported.

But not all research supports the enhancement of vegetation by herbivory. After reviewing the effects of clipping and grazing on shoot yields of individual grass species, several researchers expressed concern that many of the studies reporting increased yields had inadequate controls, insufficient replication, or no statistical evaluations. They concluded that grazing and clipping do not normally result in increased aboveground production (Belsky, 1986).

Belsky (1986) studied the effects of clipping on 15 grass species from the Serengeti grasslands in Tanzania. These grasses were chosen since they are often cited as examples of species that have long coevolved with grazers and are most likely to be mutualistically associated with them. The study examined the effects of clipping on the growth of grasses when grown in a mixture with other grass species. Belsky concluded that clipping did not stimulate the plants to overcompensate by some physiological mechansim, as some have speculated, but rather released the shorter species from competition with the taller ones.

An increase in the longevity of grazed plants has been cited to support the theory that grazing increases the fitness of plants. However, even though removal of the flowering organs has increased the life span of certain monocarp species, researchers such as Belsky believe there is little evidence that the life span of perennial grasses can be increased through tissue removal (Belsky, 1986).

Another theory of the relationship between grazing and plants proposes that plant communities overcompensate for tissues removed by grazers. Numerous studies designed to test this idea measured the effects of grazing or mowing on forage yields of entire grasslnad communities. Upon completion of the studies most researchers concluded that herbivory was beneficial to plants. Belsky (1986), however, did not concur with their conclusions, citing several concerns which in her opinion reduced the value of these studies as examples of overcompensation, increased fitness, and mutualism. She noted the differences in the species composition and site characteristics between nongrazed and grazed areas, inadequate productivity measurements, treatment replication, and statistical analyses.

The disappearance of some grass species from natural communities when grazers are eliminated has been proposed as evidence that grazing increases fitness. To test this question, Belsky conducted work in the Serengeti. Results indicated that grazing does not increase plant fitness. Instead she concluded that the disappearance or persistence of species can be accounted for by the normal processes of competition and succession (Belsky, 1986).

Researchers have also studied the effects of tissue removal on the physiological processes of plants. The following mechanisms have been discussed as possible methods by which plants compensate for tissue loss:

> . . . increased photosynthetic rate of residual tissue; increased allocation of current photosynthate to new leaf blades; reallocation of substrates from elsewhere in the plant to the shoots; removal of older, less photosynthetically active tissue; increased light intensity to underlying tissues; increased tillering or shoot development following removal of apical or dominant meristems; increased water-use efficiency through reduction of transpirational surfaces. (Belsky, 1986, p. 880)

To test the role of herbivory in the evolution of these responses, closely related taxa, with different grazing histories, were compared. The comparisons involved studying the compensatory mechanisms of Agropyron smithii in two ecotypes: a heavily grazed prairie-dog town and an adjacent ungrazed area. Results showed that shoot production of plants that were partially defoliated was greater in the grazed ecotype than in the ungrazed. However, plants from both populations showed similar transpiration rates, stomatal conductances, and photosynthetic rates and responses (Belsky, 1986).

Another experiment compared the effects of partial defoliation on two *Agropyron* bunchgrass species that were similar morphologically and phenologically but different in their grazing tolerances and evolutionary histories. *Agropyron desertorum* Fisch. (*A. cristatum* [L.] gaertn.), the grazing-tolerant species, allocated more biomass to photosynthetic organs and had a relatively greater increase of photosynthetic capacity than did *Agropyron spicatum,* the grazing-sensitive species. Additional studies conducted on *Agropyron desertorum* and *Agropyron spicatum* indicated that similarly aged foliage, in underfoliated and partially defoliated plants, had similar water status and gas exchange rates. Researchers concluded that for these grass species compensatory photosynthesis is not an important ecological element of herbivory tolerance. Differences in grazing tolerances in *Agropyron smithii*, *Agropyron desertorum*, and *Agropyron spicatum* may be accounted for by differences in morphology and allocation patterns (Belsky, 1986).

A study conducted in the 1980s compared photosynthetic rates before and after clipping of three Serengeti grass species subjected to different grazing intensities. *Kyllinga nervosa* and *Hyparrheria filipendula* were from intensely and frequently grazed areas whereas *Themeda triandra* was from an area only minimally grazed. Photosynthetic rates were stimulated more by clipping in Kyllinga nervosa and Hyparrhenia filipendula than in Themeda triandra. It was concluded that a positive relationship exists between grazing history and compensatory photosynthesis (Belsky, 1986).

The conclusions drawn by Belsky (1986) are not without criticism. In an article written in response to Belsky's review of literature, McNaughton (1986) strongly disagreed with Belsky's findings and conclusions. In response to Belsky's article, McNaughton addressed the following issues. Regarding the benefit grasses derived from grazing, he refuted Belsky's claim that in an earlier article he had concluded that herbivory is beneficial to any affected plant. While he did conclude that the experiments indicated that the compensatory growth responses of plants are a significant factor in the energy flow of an ecosystem, this did not imply that the relationship between the plants and herbivores is symbiotic or that the compensatory growth of grasslands compensates fully for removal by herbivores. He further stated that it is incorrect to conclude that grazing is entirely beneficial to plants.

Belsky (1986) believed that individual plants need grazers to reduce the biomass of their competitors and prevent the build-up of senescent tissue in the community and that normal community processes of competition and succession account for the persistence or disappearance of these species.

McNaughton (1986) noted that to fully understand the response of vegetation to herbivory, one must examine processes from the individual to the ecosystem level and that grazing has effects beyond the mere reduction of plant biomass. Soil moisture is conserved through reduction of the transpiring surface, and the soil water status regulates the extent to which grazing stimulates vegetation. Additionally, nutrient recycling, through dung and urine, contributes significantly to grazing's stimulatory effect on the growth of Serengeti grasslands. McNaughton reminded us that Belsky failed to consider the contribution that such effects of herbivory play in vegetation productivity, except to note that the water-use efficiency of laboratory plants is not increased by defoliation. He explained that several field experiments have, in fact, documented increased grain yields when the plants were defoliated under arid conditions. This, McNaughton explained, certainly demonstrated an increased fitness in annuals as a result of herbivory.

One of the few points that Belsky and McNaughton agreed upon is the interaction between herbivory and competitive success. They are in agreement that clipping and grazing frequently give shortgrasses a competitive edge over tallgrasses (McNaughton, 1986).

It is obvious then that the debate over the benefits of herbivory is one that is complicated and unresolved.

That grasses have coevolved with grazers and have adapted to being grazed is beyond debate. Since grazers and browsers help to create and maintain the open environment in which grasses thrive, many might argue that they are essential components of grasslands. Certainly the "low nutritional quality, high concentrations of silicates, sharp awns, and vegetation modes of reproduction" (Belsky, 1986, p. 885) of grasses is evidence of a lengthy and intimate association with grazers, albeit an antagonistic one.

Whether herbivory proves to be harmful, beneficial, or perhaps a little of both, there is knowledge that enables managers to reduce potential damage to rangelands from grazing. Stoddart and Smith (1955) believed that rangelands are particularly susceptible to damage from grazing when it occurs at a frequency or intensity that prevents sufficient photosynthetic tissue from remaining on the plant, when root reserves are at their minimum, during dry weather when plants are unable to replace leafage, and in early spring when because of excessively wet soil ranges are particularly susceptible to trampling damage. Trampling compacts the soil, making it a poor absorber of precipitation, and inhibits normal root development, sometimes resulting in roots only half their usual length. Additionally, hard compacted soil makes it difficult for seedlings to become established.

Knowledge of plant food reserves is also important for when reserves are low, grazing damage increases. Food storage reaches a minimum during periods of growth and reaches a maximum during late autumn when the growth rate is slow. Decreased growth rate also coincides with cold or dry weather and with the formation of flowers and fruit. Managers wanting to rid given areas of undesirable plants can take advantage of these cycles as well, for plants are also easily killed by mowing or spraying during these times (Stoddart & Smith, 1955).

Although under certain conditions animals can be quite damaging to rangelands, they can also at times be quite beneficial. Animals facilitate seed dispersal, loosen surface soil, and cover seeds that have accumulated there. Because of these beneficial effects there have been times when total protection of the range from livestock has failed to result in vegetation recovery (Stoddart & Smith, 1955).

Influences such as weather, disease, or fire, which harm certain plant species more than others, will place those most affected at a competitive disadvantage within the community. The less affected species usually increase at the expense of those more affected, thereby changing the composition of the plant community (Wagner, 1978).

All herbivorous animals, whether domestic or wild, are specialized to some extent in their dietary needs or their feeding niches. According to Wagner (1978):

> They differ in the structure and physiology of their gastrointestinal systems, in the microorganisms which inhabit those systems, in their abilities to break down or tolerate the protective

> chemical compounds which plants synthesize in their tissues to discourage herbivores, and in other ways. (p. 122)

Therefore, each animal species is able to feed on only a part of the total vegetation available to it.

Because each herbivore is relatively specialized in its feeding choices, it usually suppresses the growth of its preferred species. Consequently, as grazing alters the vegetation composition, even slight changes can result in damaging effects for wild herbivorous species, even though the range may still be well suited to domestic animals (Wagner, 1978).

CHAPTER V

THE INDIRECT EFFECTS OF GRAZING AND FIRE THROUGH IMPACTS UPON VEGETATION

Riparian health refers to the stage of geomorphic, hydrologic, and vegetative development as well as the degree of structural integrity of a riparian area (DeBano & Schmidt, 1989b). The interrelationship between riparian health and watershed condition has been well established through historical documentation (Dobyns, 1981). Historical accounts of numerous Southwest riparian areas characterize them as stable, aggrading stream networks, containing a considerable quantity of organic debris and supporting large populations of beaver (DeBano & Schmidt, 1989a).

"The balance between watershed condition and riparian health represents a dynamic interrelationship between runoff and erosive forces where precipitation forces producing runoff are counteracted by vegetative, geomorphic and structural resistance" (DeBano & Schmidt, 1989b, p. 46). When in equilibrium, the system maintains a level of stability by dissipating the potential energies that would otherwise cause rapid changes. A variety of factors acting together throughout the watershed promote this resistance to change. The most significant factor in maintaining this equilibrium is vegetation (DeBano & Schmidt, 1989b).

Because a healthy watershed/riparian system is resilient, much of the potential runoff produced by a storm immediately infiltrates the soil, thereby regulating the flow. When water reaches the limits of the channel's capacity, it overflows onto the floodplain where vegetation and debris provide resistance and act as filters, or traps, for sediment. This overflow process also facilitates the germination and establishment of certain riparian plant species (De Bano & Schmidt, 1989a).

The sensitive hydrologic interrelationships that exist are easily upset by land uses such as livestock grazing and wildfires which remove vegetation and contribute to soil compaction. These uses often create "(1) excessive runoff, (2) increased frequency and magnitude of peak streamflow, (3) steeper stream slope, (4) excess tributary sediment, and (5) accelerated bank erosion" (DeBano & Schmidt, 1989b, p. 48).

Floodplains are unique in their ability to change rapidly. Channels must constantly adjust their capacity in response to the natural episodes of large, infrequent floods and variations in sediment load. Topographic features such as meander scrolls, oxbow lakes, and abandoned channels are examples of this process (DeBano & Schmidt, 1989b).

When disturbances to the existing balance occur, the riparian ecosystem undergoes a series of adjustments in erosional and depositional processes until a new balance is established. Once the new balance is achieved it is maintained until new changes that exceed the elastic limit of the system occur, once again setting the adjustment process in motion (DeBano & Schmidt, 1989a).

Due to the semiarid climate and wide variations in annual precipitation, Southwest riparian communities and their associated watersheds are particularly vulnerable to overuse (DeBano & Schmidt, 1989b).

Throughout the past century, misuse of watersheds and their associated riparian ecosystems has frequently altered the balance between watershed condition and riparian health. In upland areas, widespread overgrazing on rangelands has frequently decreased or destroyed the plant cover, thereby decreasing the infiltration of water into the soil. Logging practices have also contributed to unsatisfactory watershed conditions in forested areas. Prior to timber harvesting, soil permeabilities are normally high. Following harvesting, surface erosion can accelerate as a result of vegetation removal, modifications to the soil structure, and reduced infiltration rates resulting from compaction. When vegetation removal is excessive, overland waterflow may increase and infiltration may be reduced. When waterflows exceed the channel's capacity, the channel is downcut and enlarged. The resulting expanded drainage networks promote undesirable flash runoffs and increased sedimentation. The incised channels may intercept and drain existing water tables.

Flows sustained by the ground water supported extensive riparian vegetation as well as populations of beaver, whose dam construction further regulated flows. In addition to playing a dynamic role in the food chain, beavers frequently expanded the areas which support favorable riparian vegetation (DeBano & Schmidt, 1989a).

Natural flood events and the presence of surface water strongly influence the microenvironments to which plant and animal populations adapt (Brinson et al., 1981). Riparian habitats are critical to wildlife, particularly in the arid Southwest where many, if not most, of the animals depend upon riparian areas for their survival (Bureau of Land Management, no date). Suitable habitat is the most critical factor affecting the abundance and distribution of wildlife species. A knowledge of habitat requirements is useful in predicting the effects of various habitat alterations on animal populations. The degree to which the composition of a community changes depends in part upon the specificity of the habitat requirements of each species. Those with narrow habitat-usage patterns are more vulnerable to changes in land-use practices than those inhabiting a greater diversity of environments (Best, Stauffer, & Grier, 1978).

Most alterations of riparian ecosystems and associated stream channels affect at least one of three properties--water delivery and geomorphology, physiological stress, and biomass removal. Alterations relating to modifications in water delivery or geomorphology result in severe, long-lasting changes which have a low probability of recovery. Depending on the magnitude and frequency at which they are imposed, physiological stress and biomass removal have a greater chance of being repaired through natural ecosystem processes such as succession or through mitigation techniques (Brinson et al., 1981).

Healthy and productive riparian areas require a balance between the riparian ecosystem, including the associated channels, and the hydrologic and geomorphic processes operating in its tributaries. Unhealthy riparian areas reflect poor watershed conditions. Lag times between changes on watershed slopes and those in riparian areas cause other combinations of riparian health and watershed conditions, although this is much less common. For example, concentrated overgrazing in a riparian area may lead to an unhealthy riparian zone within a watershed that is in good condition. However, the misuse of riparian areas will in time usually lead to channel incisions and gully development throughout the surrounding watershed. The rarest situation is to have a healthy riparian area surrounded by a watershed that is in poor condition (DeBano & Schmidt, 1989a).

A number of revegetation projects and land treatments aimed at improved hydrologic and hydraulic conditions have been used to stabilize existing riparian ecosystems or create new ones. The improvement of watershed conditions is an essential first step in restoring the balance between riparian health and watershed conditions. Rehabilitation will likely be ineffective in stream systems with poor or deteriorating watershed conditions.

Attempts to rehabilitate riparian areas and their surrounding watershed should begin by establishing the causes of degradation and determining the stage of channel evolution. Problem identification should include an assessment of the relationship of land and channel systems to current and past land-use practices or catastrophic events such as wildfires. The establishment of a stable riparian/ watershed balance is based on two types of action programs: (a) sideslope improvement, and (b) channel stabilization to reduce erosion and downcutting (DeBano & Schmidt, 1989a).

These general action programs allow managers to determine treatments and implement a wide variety of projects. Rehabilitation treatments may range from simple changes in the management of grazing, timber harvesting practices, or planting and revegetation activities, to more intricate and expensive projects involving mechanical sideslope treatments or construction of channel structures (DeBano & Schmidt, 1989a).

Frequently, plant cover can be restored with simple improvements in grazing management. However, "expanded channel networks may continue to erode and transmit unfavorable flows rapidly" (DeBano & Schmidt, 1989a, p. 6). This reinforces the need to rehabilitate slopes and surfaces in addition to improving vegetation cover through grazing management or reseeding. Providing plants with an opportunity for regaining vigor and establishing a denser ground cover is one of the simplest and most cost efficient ways of improving watershed conditions on rangelands. By increasing plant cover more water is able to infiltrate the soil mantle where it slowly moves through the soil before reappearing downslope as channel flow.

Where plant cover cannot be improved by grazing management alone, grass seeding and mechanical treatments may be necessary to retain water and aid in vegetation establishment. However, these treatments may require several years of rest from grazing to allow plants to become well established before grazing is resumed. Southwestern riparian ecosystems are particularly sensitive to overuse because they are subjected to a wide variation in annual precipitation (DeBano & Schmidt, 1989a).

It is important for land managers to realize that long periods of time are required for changes in watershed sideslopes or riparian ecosystems to take hold. This is particularly true in the Southwest, where impacts upon riparian areas and watersheds are amplified by limited water supply. The cost of restoring riparian ecosystems and their watersheds to their original conditions after damage should be commensurate with the long-term value of the work these ecosystems perform if permitted to function naturally (Brinson et al., 1981).

CHAPTER VI

MANAGEMENT

During the past decade a new perspective on land use has emerged in which land is increasingly viewed as a resource system with the capacity to produce a number of goods and services. As with any management plan, determining the management goals for a piece of land involves choices about the products it should produce or functions it serves. Achieving a desirable balance requries comprehensive understanding of the alternatives, trade-offs, and consequences involved. Frequently this involves recognizing and weighing all of the interests, desires, and values of society and of special interest groups (Wagner, 1978).

A heightened awareness of the value and potential productivity of our natural resources, of rising resource costs, and of the potential for resource scarcities has also emerged. Increasingly, land managers and the public in general have supported efforts to manage and protect resources in order to maintain productivity and enhance environmental quality (Cutler, 1979).

Change, a natural and inevitable part of all ecosystems, is a widely recognized feature of semidesert vegetation. Both nature and man can bring about dramatic changes of either a temporary or relatively permanent nature. Man may induce major or minor changes through either intentional or inadvertent activities such as grazing of livestock, management of watersheds, and supression of fire (Martin & Turner, 1977). Fortunately, most natural changes are at a rate and on a scale that is not disastrous to mankind. However, man's activities frequently serve to "accelerate the rate and widen the scope of natural changes in ways that lower the potential of the environment to sustain him" (Sears, 1956, p. 481).

Of all man's environmental modifications, his influence upon native vegetation has been among the most conspicuous (Graham, 1956). The overuse and abuse of Southwest wildlands through fire supression and livestock grazing, has severely depleted vegetation and resulted in increased soil erosion, sedimentation of streams, reduction of valuable riparian habitat, and alteration of the balance between riparian ecosystems and their associated watersheds (DeBano & Schmidt, 1989b). By the early 1920s Leopold (1924) recognized that the Southwestern landscape had been dramatically altered by overgrazing. Because livestock have been a major part of the Western economy during the past century, the values and desires of the industry have long influenced the region's attitudes and land use decisions. For many years a significant number of public land managers have been concerned almost exclusively with range management of this industry (Wagner, 1978). Even the fire practices of the region have, in part, been dictated by the livestock industry (Pyne, 1982). Unfortunately, the superimposition of land uses on an intense natural regime of lightning fire (Pyne, 1982) and the semidesert characteristics of low and poorly distributed precipitation and shallow and unstable soils have only served to intensify the detrimental effects of grazing and fire suppression. After years of accelerated erosion the land is frequently no longer capable of supporting native vegetation (Stoddart & Smith, 1955).

For some time now, land managers have also been concerned about riparian ecosystems and the misuse and deterioration of watersheds. Causes of riparian ecosystem degradation are numerous and vary in importance between regions. Some of the most dramatic losses of riparian lands have occurred in the Southwest, where the demands for water and productive farmlands have resulted in the loss of important bottom land vegetation (Brinson et al., 1981).

Among the more important attributes that distinguish riparian ecosystems from upland areas are the abundance of water and rich alluvial soils. Because of these special attributes and life-supporting features, humans have long made riparian areas focal points for urban settlements, agriculture, and recreational use. Throughout the Southwest, riparian communities continue to be subjected to destructive practices (Brinson et al., 1981). When human activities modify the natural temporal and spatial pattern of water flow, the essential features upon which riparian ecosystems depend are endangered. The consumptive use of water, stream alterations, and manipulation of streamflows are among the many uses that are incompatible with the natural ecological functions of riparian areas. Although the impacts are more subtle, pollution, grazing, and recreational use can cause chronic problems that reduce the functional quality of remaining areas. Many of these alterations can be considered acute since they prevent riparian systems from providing valuable life support services to society, such as water purification, flood water control, and the production of quality timber, fish, and wildlife (Brinson et al., 1981).

As of 1981 no single comprehensive inventory of riparian ecosystems had been taken in the United States to determine the original amount of riparian land and the proportion of that area currently supporting natural riparian communities. Overall, 70% of riparian ecosystems have been altered, but it is evident that losses at some locations far exceed this estimated national average. Natural riparian communities now comprise less than 2% of the land area in the United States and only a limited portion of the original floodland area is still occupied by native vegetation. In relation to other vegetation types, riparian ecosystems are among the most severely altered landforms in the nation. Without long-term protection the remaining areas are threatened by further direct losses or by the impacts of man's activities in adjacent aquatic and upland ecosystems. Despite their ecological importance, humans have continued to alter, develop, and destroy riparian ecosystems. Certainly one can conclude that these important areas have not received adequate attention or protection (Brinson et al., 1981).

Although man's impact upon vegetation is frequently destructive, with careful planning and control man can positively influence the composition of plant communities within the limits of the natural factors influencing the area (Graham, 1956). In recent times science has provided mankind with the capability of monitoring environmental quality, therby allowing him to determine ways in which cultural values and behavior should be modified to insure a healthy environment (Sears, 1956). As concern over Western rangeland conditions has grown in this century, increased attempts have been made to rehabilitate ranges which have changed from their earlier, more productive state (Wagner, 1978).

If freed of disturbance, ecosystems can frequently rehabilitate themselves through natural processes. In consideration of this, Bureau of Land Management and Forest Service personnel have reduced livestock numbers on many national lands and employed a rotating grazing schedule that allows for periods of rest and recovery. Unfortunately, the natural recovery process in semiarid lands may take at least a century (Wagner, 1978). Therefore numerous techniques--mechanical, fire, and herbicidal removal of woody species--have been employed to artificially rehabilitate rangelands. After these treatments the areas may be left to natural grass regrowth. When grasses and other preferred species are too few to adequately reproduce, artificial seeding may be employed to speed up the process. During the past 40 to 50 years, millions of acres throughout the West have been treated in this fashion. Initial rehabilitation programs focused almost exclusively on livestock, and seeding was usually limited to grass. In an attempt to provide for deer as well as livestock more recent rehabilitation programs have included shrub and forb seeds with the grass (Wagner, 1978).

For some time now, land managers have also been aware of the need for rehabilitation programs to correct misused and deteriorated watersheds throughout the Western United States. This awareness has led to the widespread implementation of watershed rehabilitation programs and projects designed to reverse the processes responsible for initially destroying the riparian areas (DeBano & Schmidt, 1989a).

The significance of alterations to riparian ecosystems lies in the relative irreversibility of man's impacts. While in theory many of the water resource and agricultural developments can be reversed, the economic expense of such a task is currently prohibitive. In addition, most organizations lack the needed incentives. Most importantly, riparian area rehabilitation requires restoration of complex natural hydrologic regimes. However, since conversion of flood-prone areas to other uses frequently involves permanent drainage or impoundment, the possibility for mitigation and recovery through natural succession are virtually nonexistent (Brinson et al., 1981).

Riparian ecosystems, however, are not immune to management. In light of the numerous alterations that have already occurred in many watersheds, sound management may be the preferred alternative. The types of alterations that can be sustained and the extent to which alterations can be made without threatening the carrying capacity of riparian ecosystems needs to be understood. It is essential to understand the functions, properties, and importance of energy and material flows of riparian ecosystems in order to design and implement proper management decisions (Brinson et al., 1981).

In general, management and rehabilitation has been focused on improving plant cover and reducing runoff and erosion through revegetation techniques or engineering structures or both. Generally, these projects have reversed the processes responsible for initially destroying the riparian areas. The result has been to produce a new equilibrium that enables the riparian/watershed system to respond to a wider range of storm events and flow fluctuations without causing drastic or irreversible changes in the relative blaance (DeBano and Schmidt, 1989a).

Determining the most suitable approach to rehabilitation can itself be a task of significant magnitude. After discussions with range managers and a careful review of literature, the Bureau of Land Management concluded that because of the large number of sites, diverse resource conditions, and variety of human perspectives involved, there is no single formula that will identify the type of management plan that will be best for any livestock operation or fire management plan. Nor has any given management system conclusively shown to result in consistent improvement of degraded western rangeland, upland, or riparian areas. Thus each management plan for grazing, as well as fire suppression, whose goal is to minimize soil, vegetation, and water quality impacts, must be carefully and specifically designed for each area and tailored to the conditions, problems, and objectives on a site-specific basis. The management plan must consider the soil and vegetation capabilities, water quality considerations, and livestock and wildlife requirements of each area (Kinch, 1989).

REFERENCES

Ahlgren, I. F. (1974). Introduction. In T. T. Kozlowski & C. E. Ahlgren (Eds.), *Fire and ecosystems* (pp. 1-5). Madison, WI: University of Wisconsin.

Belsky, A. J. (1986). Does herbivory benefit plants? A review of the evidence. *The American Naturalist*, *9*(6), 870-892.

Best, L. B., Stauffer, D. F., & Geier, A. R. (1978). Evaluating the effects of habitat alteration on birds and small mammals occupying riparian communities. *Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems* [Symposium] (pp. 117-124) (General Technical Report WO-12). Washington, DC: USDA Forest Service.

Brinson, M. M., Swift, B. L., Plantico, R. C., & Barclay, J. S. (1981). *Riparian ecosystems: Their ecology and status*. Kearneysville, WV: U.S. Department of the Interior, Fish and Wildlife Service.

Brown, D. E. (1982). Semidesert grassland. In D. E. Brown (Ed.), *Desert plants, Special issue: Biotic communities of the American Southwest, United States and Mexico*, *4*(1-4), 123-131.

Brown, W., Brinson, M. M., & Lugo, A. E. (1978). Structure and function of riparian wetlands. *Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems* [Symposium] (pp. 17-31) (General Technical Report WO-12). Washington, DC: USDA Forest Service.

Bureau of Land Management. (n.d.). *Riparian area management*. Billings, MT: Author.

Bureau of Land Management. (1989). *Safford district resource management plan--Environmental impact statement*. Safford District, AZ: Author.

Cable, C. R. (1972). Fire effects in southwestern semidesert grass-shrub communities. *Proceedings of the Eleventh Annual Tall Timbers Fire Ecology Conference* (pp. 109-127). Tallahassee, FL: Tall Timbers Research Station.

Chaney, E., Elmore, W., & Platts, W. S. (1990). Livestock grazing on western riparian areas. Eagle, ID: Northwest Information Center.

Cox, J. R., Morton, H. L., LaBaume, J. T., & Renard, K. G., (1983). Reviving Arizona's rangelands. Journal of Soil and Water Conservation, 38, 342-345.

Cutler, M. R., (1979). Overview framework of range management. Rangeland Policies for the Future [Symposium] (pp. 1-5) (General Technical Report WO-17). Tucson, AZ: USDA Forest Service.

DeBano, L. F., & Schmidt, L. J. (1989a). Improving southwestern riparian areas through watershed management (General Technical Report RM-132). Fort Collins, CO: USDA Forest Service.

DeBano, L. G., & Schmidt, L. J. (1989b). Interrelationship between watershed condition and health of riparian areas in southwestern United States. In R. E. Gresswell, B. A. Barton, & J. L. Keshner (Eds.), Practical Approaches to Riparian Resource Management [Educational workshop] (pp. 45-52). Billings, MT: U.S. Bureau of Land Management.

Dobyns, H. (1981). From fire to flood: Historic human destruction of Sonoran Desert riverine oases. Socorro, NM: Ballena.

Forbes, J. D. (1980). The Indians in America's past. Englewood Cliffs, NJ: Prentice Hall.

Graham, E. H. (1956). The re-creative power of plant communities. In W. L. Thomas, Jr. (Ed.), Man's role in changing the face of the earth (pp. 677-791). Chicago: University of Chicago.

Haley, J. L. (1981). Apaches, a history and culture portrait. Garden City, NY: Doubleday.

Hastings, J. R., & Turner, R. (1965). The changing mile: An ecological study of vegetation change with time in the lower mile of an arid and semiarid region. Tucson: University of Arizona.

Hoffmeister, D. F. (1986). Mammals of Arizona. Tucson: University of Arizona.

Hough, W. (1926). Fire as an agent in human culture. Washington, DC: Government Printing Office.

Humphrey, R. R. (1963). The role of fire in the desert and desert grassland areas of Arizona. Proceedings of the Second Annual Tall Timbers Fire Ecology Conference (pp. 45-61). Tallahassee, FL: Tall Timbers Research Station.

Kearney, T. H., & Peebles, R. H. (1960). Arizona flora. Berkeley: University of California.

Kinch, G. (1989). Riparian area management: Grazing management in riparian areas (Technical Reference 1737-4). Denver: Bureau of Land Management.

Kirby, R. E., Lewis, S. J., & Sexson, T. N. (1988). Fire in North American wetland ecosystems and fire-wildlife relations: An annotated bibliography (Biological Report 88[1]). Washington, DC: Fish and Wildlife Service.

Komarek, E. V., Sr. (1964). The natural history of lightning. Proceedings of the Third Annual Tall Timbers Fire Ecology Conference (pp. 139-183). Tallahassee, FL: Tall Timbers Research Station.

Komarek, E. V., Sr. (1964). Fire Ecology--grasslands and man. Proceedings of the Fourth Annual Tall Timbers Fire Ecology Conference (pp. 143-170). Tallahassee, FL: Tall Timbers Research Station.

Komarek, E. V., Sr. (1967). Fire and the ecology of man. Proceedings of the Sixth Annual Tall Timbers Fire Ecology Conference (pp. 143-170). Tallahassee, FL: Tall Timbers Research Station.

Kozlowski, T. T., & Ahlgren, C. E. (1974). Preface. In T. T. Kozlowski & C. E. Ahlgren (Eds.), Fire and ecosystems (pp. xi-xii). Madison, WI: University of Wisconsin.

Leopold, A. (1924). Grass, brush, timber, and fire in southern Arizona. Journal of Forestry, 22(6), 1-10.

Martin, S. C., & Turner, R. M. (1977). Vegetation change in the Sonoran Desert region, Arizona and Sonora. Journal of the Arizona Academy of Science, 12(2), 59-69.

McNaughton, S. J. (1986). On plants and herbivores. The American Naturalist, 128(5), 765-770.

Medin, D. E., & Clary, W. P. (1989). Small mammal populations in a grazed and ungrazed riparian habitat in Nevada (Research Paper INT-413). Washington, DC: USDA Forest Service.

Meyers, L. H. (1989). Riparian area management: Inventory and monitoring riparian areas (Technical Reference 1737-3). Denver: Bureau of Land Management.

Odum, E. P. (1978). Ecological importance of the riparian zone. Strategies for Protection and Management of Floodplain Wetlands an Other Riparian Ecosystems [Symposium] (pp. 2-5) (General Technical Report WO-12). Washington, DC: USDA Forest Service.

Platts, W. S., & Nelson, R. L. (1989). Characteristics of riparian plant communities and streambanks with respect to grazing in northeastern Utah. In R. E. Gresswell, B. A. Barton, & J. L. Keshner (Eds.), Practical Approaches to Riparian Resource Management [Educational workshop] (pp. 73-81). Billings, MT: U.S. Bureau of Land Management.

Pyne, S. (1982). Fire in America. Princeton, NJ: Princeton University.

Riparian Habitat Task Force. (1990). Handbook of riparian measurements. Unpublished manuscript. State of Arizona.

Ryden, H. (1970). America's last wild horses. New York: E. P. Dutton.

Sears, P. B. (1956). The processes of environmental change by man. In W. L. Thomas, Jr. (Ed.), Man's role in changing the face of the earth (pp. 471-584). Chicago: University of Chicago.

Shreve, F. (1917). A map of the vegetation of the United States. Geographical Review, 3, 119-125.

Shreve, F., & Wiggins, I. L. (1951). Vegetation and flora of the Sonoran Desert: Vol. 1. Vegetation of the Sonoran Desert. Washington, DC: Carnegie Institute of Washington.

Shreve, F., & Wiggins, I. L. (1964). Vegetation and flora of the Sonoran Desert: Vol. 1. Vegetation of the Sonoran Desert. Stanford, CA: Stanford University.

Simmons, I. G. (1989). Changing the face of the earth. New York: Basil Blackwell.

Stefansson, V. (1908). My life with the Eskimo. New York: Macmillan.

Stewart, O. C. (1956). Fire as the first great force employed by man. In W. L. Thomas, Jr. (Ed.), Man's role in changing the face of the earth (pp. 115-133). Chicago: University of Chicago.

Stoddart, L., & Smith, A. (1955). Range management. New York: McGraw-Hill.

Terrell, J. U. (1972). Apache chronicle. New York: World Publishing.

Vogel, R. (1974). Effects of fire on grasslands. In T. T. Kozlowski & C. E. Ahlgren (Eds.), Fire and ecosystems (pp. 139-194). New York: New York Academic Press.

Wagner, F. H. (1978). Livestock grazing and the livestock industry. In H. P. Brokaw (Ed.), Wildlife and America (pp. 121-144). Washington, DC: Council on Environmental Quality.

West, N. E. (1983). Overview of North American temperate desert and semi-deserts. In N. E. West (Ed.), Ecosystems of the world: Vol. 5. Temperate deserts and semideserts (pp. 321-330). New York: Elsevier Scientific.

Worchester, C. E. (1979). The Apaches--Eagles of the southwest. Norman, OK: University of Oklahoma.

York, J. C., & Dick-Peddie, W. A. (1969). Vegetation changes in southern New Mexico during the past hundred years. In W. G. McGinnies & B. J. Goldman (Eds.), Arid lands in perspective (pp. 157-166). Washington, DC: The American Association for the Advancement of Science.