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Assessment of Grassland Ecosystem Conditions in the Southwestern United States

Volume 1

Editor
Deborah M. Finch

Abstract

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This report is volume 1 of a two-volume ecological assessment of grassland ecosystems in the Southwestern United States. Broad-scale assessments are syntheses of current scientific knowledge, including a description of uncertainties and assumptions, to provide a characterization and comprehensive description of ecological, social, and economic components within an assessment area. Volume 1 of this assessment focuses on the ecology, types, conditions, and management practices of Southwestern grasslands. The second volume, due to be published in 2005, describes wildlife and fish species, their habitat requirements, and species-specific management concerns, in Southwestern grasslands. This assessment is regional in scale and pertains primarily to lands administered by the Southwestern Region of the USDA Forest Service (Arizona, New Mexico, western Texas, and western Oklahoma). A primary purpose of volume 1 is to provide information to employees of the National Forest System for managing grassland ecosystems and landscapes, both at the Forest Plan level for Plan amendments and revisions, and at the project level to place site-specific activities within the larger framework. This volume should also be useful to State, municipal, and other Federal agencies, and to private landowners who manage grasslands in the Southwestern United States.

Key words: grasslands, ecological assessment, Southwestern United States, ecosystem conditions, Arizona, New Mexico, Texas, Oklahoma

Acknowledgments

This project is a collaborative effort between the Rocky Mountain Research Station and the Southwestern Region of the U.S. Department of Agriculture, Forest Service. In early 2000, a Southwestern Grassland Ecosystem Sustainability Team was formed to determine the assessment approach and general content. By 2001, the team had evolved into a core group that finalized the assessment topics and authored the report. This current work is volume 1 of a two-volume report.

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Editor's Note

Although this report may be used as input in processes initiated under the National Environmental Policy Act (NEPA), National Forest Management Act (NFMA), and other applicable laws, it is not a decision document, does not allocate resources on public lands, and does not make recommendations to that effect. The information in this report is general in nature rather than site-specific. The opinions expressed by the authors do not necessarily represent the policy or position of the U.S. Department of Agriculture and the Forest Service.

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Cover photos clockwise from upper left: A grassland-juniper ecotone, Bernalillo watershed, Cibola National Forest, New Mexico (photo by Rosemary Pendleton). Urban development of native grassland near Flagstaff, Arizona (photo by John Yazzie). Prescribed fire on the Sevilleta National Wildlife Refuge, New Mexico (photo by Burt Pendleton). Introduced bison coexist with reintroduced black-tailed prairie dogs on the Armendaris Ranch, New Mexico (photo by Paulette Ford).

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Contents

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Chapter 1:

Purpose and Need for a Grassland Assessment

Purpose

This report is volume 1 of an ecological assessment of grassland ecosystems in the Southwestern United States, and it is one of a series of planned publications addressing major ecosystems of the Southwest. The first assessment, General Technical Report RM-GTR-295, *An Assessment of Forest Ecosystem Health in the Southwest* (by Dahms and Geils, technical editors, published July 1997), covered forested ecosystems. Given the complexities of grassland ecology and the increasing number of challenges facing grassland managers, the USDA Forest Service Southwestern Region, in partnership with the agency's Rocky Mountain Research Station, focused on grasslands in its second assessment. The assessment is regional in scale and pertains primarily to lands administered by the Southwestern Region (Arizona, New Mexico, Texas, and Oklahoma).

Broad-scale assessments are syntheses of current scientific knowledge, including a description of uncertainties and assumptions, to provide a characterization and comprehensive description of ecological, social, and economic components within an assessment area (USDA Forest Service 1999b). A primary purpose of this assessment is to provide context to National Forest System land management planning efforts involving grasslands, both at the Forest Plan level for Plan amendments and revisions, and at the project level to place site-specific activities within the larger framework. The assessment is not a decision document because it surfaces issues and risks to grassland ecosystems that provide the foundation for future changes to Forest Plans or project activities but does not make any site-specific decisions or recommendations. The report also provides a scientific basis for conducting ecosystem restoration projects, provides a starting point for public discussion on desired conditions for the future, and contributes to the overall understanding of the physical, biological, and human dimensions of grassland ecosystems in the Southwest.

The report is divided into two volumes. The first volume (herein) focuses on the ecology, types, conditions, and management practices of Southwestern grasslands. The second volume emphasizes wildlife and fish species and their habitat requirements in Southwestern grasslands.

To prepare this document, we assembled a team of authors from the Southwestern Region and the Rocky Mountain Research Station whose expertise focused on or included grassland ecosystems. An outline of chapter titles and chapter contents was prepared using a group consensus process. Authors volunteered to write specific chapters that were then reviewed by the team. Following team review, each individual chapter was sent to a minimum of two peer reviewers for critique, and in addition, the entire revised volume was sent to two reviewers. Also, the team interviewed Forest Service employees (see appendix).

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Southwestern Grassland Ecosystems

In the Southwestern Region, the Forest Service has adopted the Soil Conservation Society (SCS) of America's definition of grasslands, that is, "lands on which the existing plant cover is dominated by grasses" (SCS 1982). Risser (1995) defined grasslands as "biological communities that contain few trees or shrubs, are characterized by mixed herbaceous vegetation, and are usually dominated by grasses." Supported by the National Science Foundation, the U.S. International Biological Program (IBP) characterized *natural grasslands* as climatically determined by soil water availability and precipitation volume and seasonality, *successional grasslands* where forest vegetation has been removed, and *agricultural grasslands* where a few native or introduced species are maintained. This report addresses natural grasslands.

This assessment includes the following Southwestern grassland types:

- Montane grassland
- Colorado Plateau
- Desert grassland
- Great Basin grassland
- Plains grassland

Subalpine grasslands are discussed within the montane grassland category. Alpine grasslands are not discussed as a separate category because they have a limited extent in the Southwest. Where they are mentioned, they are discussed in conjunction with montane grasslands, although they occur on a different mountain gradient. Riparian and/or wetland inclusions occur in all grassland types and are discussed separately where appropriate.

Ecologists and geographers identify broad-scale Southwestern grasslands (that is, biome level) as **temperate grassland**. These biome classifications are according to macroclimate conditions defined by Köppen, Threwartha and others (Coupland 1992, French 1979). Further subdivisions according to physiographic province include Great Basin grassland and Plains grassland. In general, the term **plains** refers to grasslands in areas of flat topography (Coupland 1992). Plains grassland is then subdivided into short, tall, and mixed grasslands. At a finer scale, grasslands are designated by vegetation community/plant association as classified by Küchler (1964), Clements (1920), and others.

Temperate grasslands are areas at mid-latitude that are dominated by perennial grasses and forbs. Climate is moderately dry (semiarid) with discrete wet/dry seasons and temperature and precipitation extremes (Sims 1988). Soils are predominantly characterized as Aridisol or Mollisol with large amounts of humus (Aber and Melillo 1991, Sims 1988, Whittaker 1975). Temperate grasslands include tall, mid and short grasses (Odum 1971, Whittaker 1975). Tall grasses are about 150 to 245 cm (5 to 8 feet) high, mid grasses approximately 60 to 120 cm (2 to 4 feet) high, and short grasses 15 to 45 cm (0.5 to 1.5 feet) in height. Short grasses include buffalograss *(Buchloe dactyloides*), blue grama *(Bouteloua gracilis*), and other gramas. Mid grasses include little bluestem *(Schizachyrium scoparium*), needlegrass *(Stipa* spp.), western wheatgrass *(Pascopyrum smithii*), and Indian rice grass (*Achnatherum hymenoides*) (Odum 1971).

Temperate grassland biomes include **prairie** and **steppe** (French 1979, McKnight 1993, Odum 1971, Whittaker 1975). Prairie, including the true tall-grass prairie, mixed-grass prairie, and short-grass prairie, is dominated by grasses and forbs, has a scarcity of shrubs, and has no trees. Mixed-grass prairie is an ecotone between tall and short-grass prairie. The term steppe refers to a temperate biome that is dominated by short grasses and bunchgrasses (McKnight 1993) and is dryer than prairie. Steppes receive approximately 25 to 50 cm (10 to 20 inches) of rain per year and experience hot summers and cold winters; these climatic conditions support plants such as blue grama, buffalograss, big bluestem (*Andropogon gerardi*), cacti, and sagebrush.

Grasslands can be subdivided using the U.S. National Vegetation Classification system (Federal Geographic Data Committee 1997) and other methods according to class, subclass, group, formation, regional biome type, alliance, plant associations, or habitat types. Grassland categories for the Southwestern United States include the Plains grassland, Great Basin grassland, and the Colorado Plateau as discussed above, as well as montane grassland and desert grassland. **Montane grassland** can be found in small patches within the mixed conifer and ponderosa pine forests. Montane or high-mountain grasslands consist of meadows below timberline (French 1979), while **alpine grassland** is located above timberline (Whittaker 1975). **Desert**

View of Animas Valley, looking east toward Animas Mountains, New Mexico. Mesquite in perennial grassland. (Photo by Ronald Bemis)

grassland occurs in arid and semi-arid climates. It varies in composition from mixed herbaceous species with few shrubs to primarily a combination of shrub species (French 1979) such as mesquite (*Prosopis* spp.) and creosote bush (*Larrea tridentata*). Subregions include the Chihuahuan Desert grasslands of southern New Mexico, characterized by black grama (*Bouteloua eriopoda*), and the Sonoran Desert grasslands of southeastern Arizona.

Biome classification systems use the term *savanna* to describe tropical grasslands that are primarily located in Africa and Australia, South America and southern Asia/India (Whittaker 1975, McKnight 1993). Similarly, Bailey's ecosystem classification uses the term savanna to describe a Division within the Humid Tropical Domain. However, some people use the term temperate savanna to describe areas in the Southwestern United States. McPherson (1997) defines North American savannas as "ecosystems with a continuous grass layer and scattered trees or shrubs." The woody plant overstory has approximately 30 percent cover or less with a grass understory. He further defines and maps areas of the Southwest as Piñon-Juniper Savanna, Southwestern Oak Savanna, Ponderosa Pine Savanna, and Mesquite Savanna. Although this terminology is sometimes used, a savanna biome does not occur in the Southwestern United States according to broad-scale vegetation classification systems based on climate. The Forest Service also uses a classification system based on a geographic approach, also referred to as regionalization, which is a process of classification and mapping to identify homogeneous map units at various scales. The National Hierarchical Framework of Ecological Units adopts Bailey's classification by ecoregions (Bailey 1995); the hierarchy consists of domain, division, province, section, subsection, landtype, landtype association, and landtype phase. At the regional scale, provinces and sections are the most useful units for assessments.

Relationship of Assessment to Ecosystem Management

Ecosystem management is an evolving philosophy that has been adopted by many government agencies including the Forest Service. The Forest Service has defined ecosystem management as "a concept of natural resource management wherein national forest activities are considered within the context of economic, ecological, and social interactions within a defined area or region over both short and long term"

(Thomas and Huke 1996). National Forest activities in this context are all activities occurring on National Forest System lands, including grassland ecosystems. Ecosystem management is sometimes referred to as ecology-based multiple-use management in that there is a shift from focusing exclusively on sustaining production of goods and services to sustaining the viability of ecological, social, and economic systems. While other agencies and organizations have developed their own definitions of ecosystem management reflecting their differing missions, they typically have a goal of ecosystem sustainability or maintaining ecological integrity (Grumbine 1994, Kaufmann and others 1994) while recognizing that people are part of the ecosystem and that human needs should be reflected in ecosystem sustainability (Keystone Center 1996, USDA Forest Service 1994).

A significant difference among various ecosystem management philosophies may be the degree in which people are included within the sustainability concept. Former Forest Service Chief Dombeck stated, "We will still track traditional outputs of goods and services but they will be accomplished within the ecological sideboards imposed by land health" (Dombeck 1999). The Keystone Center's National Policy Dialogue Group on Ecosystem Management placed the goals of sustaining vibrant, livable, and economically diverse human communities and the involvement of stakeholders on a par with the goals of maintaining ecosystem integrity and sustaining biodiversity and ecosystem processes at a regional scale (Keystone Center 1996). The Southwestern Region's philosophy embraces all these goals as well. In developing the human dimension principles and strategies for the Southwestern Region, the Human Dimensions Team recognized that human needs and wants must be balanced with ecosystem capabilities. However, on a practical level, they also recognized that the goal of meeting human needs frequently conflicts with the goal of sustaining natural ecosystems (USDA Forest Service 1994). When this is the case, ecosystem management may be considered an optimal integration of ecological sustainability and human dimensions (including both economic considerations and societal needs and desires) (Jensen and others 1996).

Assessments are a tool in ecosystem management to develop a holistic understanding of ecological sustainability as well as the human dimension of ecosystems. As an introductory chapter, chapter 1 not only describes the purpose and need for assessments, it also gives a brief overview of Southwestern grassland types and defines two terms in frequent use in later chapters: ecosystem sustainability and adaptive management. Chapter 1 also explores and emphasizes the role of, and need for, monitoring of grassland conditions and trends, a topic not covered in detail in later chapters. In this report, chapter 2 describes the extent and types of grasslands in the Southwest. The general ecology of Southwestern grasslands is evaluated in chapter 3. A discussion of the biological diversity, functional processes, and consequences of grassland fragmentation is provided in chapter 4. Cultural dimensions of grassland management, both from a historic and contemporary perspective, are covered in chapter 5. Historic and current conditions of southwestern grasslands in relation to land management are covered in chapter 6. Chapter 7 discusses the concept of grassland sustainability and why understanding sustainability is critical for managers and stakeholders to collaboratively develop desired conditions for grassland areas. Management decisions will need to be site-specific based on the unique characteristics of the area. Because there is no one-size fits-all management strategy for an area, chapter 8 discusses a wide range of tools available for use by grassland managers.This chapter has an eye toward highlighting some of the more innovative work being done in grasslands rather than attempting to document all possible tools. Chapter 8 also covers research needs, since adaptive management and the utilization of the best scientific knowledge are important components of ecosystem management.

Ecosystem Sustainability

Ecosystem sustainability is the ability of an ecosystem to maintain ecological processes and functions, biological diversity, and productivity over time (Kaufmann and others 1994). It was the subject of the 1992 Earth Summit/United Nations Conference on Environment and Development in Rio de Janeiro and a 1987 World Commission on Environment and Development report called *Our Common Future* (also known as the Bruntland Report; Bruntland 1987). **Sustainable ecosystems** are able to maintain their ecological integrity.

Ecological integrity is achieved when ecosystem structure, function, processes, and services are preserved over space and time (Grumbine 1994). *Ecosystem structure* is the spatial arrangement of the living and nonliving elements of an ecosystem, for example, abiotic elements (temperature, light, wind, relative humidity, rainfall) and community structure (species richness and the distribution of *heterotroph*s, *autotrophs*, and consumers). *Ecosystem function* refers to the processes whereby the living and nonliving elements of ecosystems change and interact, such as biogeochemical processes and succession. *Ecological processes* are the actions or events that link organisms and their environment. Ecosystem processes include disturbance, succession, evolution, adaptation, natural extinction rates, colonization, dispersion, fluxes of materials, and decomposition (Kaufman and others 1994).

Sustainable ecosystems continue to provide essential services. The Ecological Society of America (1997) has identified the following services provided by ecosystems:

- moderating weather extremes and their impacts
- seed dispersal
- drought and flood control
- protection from ultraviolet rays of the sun
- nutrient cycling
- protection from channel and coastal erosion
- waste decomposition and breakdown
- agricultural pest control
- maintaining biodiversity
- generating and preserving soils; renewing soil fertility
- helping to stabilize climate
- cleaning the air and water
- regulating organisms that carry disease
- pollination

Sustainable grasslands can be described as productive grasslands with erosion rates that do not exceed soil tolerance for sediment and nutrient loss; having natural fire frequencies; biologically diverse; not overgrazed; and containing important social and aesthetic values (Mac and others 1998).

Adaptive Management

This assessment utilizes existing information, and the assessment team did not initiate any new data collection activities. However, the assessment team synthesized information on grasslands that had been available only in a piece-meal fashion up to this point, and the presentation of that information at the regional scale provides new insights that can be used in adaptive management.

Adaptive management is a proactive approach to implement ecosystem management. The theory and practice of adaptive management has evolved over the past two decades through the works of Holling (1978) and Lee (1993). One of the fundamental tenets of adaptive management is that ecosystems and people are unpredictable as they evolve together. Ecological conditions change as do societal values and economic developments. In addition, our understanding of ecosystem behavior is imperfect, and managers will never be able to completely predict responses to management activities. The purpose of adaptive management is to rapidly increase our level of knowledge of ecosystem dynamics and the effects of management. In the adaptive management approach, management plans and activities are continually altered in response to changing societal values and goals, emerging issues, and new scientific understanding (Haynes and others 1996).

The dynamic nature of grasslands requires an adaptive management approach. Change has been and continues to be rapid in grassland ecosystems. The intensity of grazing has fluctuated through time, new uses such as off-road vehicle recreation and energy development have emerged, and development pressures continue to escalate. Some wildlife species have declined or disappeared altogether, while other species have increased in population. These changes, along with others such as climatic variation, introduced plants, and changing fire regimes, have all shaped grassland ecosystems, sometimes to the point where the land cannot be returned to a previous state. Adaptive management experiments would be useful in understanding where these thresholds occur and management's ability to reverse changes as the system approaches these thresholds.

Adaptive management encompasses both deliberate experimentation to gain new knowledge (active adaptive management) as well as the ongoing process of monitoring and inventorying to assess the effects of management actions (passive adaptive management). Passive adaptive management may seem a misnomer because it requires an active program for the monitoring and evaluation of project activities. Active adaptive management is a departure from traditional management in that it views management actions as experiments from which to learn. Conducting adaptive management experiments involves being explicit about expected outcomes, designing methods to measure responses, collecting and analyzing information to compare expectations to actual outcomes, learning from the comparisons, and changing actions and plans accordingly. In both forms of adaptive management, monitoring plays a crucial role in surfacing any needed changes to management plans and activities, and monitoring can lead to the need for new assessments.

Ecological Monitoring of Grasslands

The basis of ecosystem management is management driven by explicit goals and made adaptable by monitoring and research based on ecological interactions and processes necessary to sustain ecosystems (Gordon et al. 1992). Monitoring, as an overall process, is a measurement process that establishes a baseline with periodic measurements for the purposes of change detection and adaptive management. Monitoring is defined as the systematic observation of parameters related to a specific problem, designed to provide information on the characteristics of the problems and their changes with time (Spellerberg 1991). In a natural resources context, ecological monitoring assesses the status and trends of ecological, social, or economic outcomes (Powell, unpublished paper 2000). The information being collected depends on the purpose of the monitoring—that is, which questions are to be addressed—and the scale at which a question needs to be answered. The results of monitoring are expected to generate an action of some kind, even if the action is to maintain current management (Johnson and others 1999).

Monitoring is a step-wise process that involves:

1. Framing a question(s) and developing a study plan to address the question(s) using a standard protocol.
2. Collecting data according to the monitoring plan.
3. Storing the data for retrieval.
4. Evaluating the results.

Monitoring should include goals, thresholds for change, and remedial actions that occur when thresholds are met or exceeded. An ecological systems approach to monitoring ensures a strong foundation in ecological theory, adequate consideration and understanding of cause-and-effect relationships, and a systematic approach to selecting and evaluating parameters that are monitored (USDA Forest Service, unpublished paper 1999a). Furthermore, the questions should focus on key ecological processes and interactions, rather than on individual parts of the system. Powell (unpublished papers 2000, 2001) listed the following characteristics for successful monitoring:

- Purposeful and conducted to answer specific questions. Conducted at the appropriate spatial and temporal scale to answer the question.
- Conducted in collaboration with others (for example, agencies, interested publics, researchers, and nongovernmental organizations) to share the workload (including obtaining data from other sources), gain expertise, and build credibility and trust.
- Conducted using the best available science and established protocols to collect and evaluate the data.
- Conducted using modern information management techniques and tools.
- Stringent selection criteria applied so that a monitoring activity is only conducted if it is feasible, realistic, and affordable.
- Evaluation emphasized as much as the collection of the data.

Protocols for monitoring should include standard sampling and analytical methods that determine the precision and accuracy of measurements. Proper training and supervision of field and laboratory staff is necessary to ensure adherence to the protocol and the success of the monitoring program.

Based on these criteria, one of the activities in the interagency Southwestern Strategy effort is to develop a unified set of tools and techniques, approved by Federal and State agencies, for rangeland monitoring training in New Mexico (de la Torre, personal communication).

Monitoring addresses both management activities on the land and conditions of the ecosystem being monitored. The monitoring of management activities can further be subdivided into three arenas: implementation, effectiveness, and validation monitoring. Implementation monitoring addresses, evaluates, and determines whether plans, projects, and activities were implemented as designed and in compliance with Forest Plan objectives, standards, and guidelines. Effectiveness monitoring addresses, evaluates, and determines whether plans, projects, and activities met Forest Plan management direction, objectives, standards, and guidelines. Validation monitoring determines whether the assumptions and relationships between the activity and the expected results were valid. In 2000, the Southwestern Region undertook an analysis of the Forest Plan Monitoring Reports for fiscal years 1998 and 1999. Most of the forest level monitoring in the Region was implementation monitoring (65 percent). About 32 percent of the reported monitoring was evaluation monitoring, and the remaining 3 percent was validation monitoring. The information was not detailed enough to know whether grassland monitoring followed this general trend (USDA Forest Service, unpublished data, 2000). To monitor ecological conditions, quantitative or qualitative parameters are monitored; these parameters are typically referred to as indicators, attributes, variables, or monitoring elements. In connection with condition and trend monitoring, it is useful to establish baseline reference conditions to serve as a context for determining desired conditions and interpreting existing conditions.

In the Forest Service, monitoring and evaluation of management actions is one step in the overall process for proposed action identification, design, and monitoring. In outlining the process for Forest Plan implementation, the Southwestern Region has developed a desk guide, *Integrated Resource Management: The Road to Ecosystem Management*. Monitoring forms the base of the Southwestern Region's Integrated Resource Management Process triangle (USDA Forest Service 1993). In an effort to capture monitoring activities on National Forest System Lands, since 1997 an annual "Forest Monitoring and Evaluation Report and State of the Region Evaluation" has been included in the Forest Service's performance reporting system (MAR—Management Attainment Report). Despite being integrated into Forest Service processes, monitoring within the agency tends to be uncoordinated and inconsistent (Powell 2000). There is a perception in the Southwestern Region that monitoring activities chronically suffer from a lack of funding and personnel (see appendix, question 1). These problems may be even greater in grassland areas than in forested ecosystems, since progress has been slower at developing an agreed-upon set of ecological indicators to monitor.

Ecological Indicators of Sustainability

Numerous *ecological indicators* have been proposed or are being used to address grassland sustainability. Indicators can be used to define any expression of the environment that estimates the condition of ecological resources, magnitude of stress, exposure of a biological component to stress, or the amount of change in a condition (Breckenridge and others 1994). Indicators, by their very nature, will vary depending on the scale to which they are applied. At the project-level and landscape-level scales, a commonly agreed-upon set of indicators for grasslands is lacking. Selecting indicators at these scales is an enormous challenge because of the complexity of ecosystems. There are potentially hundreds if not thousands of ecosystem characteristics that could be measured in response to environmental or management-induced changes, and it is difficult to separate responses occurring at the fine scale from potential influences occurring at the broader scale. One approach used at the Sevilleta Long Term Ecological Research (LTER) site is to identify a matrix of indicators at many levels and scales to sample the diversity of responses across spatial and temporal scales. Identifying the key indicators and the scales they reflect is an ongoing process (Gosz and others 1992).

On a national scale, there have been several major efforts to identify ecological indicators, though none specifically address grasslands. The National Academy of Science's National Research Council recommended the following indicators to portray ecological condition: land cover and land use; total species diversity; native species diversity; nutrient runoff estimates, especially for nitrogen and phosphorus; soil organic matter; productivity (including carbon storage, net primary production, and production capacity); lake trophic status; stream oxygen concentration; and nutrient-use efficiency and nutrient balance (Committee to Evaluate Indicators for Monitoring Aquatic and Terrestrial Environments and others 2000). Land cover—including cropland, forest, nature reserves, rivers, wetlands, and riparian zones—is usually detected and monitored using remote sensing, while land use is typically classified by measurements from the ground. Soil organic matter is important as a nutrient and energy source for soil biota. It improves soil structure by strengthening soil aggregates, increases water retention and available water capacity, reduces the sealing of soil surfaces thereby promoting infiltration and reducing erosion, increases cation exchange capacity, chelates metals, and influences the fate of pesticides. Nutrient-use efficiency is a good indicator for crops, industries, counties, or watersheds.

A multiscale effort to develop indicators has sprung from the Montreal Process, which initially identified international criteria and indicators for sustainability of forest ecosystems. The United States has endorsed the seven criteria and 67 indicators, and the Forest Service is in the process of applying them within the agency. The criteria that have been identified are:

- Conservation of biological diversity.
- Maintenance of productive capability of forest ecosystems.
- Maintenance of forest ecosystem health and vitality.
- Conservation and maintenance of soil and water resources.
- Maintenance of forest contribution to global carbon cycles.
- Maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies.
- Legal, institutional, and economic framework for forest conservation and sustainable management.

The Forest Service has been conducting a series of pilot projects in an effort to identify indicators that would be appropriate at a Forest Management Unit scale, including those appropriate for rangelands (USDA Forest Service Inventory and Monitoring Institute 2001). Concurrently, a national team reviewed the Montreal indicators for their applicability to rangelands at a national scale. While the results should be considered preliminary, for the first five criteria, the review concluded that almost all the indicators were as relevant for rangeland ecosystems as for forested ecosystems, although the lack of a generally accepted classification system, consistent definitions, data, and/or mechanisms to measure most of the indicators makes quantification at a national scale problematic. Also, it was noted that some of the indicators may be more effective if they were refined to capture aspects of the criteria goals that are more relevant to rangeland dynamics (Flather and Sieg 2000, Joyce 2000, Joyce and others 2000, McArthur and others 2000, Neary and others 2000).

Why Assess Grassland Ecosystems?

The selection of grassland ecosystems as the second major Southwestern ecosystem type to be assessed was based on several factors. A huge amount of grasslands acreage has been lost to development, agriculture, and other uses across the country, yet it is likely that the average person is unaware that grasslands are endangered. Over half of the critically endangered ecosystems in the United States (those with over 98 percent of area either lost or degraded) are grasslands (Noss and others 1995). In the Southwest as elsewhere, grasslands are often lumped together with other grazed ecosystems and analyzed as rangelands or grazing lands. Between 1982 and 1992, Arizona had a net loss of 382,000 acres of non-Federal grazing land; New Mexico, 351,000 acres. During this period in New Mexico, 23 percent of non-Federal rangeland that was converted to other uses, was converted to developed land, and 16 percent was developed for agricultural uses (Goddard and others 1999). The amount of National Forest lands, including rangelands, has not increased significantly in the past 20 years (Mitchell 2000) and the amount of Federal grasslands may even be lower in the Southwest due to the loss of montane meadows and the expansion of woodlands. The loss of grazing land in the non-Federal sector increases the importance of the remaining Federal lands.

The effects of grazing on grasslands have always been a contentious issue, particularly in the Southwest, since domestic livestock was introduced to the Southwest much earlier than other Western States. Sheep were brought to the Rio Grande pueblos in 1598, and other livestock soon followed. By 1890, it is estimated that over 1.5 million cattle were grazing in the Southwest, and range deterioration was being reported (Baker and others 1988). In a document submitted to the Senate in 1936, the Secretary of Agriculture indicated that the Southwest contained the most severe range depletion in the Western United States. Much of the Southwest was in the "extreme" range depletion class, where 76 to 100 percent of the area was considered depleted (Secretary of Agriculture 1936). Determining the condition of rangeland today is complicated by not having a standard process in place across all ownerships. As assessment of non-Federal lands was made in 1989 from data collected in 1982 during the National Resource Inventory (USDA Soil Conservation Service 1990) and supplemented in 1992. Conditions were determined based on how closely the species composition of a range site met those projected for the climax plant community for that site. For both 1982 and 1992, only 2 percent of non-Federal rangelands were considered to be in excellent condition in Arizona and New Mexico. During this period, percentages of good condition rangeland increased from 16 to 27 percent in Arizona, and from 30 to 36 percent in New Mexico.

The USDI Bureau of Land Management (BLM) is a major Federal land manager of rangelands in the Southwest. The agency's assessments of range condition are by ecological status categories: potential natural community (PNC), late seral, mid seral, and early seral. In 1996, while only 5,607,000 acres had been classified out of the 11,643,000 acres of BLM managed Arizona rangelands, the results were 9 percent PNC, 40 percent late seral, 40 percent mid seral, and 12 percent early seral. In New Mexico, the percentage of acres classified is much greater. Out of 12,597,000 acres of BLM rangelands in New Mexico, 9,752,000 have been classified. These fall into the following categories: PNC 1 percent, late seral 36 percent, mid seral 48 percent, and early seral 15 percent (USDI Bureau of Land Management 1997). Forest Service assessments have also been in upland rangelands rather than specifically grasslands. These are not assessed by range condition categories, but by whether the vegetation meets or is moving toward Forest Plan Management Objectives (FPMO). Of all the Regions assessed during the 1995 to 1997 period, the Southwestern Region had the largest percentage of rangelands not meeting FPMO objectives and not moving toward FPMO, roughly 25 percent (Mitchell 2000).

Another primary factor for selecting grasslands for an assessment effort is the complexity of grassland ecology combined with the variety of anthropogenic influences on grassland ecosystems. As was mentioned previously in the adaptive management section, changes have occurred rapidly in grassland ecosystems, and some areas may have passed a threshold where they cannot be returned to an earlier condition.

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Chapter 2: Grassland Assessment Categories and Extent

Introduction

This chapter establishes a general framework for describing the various kinds of grasslands outlined in subsequent chapters. This framework outlines the major categories or classes of grasslands that occur as part of Southwestern terrestrial ecosystems within National Forest System lands and provides an ecological and environmental context in regards to how they differ in their floristic, geographic, spatial, and climatic settings. More detailed information about these grassland systems is also presented in chapter 6.

Grasslands of the Southwest vary according to vegetation, climate, soils, and topography and disturbance regimes. They are distinctly different from other vegetation assemblages in that the dominant and codominant plants are graminoid species. For example, other forbs and shrub plant species occur within the grasslands but are subordinate to grass in the total cover and composition.

The major grassland categories used in this assessment—that is, those categories that represent the major grasslands formations in the Southwestern Region on National Forest System lands (Carleton and others 1991)—are Desert, Plains, Great Basin, Montane, and Colorado Plateau grasslands. These generalized groupings reflect the geographic and ecological differences that are determined by unique floristic, edaphic, physiographic, and climatic characteristics. Although not taxonomic with respect to any vegetation hierarchy, these categories are intended to aid the reader in understanding the uniqueness, distribution, and extent of these systems. Other classification systems of Southwestern grasslands exist (Barbour and Billings 2000, Brown 1994, Dick-Peddie 1993, Küchler 1964) and emphasize biogeographic, ecological, and biophysical features that are consistent with the scale and level of generalization being used here. The general distribution of grasslands for this assessment is located on the National Forest System lands in Arizona, New Mexico, Texas, and Oklahoma (fig. 2-1).

Grassland Categories

Descriptions of each grassland assessment category follow.

The **Desert Grassland** encompasses annual and perennial graminoid and adjacent shrub communities at low elevations adjacent to the Chihuahuan, Mohave, and Sonoran deserts. These grasslands occur between the Great Basin grasslands, chaparral, and woodland ecosystems and have been commonly referred to as semidesert grasslands by Brown (1994). The distribution of these grasslands are mainly within the Basin and Range, Sonoran-Mohave Desert, Tonto Transition ecoregion sections, and limited areas within the White Mountain-San Francisco Peaks, Northern Rio Grande Intermontane, and Sacramento-Monzano Mountain ecoregion sections (McNab and Avers 1994). Desert grasslands intermingle with desert scrub communities (Dick-Peddie 1993) and have evolved through natural and anthropogenic successional disturbance processes. Grass species that are diagnostic to this category include

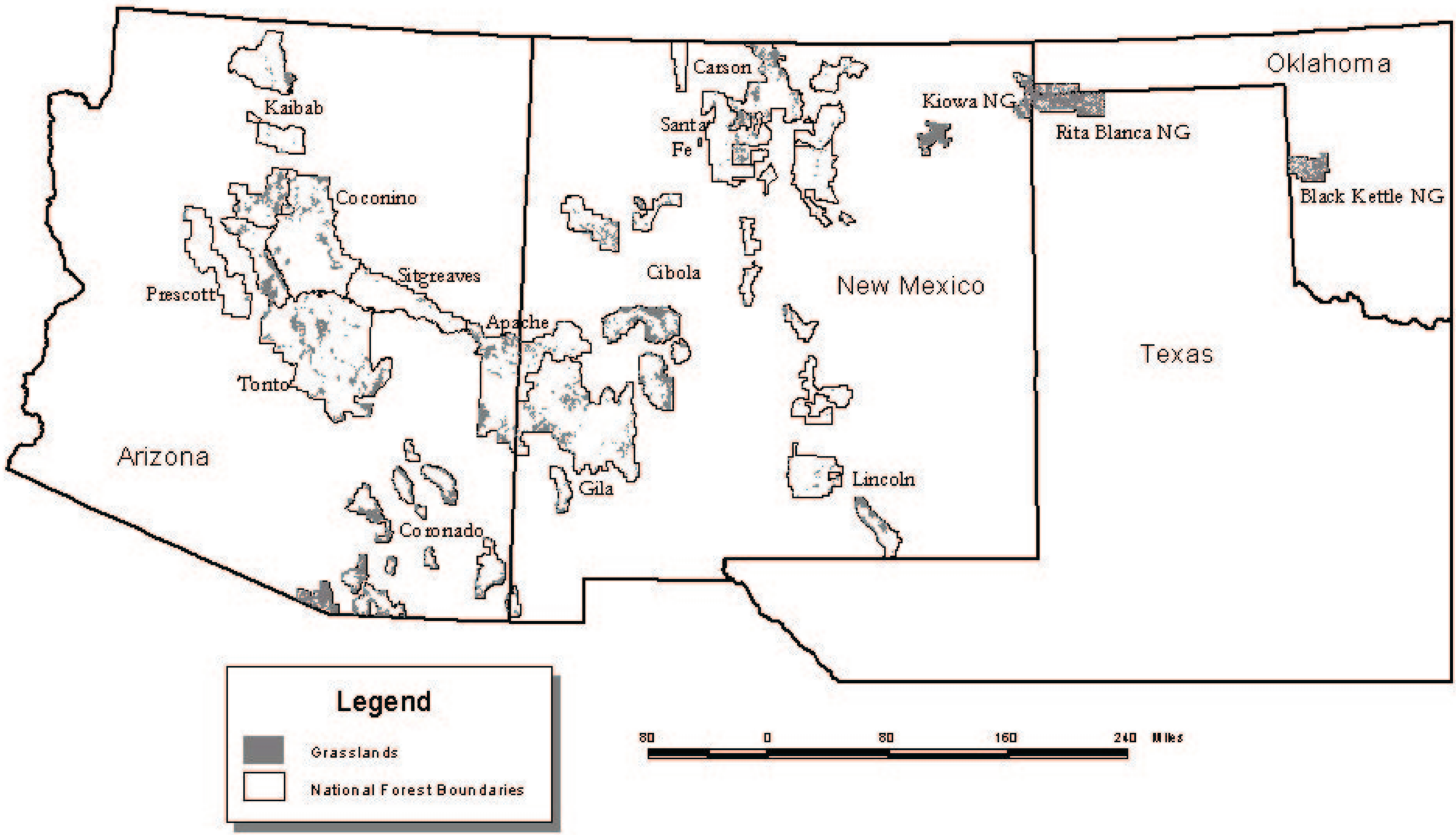

Figure 2-1. Grasslands of National Forest System lands in Arizona, New Mexico, Texas, and Oklahoma.

black grama (*Bouteloua eriopoda*), tobosa (*Pleuraphis mutica*), and curly mesquite (*Hilaria belangeri*). Other key graminoid species that occur within this formation include bushmuhly (*Muhlenbergia porteri*) and burrograss (*Scleropogon brevifolius*). Major shrubs that occur in association with these species include creosote bush (*Larrea tridentata*), velvet mesquite (*Prosopis velutina*) in Arizona, western honey mesquite (*Prosopis glandulosa var. torreyana*) in southern New Mexico, tarbush (*Flourensia cernua*), turpentine bush (*Ericameria laricifolia*), desert ceanothus (*Ceanothus greggii*), and soaptree yucca (*Yucca elata*).

The **Great Basin Grassland** occurs within the White Mountain-San Francisco Peaks, Saramento-Manzano Mountains, Central Rio Grande Intermontane, and higher elevations of Basin and Range and Sonoran Desert ecoregion sections (McNab and Avers 1994) of the Southwestern region. These grasslands are higher in elevation and climatically cooler and moister than desert grasslands and are adjacent to and intermingle with juniper (*Juniperus* spp.) savanna ecosystems. The Great Basin grasslands are similar to Brown's (1994) Plains and Great Basin grasslands and Dick-Peddie's (1993) Plains–Mesa grasslands except the geographic range of this category for this assessment is restricted to the Basin and Range Physiographic province (Fenneman 1928). Diagnostic plant species include blue grama (*Bouteloua gracilis*), galleta (*Pleuraphis jamesii*), Indian ricegrass (*Achnatherum hymenoides*), and sideoats grama (*Bouteloua curtipendula*). Some dropseeds, (*Sporobolus* spp.) and wolftail (*Lycurus phleoides*) are codominant and add to the diversity of this category. The Great Basin grasslands tend to be drier than the Shortgrass Steppe grasslands and have a blend of warm and cool season graminoid and forb species. Shrubs that are present in association with grassland vegetation of this category include fourwing saltbush (*Atriplex canescens*), sacahuista (*Nolina microcarpa*), small soapweed yucca (*Yucca glauca*), skunkbush sumac (*Rhus trilobata*), and catclaw mimosa (*Mimosa biuncifera*). As this grassland integrades with savanna ecosystems, minor amounts of trees such as emory oak (*Quercus emoryi*), alligator juniper (*Juniperus deppeana*), and Utah juniper (*Juniperus osteosperma*) dominated woodlands are evident.

The **Colorado Plateau Grassland** is located in northern Arizona above the Mogollon Rim and northern New Mexico in association with the Colorado Plateau and adjacent to small areas of the Rocky Mountain physiographic provinces (Fenneman 1928). It occurs within the Grand Canyonlands, Painted Desert, Tonto Transition, White Mountains-San Francisco Peaks, Navajo Canyonlands, Southcentral Highlands, South-Central Highlands, Southern Parks and Ranges, and Upper Rio Grande Basin ecoregion sections (McNab and Avers 1994). Colorado Plateau Grasslands—a new category of Southwestern grassland—primarily splits the expansive Great Basin Grassland category based upon recent ecological mapping (Laing and others 1986, Miller and others 1995, Robertson and others 2000) and what Kuchler (1970) referred to as the Galleta-Threeawn Shrub Steppe. These grasslands occur on nearly level, wind-desiccated geomorphic surfaces of sedimentary and igneous origin. Grass species that characterize this category include western wheatgrass (*Pascopyrum smithii*), needle and thread (*Hesperostipa comata*), blue grama (*Bouteloua gracilis*), galleta (*Pleuraphis jamesii*), and New Mexico feathergrass (*Hesperostipa neomexicana*), and various species of three-awn (*Aristida* spp). Common shrubs include big sagebrush (*Artemisia tridentata*), black sagebrush (*Artemisia nova*), fourwing saltbush (*Atriplex canescens*), and Mormon tea (*Ephedra trifurca*). Oneseed juniper (*Juniperus monosperma*) and Utah juniper (*Juniperus osteosperma*) woodlands and savannas are adjacent to Colorado Plateau grasslands.

The **Plains Grasslands** consist of the shortgrass, midgrass, and tallgrass prairies of the National Grasslands. These grasslands extend throughout the Great Plains physiographic province (Fenneman 1928) and occur within the Southern High Plains, Pecos Valley, Redbed Plains, and Texas High Plains ecoregion sections (McNab and Avers 1994). Climate ranges from subhumid to semiarid as these grasslands extend from east to west. The characteristic plant species that are abundant throughout the shortgrass prairie include blue grama (*Bouteloua gracilis*) and buffalo grass (*Buchloe dactyloides*). The midgrass prairie ecosystem is codominated by little bluestem (*Schizachyrium scoparium*), blue grama (*Bouteloua gracilis*), and plains bristlegrass (*Setaria vulpiseta*). The tallgrass prairie is dominated by big bluestem (*Andropogon girardii*). These different prairie ecosystems are aggregated and reduced to one category for this assessment and reflects a wide range of ecological properties and processes.

The **Montane Grasslands** category includes the montane, subalpine and alpine meadows, valleys, and high elevation grasslands that occur throughout the Grand Canyonlands, Painted Desert, Tonto Transition, White Mountain-San Francisco Peaks, Basin and Range, Central Rio Grande Intermontane, South-Central Highlands, Sacramento-Manzano Mountain, Southern Parks and Ranges, and Upper Rio Grande Basin ecoregion sections (McNab and Avers 1994). These grasslands are similar to Subalpine-Montane Grasslands described by Dick-Peddie (1993) and the Alpine and Subalpine and Montane Meadow grasslands of Brown (1994). Carleton and others (1991) classified montane, subalpine and alpine terrestrial ecosystems as edaphic-fire and topo-edaphic-zootic disclimaxes with temperate continental climates. Diagnostic plant species that characterize these ecosystems include Arizona fescue (*Festuca arizonica*), mountain muhly (*Muhlenbergia montanus*), Kentucky bluegrass (*Poa *pratensis*), timber oatgrass (*Danthonia intermedia*), Thurber fescue (*Festuca thurberii*), tufted hairgrass (*Deschampsia caespitosa*), alpine avens (*Geum rossii*), and Bellardi bog sedge (*Kobresia myosuroides*).

Mapping

The delineation of grasslands for this assessment involved integrating and cross-walking the categories of vegetation types within existing land cover classes and ecological units from five main sources: (1) General Terrestrial Ecosystem Survey (GTES) (Carleton and others 1991), (2) New Mexico Gap Analysis Project (Thompson and others 1996), (3) Texas Gap Analysis Project (Parker 2001), (4) Oklahoma Gap Analysis Project (Fisher 2001), and (5) Arizona GAP Analysis Project (Thomas 2001). These five primary sources were used for assessing distribution and extent of the five grasslands assessment categories.

The University of New Mexico, Earth Data Analysis Center, Albuquerque, performed data processing and geographic information system analysis.

The grassland assessment categories were nested within the Ecoregion and Subregions map of ecological units (Bailey and others 1994, McNab and Avers 1994). The Ecoregion and Subregions map and descriptions contain integrated biophysical information about broadscale ecological characteristics including climate, soils, geomorphology, potential natural vegetation, surface water characteristics, disturbance regimes, and land use. This integrated approach to regionalization of ecosystems allows managers, planners, and scientists to study management issues on a multi-Forest and Statewide basis. More mapping particulars are given in figures 2-2 and 2-3.

GAP land cover classes and GTES vegetation taxa (series) were combined through a process of correlation (table 2-1). This process involved aggregating categories with similar physiognomic, floristic, and geographic ranges into the five assessment classes. Differences occur between nomenclature and image resolution of land cover classes for each State GAP product. Furthermore, some States had broader land cover classes that include plant communities of adjacent vegetation formations. Consequently, the spatial resolution as predicted by the map may depict grasslands to be of more variable extent than what would be evident at finer scales with higher resolution. This is particularly true for the Desert and Great Basin grasslands where these communities integrade and commingle with adjacent shrubland steppe communities. Conversely, some areas of known grasslands on National Forest lands in Arizona and New Mexico failed to be recognized and delineated because of map scale limitations based upon a 200-ha threshold that excluded these smaller isolated areas that were eliminated to maintain cartographic integrity and utility of the map product. These areas typically occurred at the edges of the National Forest System boundary.

Practical Application

The categorization of grasslands into generalized vegetation types assists natural resource managers in understanding the geographic variability and spatial distribution across National Forest Lands in the Southwestern Region. This understanding will potentially lead to progressive management actions to maintain and restore these grasslands to ensure their ecological sustainability.

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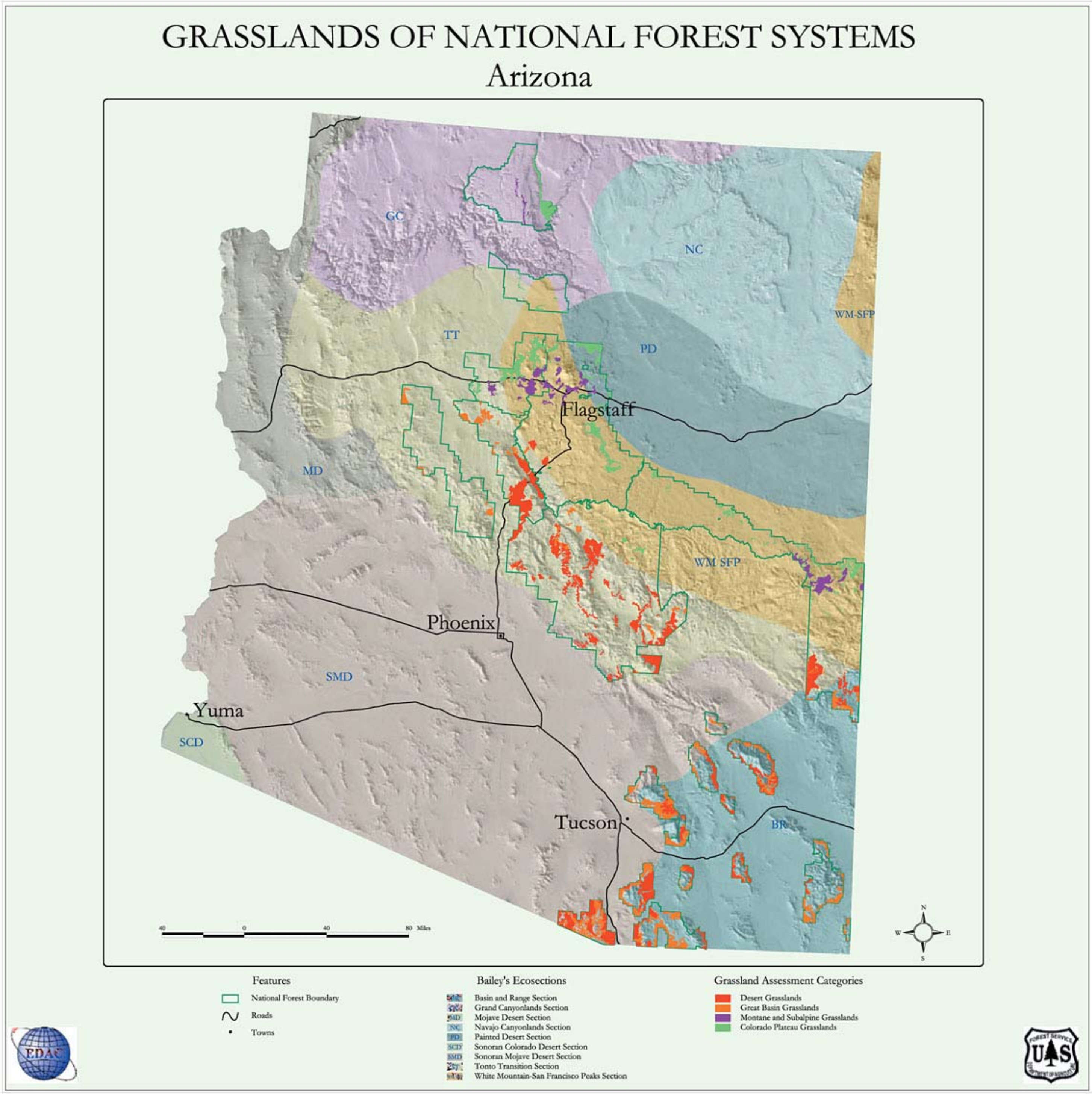

Figure 2-2.—Grasslands assessment category by National Forest and ecosection for Arizona.

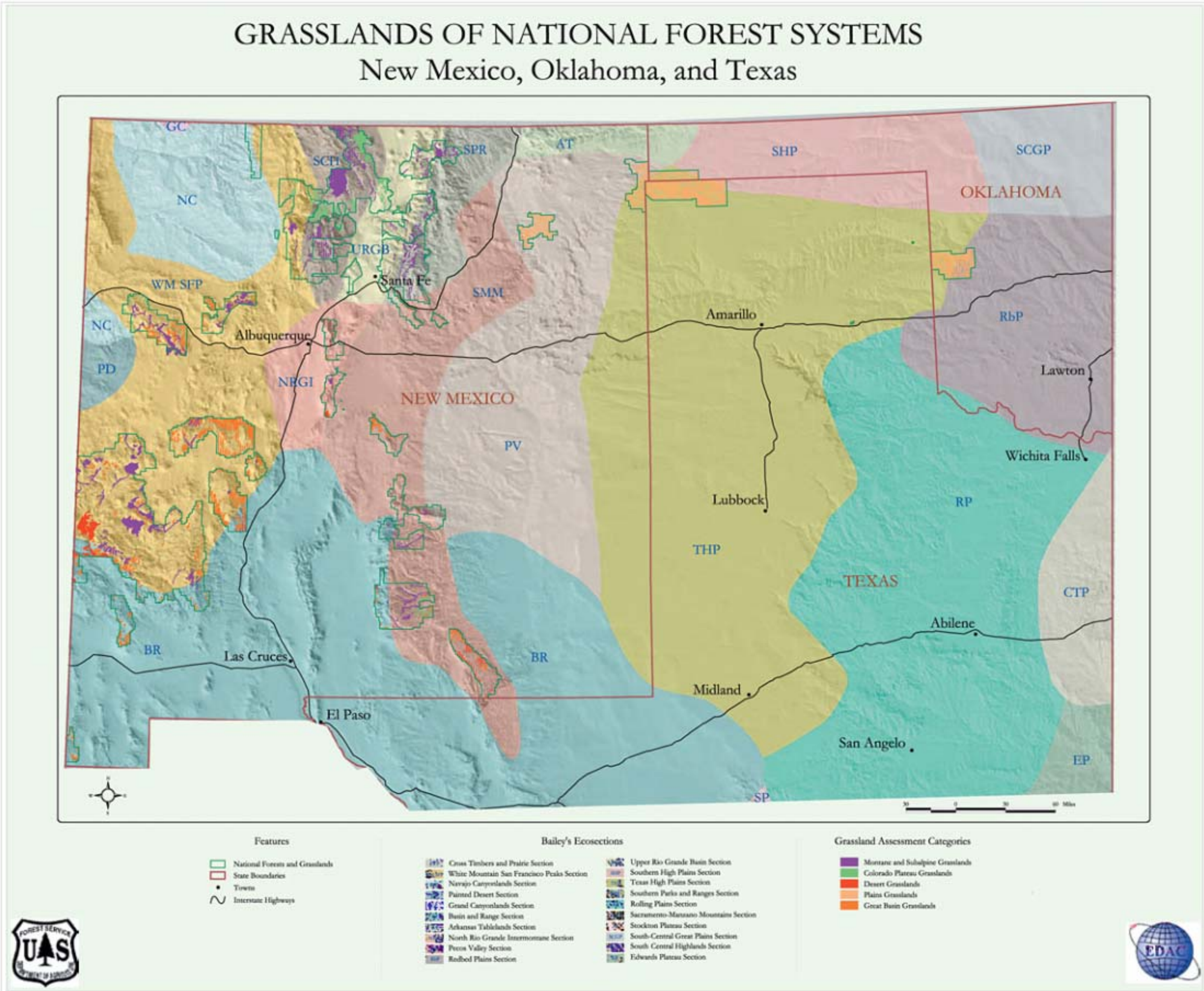

Figure 2-3.—Grasslands assessment category by National Forest and ecosection for New Mexico, Oklahoma, and Texas.

Table 2-1. Crosswalk of grassland assessment categories, Arizona and New Mexico GAP landcover classes and general terrestrial ecosystem survey vegetation categories.

Grassland assessment categories	Arizona GAP landcover classes	New Mexico GAP landcover classes	General terrestrial ecosystem survey vegetation classes
Montane	Rocky Mountain-Great Basin Dry Meadow	Rocky Mountain Alpine Forb Tundra Grasslands Rocky Mountain Alpine Graminoid Tundra Grasslands Rocky Mountain Subalpine and Montane Grasslands	*Kobresia myosuroides* *Festuca thurberi* *Festuca arizonica* *Bromus anomalus* *Poa pratensis*
Colorado Plateau	Great Basin Mixed Grass Great Basin Grass-Mixed shrub Great Basin Grass-Mormon tea Great Basin Grass- Saltbush Great Basin Riparian/Sacaton Grass scrub Great Basin Riparian/Wet Mountain Meadow Great Basin Sagebrush-Mixed Grass- Mixed Scrub Great Basin Shadscale-Mixed Grass-Mixed Scrub	Great Basin Foothill-Piedmont Grassland Great Basin Lowland/Swale Grassland Shortgrass Steppe	*Artemisia tridentata* *Bouteloua gracilis* *Hesperostipa neomexicana* *Pleuraphis jamesii*
Great Basin	Semidesert Mixed Grass- mesquite Semidesert Mixed Grass- mixed scrub Semidesert Mixed Grass- Yucca-Agave Semidesert Tobosa Grass Scrub	Great Basin Foothills-Piedmont Grassland Great Basin Lowland/ Swale Grassland	*Bouteloua curtipendula*
Plains	Not described	Midgrass prairie Shortgrass steppe	Not described
Desert	Sonoran Paloverde-Mixed Cacti Semidesert Grassland	Chihuahuan Foothill-Piedmont Desert Grassland Chihuahuan Lowland/ Swale Desert Grassland	*Prosopis glandulosa* *Prosopis velutina*

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Chapter 3:

Southwestern Grassland Ecology

Introduction

This chapter provides a brief overview, and selected in-depth coverage, of the factors and processes that have formed, and continue to shape, our Southwestern grasslands. In general, this chapter looks at how distributions of grasslands are regulated by soils and climate, and modified by disturbance (natural and/or anthropogenic). The attendant ecological components of grasslands will vary according to climate, soil, and other biotic factors including competition, predation, and mutualism. A shifting equilibrium typically exists between grasslands, deserts, and shrublands in the Southwest, such that changes in the severity or frequency of disturbance events (such as grazing, fire, or drought) can cause a change from one community type to another.

A problem of increasing importance to Southwestern land managers is that of exotic weed invasions. This chapter highlights the often-overlooked importance of fungi and soil crusts to grassland ecosystem function, and points out possible linkages to exotic weed invasions.

Grassland Evolution and Distribution

Humans have manipulated grassland vegetation for thousands of years through the use of fire, livestock grazing, and other means. Therefore, it can be difficult at times to separate our influence from that of nature in the formation and maintenance of grassland ecosystems. The extensive North American grasslands evolved in the Miocene and Pliocene, during a period of global climate change (Axelrod 1958). In the late Miocene, C_4 dominated Southwestern grasslands expanded at the expense of C_3 vegetation. CO_2 levels had decreased prior to the Miocene, and C_4 plants were more tolerant of the lower CO_2 than C_3 plants. Periods of increased aridity, caused in part by the uplift of the Asian land mass and changes in seasonal precipitation patterns, facilitated the rapid expansion of drought-tolerant grasses and forbs, while restricting the growth of forests and woodlands (Jacobs and others 1999, Pagani and others 1999). Subsequently, natural and aborigine-caused fires swept across the grasslands at sufficient frequency to restrict the occurrence of trees and shrubs (Axelrod 1985, Dix 1964, Erickson 2001). Regional climate change and the Industrial Era increase in atmospheric CO_2 might have played a role in the current expansion of woody ecosystems into grasslands (Brown and others 1997).

Currently, fire helps to maintain the stability of grasslands by reducing the establishment of trees and shrubs. It also facilitates nutrient cycling by releasing nutrients from litter (Odum 1997) and accelerating the rate of decomposition in the soil. The reestablishment of periodic fire as a disturbance agent

Plains Mesa/Desert grassland ecotone, Sandoval County, New Mexico. (Photo by Rosemary Pendleton)

can be fundamental to the ecological restoration of Southwestern grasslands. However, prior to proceeding with large-scale fire reintroduction, appropriate fire frequencies and season need to be determined for each grassland type. In general, the response of grasslands to fire seems to depend primarily on pre- and postfire levels of precipitation (Ford 1999). Therefore, the use of fire as a management tool in a drought year should be carefully considered and aligned with management goals.

An excellent resource to find out more information about fire effects on vegetation and wildlife in Southwestern grasslands is the Fire Effects Information System (FEIS) http://www.fs.fed.us/database/feis. The FEIS database contains literature reviews taken from current English-language literature of almost 900 plant species, about 100 animal species, and 16 Kuchler plant communities found on the North American continent. The emphasis of each review is on fire and how it affects each species (FEIS 2004).

Central North American grasslands, including mixed-grass prairie, shortgrass steppe, and desert grasslands of the Southwestern United States and Mexico, are considered temperate grasslands. Temperate grasslands, broadly distributed between 30° and 60° latitude, are generally characterized by (1) rainfall intermediate between temperate forest and desert, (2) a long dry season, (3) seasonal extremes of temperature (alternating long warm summers and short cold winters), (4) dominance of grasses, and (5) large grazing mammals and burrowing animals (Brown and Lomolino 1998, Lincoln and others 1998).

The periodic precipitation of temperate grasslands varies seasonally and annually. However, they generally average 250 to 750 mm of rain each year, though periods of drought are common and often prolonged. The rate of precipitation in temperate grasslands allows plants to release nutrients slowly into the ground over long periods. Dry temperate grasslands have maximum precipitation in the summer, and soil moisture is recharged by snowmelt in the spring. Precipitation in the arid and semiarid Southwestern grasslands is characterized by low rainfall, high evapotranspiration potential, low water yields (Branson and others 1981), and intermittent stream flow.

North American Plains grassland is represented mainly by mixed-grass prairie and shortgrass steppe in the Southwest. This largely midsummer flowering grassland extends from approximately 55° latitude in the Canadian Provinces of Alberta and Saskatchewan southward to below 30° latitude in Mexico, and once covered most of the American "Midwest" from the Eastern deciduous forest westward to the Rocky Mountains and beyond. More than 70 percent of the Plains grassland is now under cultivation (Garrison and others 1977). The plains grasslands developed under grazing by large herbivores and are generally tolerant of grazing (Engle and Bidwell 2000, Mack and Thompson 1982, Milchunas and others 1988).

Great Basin grassland merges with Plains grassland over a large transition area adjacent to the Rocky

Mountains in Montana, Wyoming, Colorado, New Mexico, and Arizona. Much of this grassland has been converted to cultivated cropland through irrigation, and most of the remainder has experienced a degree of shrub expansion due to grazing and fire suppression (Brown 1994). The spring-flowering Great Basin or Intermountain grassland is restricted to those areas west of the Rockies and east of the Sierra-Cascades that possess favorable soils, climate, and grazing history. Much of this grassland has been appropriately described as a shrub-steppe in that pure-grass landscapes without shrubs are limited (Franklin and Dyrness 1973).

In the Southwest, warm temperate grasslands are represented by a semidesert grassland with a more or less biseasonal to summer precipitation pattern. Since the 1970s, populations of woody plants, leaf succulents, and cacti have expanded, replacing perennial grass cover (Brown 1994). Factors attributed to changes in woody plant cover include regional climate shifts, increases in CO_2 concentrations, changes in fire frequency, and herbivory (Brown and others 1997, Detling 1988, Pagani and others 1999). Semidesert grassland adjoins and largely surrounds the Chihuahuan Desert, and with the possible exception of some Sonoran Desert areas in west central Arizona, it is largely a Chihuahuan, semidesert grassland. Extensive areas of this grassland occur in the Southwest in Chihuahua, western Coahuila, Trans-Pecos Texas, the southern half of New Mexico, southeast Arizona, and extreme northeastern Sonora (Brown 1994). Unlike the plains grasslands, desert grasslands were without megafaunal grazers for the last 10,000 to 11,000 years (Haynes 1991). Therefore the plant communities currently in place in the Southwest may be more susceptible to livestock grazing disturbance than other grasslands (Bock and Bock 1993, Loftin and others 2000, Mack and Thompson 1982). (For a comprehensive review of Southwestern grassland history and evolution see Van Devender 1995).

Soils and Climate

Precipitation and temperature are the main parameters of climate and are important properties that strongly influence both soil function and plant growth. Soil moisture and temperature directly affect the nature and development of soils. Grassland ecosystems are generally considered highly productive due to the hydrological, biological, and geochemical cycling between soil properties, and the resulting outputs of mass and energy. Soils vary considerably for grasslands in the Southwestern Region. The different grassland types in the Southwest result from unique combinations of climate, soil, topography, and parent materials. Given the wide geographic range of these ecosystems, a high degree of inherent variability of climates, geology, landforms, and plant communities exists that directly influences rates of weathering, degree of stability, and site productivity.

Additions to, removals from, and vertical transfers and transformations within the soil are all basic kinds of soil-forming processes. These are, in turn, influenced by natural and anthropogenic disturbances, both at the soil surface and within the soil profile (Hendricks 1985). Historic and current disturbances have had an effect on the rates to which soil-forming processes have taken place. They also affect the degree to which the grassland system maintains its resiliency and ability to sustain soil functions that reflect stability and productivity.

Desert Grasslands

At lower elevations of the Southwest, grasslands encompass the Chihuahuan and Sonoran Deserts and are characterized by an arid climate with limited precipitation. The seasonal distribution of rainfall differs between these two desert environments. Chihuahuan Desert rainfall is unimodal and receives the majority of moisture within the summer months during the monsoon season. Sonoran Desert rainfall is bimodal with two distinct periods of precipitation. One period is during the winter months in which half or more of the annual precipitation is received. The remaining rainfall is received during the summer months.

Soil temperature is an important property that has a strong influence on plant growth and soil formation. Two soil temperature regimes are recognized in desert grasslands that include thermic and hyperthermic

Tulip prickly pear *(Opuntia phaeacantha)*. (Photo by Rosemary Pendleton) classes. Generally the hyperthermic temperature regime occurs in Arizona where precipitation is 250 mm or less, whereas the thermic temperature regime occurs in both Arizona and New Mexico desert grasslands where precipitation ranges from 250 to 410 mm (Hendricks 1985).

Landforms and parent materials of desert grasslands vary according to the degree of weathering, slope, relief, and mode of transportation of geologic materials. Soils of these landforms have evolved under paleoclimatic conditions and continue to change through the influences of today's climate and disturbances. Fluvial erosion, deposition, and volcanic activity are the primary geomorphic processes responsible for the origin and development of landforms supporting desert grasslands.

Desert grasslands characteristically occur on gently sloping floodplains, lower alluvial fans, concave playas, bajadas, and nearly level valley plains. Some remnant desert grasslands occur on isolated terraces, mesas, and sideslopes of moderately steep and steep hills. Minor patches of desert riparian vegetation with grassland affinities occur along drainage ways and dissected alluvial fans. Typically, these landforms contain both igneous and sedimentary lithologies of alluvial origin, although areas within desert grasslands and adjacent desert shrublands have areas of eolian, wind-deposited features that yield undulating dunes.

Black grama (*Bouteloua eriopoda*). (Photo by Rosemary Pendleton)

Dominant soils in desert grasslands are classified as Aridisols. Vertisols are of minor extent for those desert grasslands occurring within playas and closed basin topography. Entisols are characteristic of valley plains and drainages where active fluvial processes are taking place. A number of diagnostic soil properties help differentiate these soil orders and influence the kind and amount of desert grassland vegetation and its location (Robertson and others 2000). The physical and chemical weathering of soils along with the atmospheric deposition of dust yields soluble salts and carbonates. Reaction (pH) of soils generally ranges from neutral to alkaline.

The interconnections of soil, climate, and vegetation relationships in an ecological framework are described by Carleton and others (1991). For instance, it is recognized that mesquite (*Prospsis* spp.) will occur as a shrub throughout the Chihuahuan Desert but is confined to the upper Sonoran Desert. Mesquite does occur in the lower Sonoran Desert as a tree on certain alluvial soils as a result of fluctuating water tables. The hyperthermic/aridic soil climate causes a lack of perennial grasses in the lower Sonoran Desert. The high concentration of calcium carbonate favors calciphillic plants such as creosotebush (*Larrea tridentata* (Sessé & Moc. ex DC.) Coville) and crucifixion-thorn (*Canotia holacantha* Torr.) (Hendricks 1985). The presence of heavy clay surface horizons in playas supports rhizomotous plants such as tobosa (*Pleuraphis mutica* Buckl.), whereas loamy soil textures are suitable for black grama (*Bouteloua eriopoda* (Torr.) Torr.), and sand soil textures are dominated by sandhill muhly (*Muhlenbergia pungens* Thurb.).

Major disturbances include fire, livestock grazing, recreational activities, and the introduction of exotic species. Composition and successional sequence of some desert grassland plant communities have changed due to the introduction of livestock grazing and exotic, predominantly annual, plant species. Recreational activities such as off-road vehicle use have altered the soils hydrologic function in certain areas. Climate-induced disturbances include lengthy droughts and flash floods during torrential summer monsoons.

Great Basin Grasslands

Soils of Great Basin grasslands differ from other grassland soils because of their unique combination of climate, landform, and vegetation. The soil climate of Great Basin grasslands ranges from an aridic to ustic moisture regime, and a thermic to mesic temperature regime; with combinations of ustic/thermic classes in central and southern Arizona (Robertson and others 2000), and aridic/mesic classes in central and southern New Mexico.

The region typically receives less than 250 mm of precipitation per year. Mean monthly precipitation

Indian ricegrass (*Achnatherum hymenoides*). (Photo by Rosemary Pendleton)

Blue grama (*Bouteloua gracilis*). (Photo by Rosemary Pendleton)

Needle-and-thread grass (*Hesperostipa comata*). (Photo by Rosemary Pendleton)

shows a strong winter-dominated pattern in the west, with a gradual shift eastward to more summer moisture, with less distinct wet and dry seasons compared with other deserts (Turner 1994). Air temperatures are cool in the winter months and hot during the summer months. The soil climate is quite variable and is dependent upon the fluctuations in weather patterns throughout the Southwest. With this degree of variability, the resulting genesis of geology, landform, and soils is complex. Yet, for this assessment, it is recognized that the geographic location of these grasslands is confined to the higher elevations of the Basin and Range physiographic province (Fenneman 1928): above desert grasslands and below or adjacent to pygmy conifer and evergreen oak woodlands, and therefore considered semiarid.

Volcanic and fluvial events are the primary geomorphic processes responsible for the origin and development of landforms supporting Great Basin grasslands. Landforms are typically nearly level elevated and lowland plains, gently sloping piedmont plains, and moderately steep uplands. These landforms vary in age and morphometric features, and they have experienced differing rates of erosion through anthropogenic and natural disturbances. These landforms also create a rainshadow effect along the frontal ranges of mountainous areas bordering basins and valley floors (Dick-Peddie 1993). Parent materials are derived from igneous to sedimentary sources. Mixed alluvium occurs in fan and piedmont positions and valley plains, whereas colluvium and residuum parent materials are dominant on elevated plains and upland landforms.

Soils are classified as Typic Haplustalfs, mesic or Typic Argiustolls, mesic for uplands (Carleton and others 1991). These soils are moderately deep to deep, with loam surfaces, and support blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), New Mexico Feathergrass (*Hesperostipa neomexicana* (Thurb. ex Coult.) Barkworth), and Needle-and-Thread (*Hesperostipa comata* (Trin. & Rupr.) Barkworth). Aridic Haplustalfs, thermic, and Aridic Argiustolls, thermic, are sightly drier and warmer soils that support blue grama (*Bouteloua gracilis*), black grama (*Bouteloua eriopoda*), and Threeawns (*Aristida* spp.). Sideoats grama (*Bouteloua curtipendula* (Michx.) Torr.), which occur extensively throughout this grassland and can be in association with emory oak (*Quercus emoryi* Torr.) at adjacent ecotones in southern Arizona and gray oak (*Quercus grisea* Liebm.) in southern New Mexico. Some inclusions of Typic Ustipsamments with sandy surface textures support little bluestem (*Schizachyrium scoparium* (Michx.) Nash). Another

Great Basin grassland indicator is Soaptree yucca (*Yucca elata* (Engelm.) Engelm.) (Dick-Peddie 1993). Western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve) can occur on Vertic Haplustalfs and Vertic Argiustolls on nearly level lowlands and concave swales (Laing and others 1987).

The effects of grazing, in combination with changes in climate over time, and the absence of natural fire, have resulted in encroachment of woody shrubs and trees into the Great Basin grasslands. Other disturbances include the increase in abundance of exotic weed species and the resultant change in fire regimes.

Colorado Plateau Grassland

Soils of Colorado Plateau grasslands of this assessment are characterized by an ustic bordering on xeric soil moisture regime and a mesic soil temperature regime. Typically, over half the precipitation occurs during the winter months in the form of snow and rainfall. However, some areas receive more summer than winter moisture. Air temperature is generally cold during the winter and hot during the summer.

The geographic range of these grasslands is limited to the lower elevations of the Colorado Plateau and Southern Rocky Mountain physiographic provinces (Fenneman 1928, Laing and others 1987, Miller and others 1995). These grasslands are an integrade to adjacent cold desert sagebrush (*Artemisia* spp.) or four-wing saltbush (*Atriplex canescens* (Pursh) Nutt.) steppe communities and higher pinyon (*Pinus edulis* Engelm.) -juniper (*Juniperus* spp.) woodland ecosystems (Brewer and others 1991).

The primary geomorphic processes responsible for the origin and development of landforms of the Colorado Plateau grasslands are fluvial and volcanic. Landforms are nearly level to flat mesas, plateaus, and rolling elevated plains. Parent materials are dominated by basaltic igneous and limestone or sandstone sedimentary sources. Mixed alluvium parent materials support deep soils in narrow valley plains, whereas residuum from both igneous and sedimentary sources support shallow to moderately deep soils in steeper landscape positions.

Aridic Ustochrepts, mesic, calcareous are moderately deep soils with sandy loam surface horizons. Subsurface horizons are fine-loamy textured or skeletal and are generally calcareous in the lower part of the profile. These soils occur on upland plains derived from limestone and support stands of big sagebrush (*Artemisia tridentata* Nutt.), fourwing saltbush (*Atriplex canescens*), winterfat (*Krascheninnikovia lanata* (Pursh) A.D.J. Meeuse & Smit), Needle-and-thread (*Hesperostipa comata*), and Indian ricegrass (*Achnatherum hymenoides* (Roemer & J.A. Schultes) Barkworth). Typic and Pachic Argiustolls, mesic are deep soils with loam to clay loam surfaces that occur on valley plains and lowlands derived from mixed alluvium. Dominant vegetation for these areas includes western wheatgrass (*Pascopyrum smithii*) and blue grama (*Bouteloua gracilis*).

The major disturbance of the Colorado Plateau grasslands is ungulate grazing. Composition and successional sequence of grassland plant communities have changed as a result of grazing. Climate induced disturbances include drought and flash floods during summer monsoon storms. Strong winds are common during the spring.

Plains Grasslands

The Plains Grasslands are within the Great Plains physiographic province (Fenneman 1928). Soils have ustic soil moisture regimes for shortgrass and midgrass communities and ustic bordering udic soil moisture regimes for tallgrass communities. Soil temperature is mesic for those areas of the plains that encompass the National Grasslands in eastern New Mexico, Oklahoma, and Texas. Climate ranges significantly in this region from semiarid to subhumid, which has a direct effect on the distribution of vegetation, along with the origin and genesis of the soils.

Pachic Argiustolls and Haplustolls occur on nearly level to gently sloping uplands and plains that are formed in aluvium, sandstone, and shale. These soils support mid- and tallgrass species. Typic Ustochrepts, Calciorthidic Paleustolls, and Psammentic Haplustalfs occur on sandy and loamy, calcareous recent alluvium along stream terraces, outwash plains and eolian deposits (Burgess and others 1963, Murphy and others 1960). The depth to a calcareous substrate varies according to the degree of weathering and precipitation. Stands of tall- and midgrasses, shin oak (*Quercus havardii* Rydb.), and sand sagebrush (*Artemisia filifolia* Torr.) frequently occur on these soils. Typic Ustifluvents are located along drainage ways, flood plains, and stream courses that are derived from recent alluvium. Fluventic Haplustolls that support tallgrass species occur in association with riparian hardwoods and wetland plants. Historically, the plains grasslands have evolved over time with repeated ungulate grazing in combination with natural fire.

Montane Grasslands

Soils of Montane Grasslands have ustic to udic soil moisture regimes, and temperature regimes that include frigid, cryic and pergelic. The Montane Grassland assessment category includes those environments characterized by montane, subalpine, and alpine ecosystems. Climates are extreme with mean annual air temperature ranging from –3 to 7 °C; mean annual precipitation ranges from 56 to 76 cm, of which over 50 percent is received during the months October through March. A significant portion of precipitation is snowfall.

The origin and development of landforms supporting montane grasslands are the result of glacial, fluvial, and volcanic geomorphic processes. Landforms associated with these grasslands include lowlands, upland plains, mountain slopes, and summit plains.

Typical Montane grasslands are characterized by Arizona fescue (*Festuca arizonica* Vasey) meadows on elevated plains of basaltic and sandstone residual, and alluvial parent materials that yield deep, clay loam, Typic, and Pachic Argiborolls (Brewer and others 1991, Laing and others 1987, Miller and others 1993, 1995). Other landform positions within this zone include valley plains of Kentucky bluegrass (*Poa pratensis* L.) supported by deep, loamy, Fluventic Haploborolls adjacent to riparian corridors of narrowleaf cottonwoods (*Populus angustifolia* James) growing in deep, sandy loam, Aquic Ustifluvents, frigid.

The Thurber fescue (*Festuca thurberi* Vasey) subalpine grasslands are higher in elevation than typical montane grasslands (Moir 1967). This environment is somewhat colder and wetter with a slightly shorter growing season. These grasslands are treeless expanses that border subalpine forests that are characterized by Subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), Engelmann spruce (*Picea engelmanii Parry* ex Engelm.), and aspen (*Populous tremuloides* Michx.). These meadows and grasslands occur on mountain sideslopes, elevated plains and valley plains derived from dominantly igneous with inclusions of sedimentary parent materials. Argic Pachic Cryoborolls and Haploborolls are soils that contain deep, dark loamy surface horizons (Miller and others 1993). Surface rock varies; however, internal coarse fragments are numerous, thereby classifying these soils as skeletal.

Alpine tundra ecosystems are located above timberline, where a harsh climate of extreme cold and wet conditions creates wind-swept treeless expanses along mountaintop positions. These plant communities are limited in the Southwest and occur at elevations above 11,500 feet. The growing season for this environment is less than 90 days, approximately June through August. Strong winds, high intensity solar radiation and severe evapotranspiration are characteristic of alpine tundra environments. Soils are dominantly Pergelic Cryumbrepts, which contain dark surface layers that have a low base saturation that does not meet the mollic epipedon criteria (Miller and others 1993, 1995). Shallow to moderately deep with very cobbly sandy loam surface horizons, these soils are susceptible to erosion from wind and water, and human trampling. The landforms include nearly level to steep summit plains and mountain slopes.

Soil Biota

A variety of small organisms exist in grassland soils including bacteria, actinomycetes, fungi, algae, nematodes, micro- and macroarthropod invertebrates, and larvae. These soil organisms play a vital role in maintaining Southwestern grassland function. Although the majority of species involved remain undescribed, soil microflora and fauna have profound effects on essential ecosystem processes such as decomposition, nutrient cycling, and the maintenance of soil fertility (Adams and Wall 2000).

Soil microorganisms are essential components of the biogeochemical cycles that sustain life (Wolters and others 2000). Below- and aboveground communities are inextricably linked through complex interactions and feedback mechanisms. Any disturbance or change in the environment that affects aboveground vegetation will also affect the soil biota. The reverse is equally true. For example, an increase in CO_2 concentrations above North American grasslands could disproportionately increase photosynthesis relative to decomposition, resulting in a buildup of organic matter in grassland soils (Swift and others 1998). This buildup would, in turn, cause functional shifts in the soil community as nutrient turnover rates declined.

Changes in climate (precipitation patterns or temperature) directly affect soil processes. An environmentally induced decline in the types of organisms capable of degrading complex organic molecules such as lignin or chitin could result in a buildup of surface litter and a reduction in the nutrients available for plant growth, thus effecting changes in the surface vegetation (Schimel and Gulledge 1998). Fortunately, soil organisms have a high level of functional redundancy that could help buffer many of the effects of global change (Andren and others 1995). However, some key processes—including nitrogen transformations, nitrogen fixation, and the breakdown of recalcitrant molecules—are controlled by unique or specialized organisms (Wolters and others 2000). Changes affecting these taxa may have profound effects on ecosystem functioning.

Above- and belowground organisms interact in a variety of ways. For example, (1) microorganisms and soil invertebrates are responsible for the breakdown of complex organic material into plant-available nutrients; (2) cyanobacteria, actinomycetes, and other rhizobacteria fix atmospheric nitrogen, thereby increasing soil fertility; (3) cell material and excretions from soil microorganisms affect the formation and stability of soil aggregates that, in turn, affect water and air movement within the soil; and(4) mutualistic relationships formed with mycorrhizal fungi affect a plant's ability to grow and compete. Consequently, changes that affect composition of the aboveground grassland community (that is, changes in climate, land use, atmospheric composition, or the introduction of invasive species) will likely affect the belowground community as well (Wolters and others 2000).

Bacteria, Actinomycetes, Fungi, and Algae

Bacteria are by far the most abundant group of soil microorganisms in terms of numbers, yet comprise less than half the microbial biomass because of their small size (Alexander 1977). Bacteria function in the decomposition of organic matter and in transformation and availability of many essential minerals. Several genera of free-living bacteria, such as *Azotobacter*, are capable of fixing atmospheric nitrogen. Vast communities of bacteria live on or near plant root surfaces where they feed on root secretions and dead cell material (Alexander 1977). Members of the genus *Rhizobium* form symbiotic relationships with roots of leguminous plants. Colonization by rhizobia results in the formation of root nodules, where they fix appreciable amounts of atmospheric nitrogen that is then available to the plant. Common legumes found on arid grasslands include the lupines (*Lupinus* spp.) and milkvetch or locoweed (*Astragalus* spp.). The woody mesquites (*Prosopis* spp.) also form nitrogen-fixing nodules (Geesing and others 2000).

Actinomycetes comprise the second most abundant class of soil microorganisms. They are extremely tolerant of desiccation and are found in large numbers in the grassland and steppe vegetation types common to the Southwest (Alexander 1977). Actinomycetes can utilize a variety of complex organic molecules including chitin and cellulose as energy sources. Many produce antibiotics and may be important in regulating the composition of the soil community. Members of the genus *Frankia* are capable of fixing nitrogen and form symbiotic root nodules with certain non-leguminous woody plants. Many of these species also form mycorrhizal associations (Rose 1980). These species are not common to grasslands, however.

Fungi account for large proportion of soil microbial biomass due to the extensive network of filaments (Alexander 1977). They function largely in the breakdown of complex organic molecules including lignin, a compound that is resistant to bacterial degradation. Some classes of fungi form specialized associations with plant roots. In addition to arbuscular mycorrhizal fungi, discussed in detail later in this chapter, Barrow and McCaslin (1996) observed two other major classes of fungi in roots of Southwestern grasses. Roots were commonly colonized by septate fungi that formed nondestructive interfaces within and among cortical cells, forming mycorrhizal-like associations. Chytridiomycete fungi were also commonly observed in plant roots. The function of chytrid fungi in the soil ecosystem is unknown, but they may play a role in regulating mycorrhizal colonization and nutrient uptake (Barrow and McCaslin 1996).

Soil algae are found in every conceivable location throughout the world, from the arctic tundra to thermal springs to deep within commercial caves, anywhere that light is present. Algae are often the primary colonizers following volcanic eruptions (Alexander 1977). The role of green algae and cyanobacteria (blue-green algae) in fixing carbon compounds and stabilizing soils is addressed later in this chapter. Another algal group commonly found in soils is the diatoms. These algae have an outer wall that is highly silicified. Although small, the striking beauty of their regular geometric shapes makes them a fascinating group to study. As with other kinds of algae, most diatoms are obligate photoautotrophs (requiring light) and live in the upper soil surface where light can penetrate. They are most prevalent in neutral to alkaline soils (Alexander 1977).

Soil Invertebrates

Many macroarthropods spend part of their life cycle, usually as larvae, in the soil, and many microarthropods, such as mites (Acari) and springtails (Collembola), spend virtually all their lives there. Soil- and litter-inhabiting mites and nematodes (Zak and Freckman 1991) and collembolans (springtails) (Crawford 1990) occur in vast numbers and are species-rich in nearly all Southwestern habitats. The soil mites (Acari) are the smallest of the arachnids, the group that contains spiders and their relatives. Some mites are large enough to be visible to the unaided eye (about the size of grains of ground pepper), but not large enough to allow determination of their morphological characteristics. The mite fauna in desert grassland soils include more than 30 families and 100 species (Cepeda-Pizarro and Whitford 1989). The most abundant are the generalist microbe feeders such as the nanorchestid and tydeid mites. Some mite species feed on nematodes as well as fungi, yeasts, and bacteria. Included in this fauna are a variety of predators that capture and eat other mites and nematodes (Whitford and others 1995).

Nematodes represent another group of extremely abundant microscopic soil animals. There are approximately 100,000 bacteria feeders, 1,000 fungus feeders, 10,000 omnivore-predators, and 10 to 1,000 plant root feeders (root parasites) per square meter of soil (Freckman and others 1987). Nematodes are aquatic animals confined to single-molecule-thick water films surrounding soil particles, although they are well adapted to deal with dehydration. As the water films disappear in a drying soil, the nematodes enter an inactive state called anhydrobiosis, which is immediately reversible when the soil is wetted

(Whitford and others 1995). Studies have indicated that nematodes play a substantial role in the cycling of carbon and nitrogen in the soil environment (Bongers and Ferris 1999).

Protists (single-celled organisms that feed on bacteria and algae) populate the soil of all desert grasslands, but practically nothing is known of their distribution and biology. Naked amoebae predominate, along with smaller numbers of other orders. The abundance of protists in desert soils is astounding: there are 25,000 naked amoebae, 4,900 flagellates, and 700 ciliates in every gram of dry soil (Parker and others 1984). These numbers are deceiving, however, because most of the protists in dry soil are encysted (in an inactive physiological state; Whitford 1989). After a rain they quickly resume activity and remain active until the soil dries and they once again encyst (Whitford and others 1995). Grazing by protists on microbes stimulates the rate of decomposition of organic matter (Finlay 2001).

Darkling beetle (*Eleodes longicollis*). (Photo by David C. Lightfoot)

Grassland Invertebrates

Invertebrates perform vital ecological functions in grassland ecosystems. The scope of their contributions includes soil aeration, seed dispersal, and plant pollination and consumption, in addition to facilitating the decomposition of organic debris (such as dung and animal remains). Invertebrates also provide an important prey base for grassland fish and wildlife. (See volume 2 for more information on grassland wildlife, small mammals, birds, and fish.) This section gives a brief overview of the roles of some of the common invertebrates in Southwestern grasslands.

The importance of invertebrate animals to ecosystem properties and processes is frequently underemphasized (Whitford and others 1995). When the indirect effects of invertebrates on nutrient cycling processes and their direct and indirect effects on soil heterogeneity are considered, however, their importance becomes more evident. In fact, ecosystem properties such as resilience (the ability to recover following disturbance) are directly affected by the activities of key invertebrate species (Whitford and others 1995).

The invertebrate fauna of desert grasslands, and Southwestern rangelands in general, is incredibly diverse and includes several phyla (Parmenter and others 1995, Whitford and others 1995). While mammal, bird, reptile, and vascular plant species occur in the tens to hundreds, invertebrate species in desert grasslands number in the thousands or tens of thousands. Many of the less conspicuous species have never been described by taxonomists (Whitford and others 1995), and little is known about the diversity of arthropods on Southwestern rangelands. The available data indicate that species diversity for most groups of rangeland arthropods is higher in the Southwest than in other parts of the country (Parmenter and others 1995).

Most of what we know about desert grassland invertebrates and their general life history characteristics is based on data from economically important species (Crawford 1981, Whitford and others 1995). Insects of the Southwestern rangelands are often thought of as agricultural pests because of the economically costly forage consumption by some species. Good reviews of important rangeland insect pests and research on those insects are found in Capinera (1987), Watts and others (1982, 1989), and Parmenter and others (1995). The pest species that are included in the above-cited literature represent only a small fraction of the insects and other arthropods that occur on Southwestern rangelands. Most species are not agricultural pests, many are rare, and many perform important functions in rangeland ecosystems (Crawford 1981, 1986, Lightfoot and Whitford 1990, MacKay 1991, Parmenter and others 1995, Walter 1987, Zak and Freckman 1991).

Grasshoppers (Orthoptera)

Many different species of plant-feeding insects occur on Southwestern rangelands (Crawford 1981, Watts and others 1989, Wisdom 1991). Of these, grasshoppers are among the most prevalent and conspicuous. A considerable amount of research has been conducted on grasshoppers throughout the Southwest, and more is known about the diversity and biology of grasshoppers than about other rangeland plant-feeding insects (Parmenter and others 1995). In North America, grasshopper species diversity is highest in the Southwest. Otte (1981) demonstrates that species densities of slant-faced grasshoppers (Gomphocerinae, primarily grass-inhabiting and -feeding grasshoppers) average

Regal grasshopper (*Melanoplus regalus*). (Photo by David C. Lightfoot)

around 30 species for locations in the Southwest, compared to five to 20 species for most of the rest of North America (Parmenter and others 1995).

Hewitt (1977) and Hewitt and others (1976) stated that grasshoppers compete with cattle for forage and that both herbivores have similar preferences in grass species. Estimates of the biomass consumption on rangelands in the Western United States by grasshopper herbivory have been difficult to calculate, but typically range from 6 to 12 percent of available forage, to as much as 50 percent in certain areas (Loftin and others 2000). In grassland at the San Carlos Indian Reservation in Arizona, Nerney (1961) found that grasshopper consumption ranged from 8 to 63 percent of the vegetation. However, Hewitt (1977) also reported a study from southeastern Arizona showing no correlation between grasshopper density and loss of forage. Quinn and others (1993) noted that as yet there is no measure of economic injury from grasshoppers that can be applied across all species or habitats (Loftin and others 2000).

Tarantula (*Aphonopelma* sp.). (Photo by David C. Lightfoot)

Beetles (Coleoptera), Ants (Hymenoptera), Termites (Isoptera), and Spiders (Araneida)

The order Coleoptera contains a third of all known insects—approximately 300,000 species worldwide and approximately 30,000 species in North America (Milne and Milne 1996). Beetles are conspicuous components of the terrestrial invertebrate fauna of arid and semiarid ecosystems of North America (Allsopp 1980). They have an important role in the functioning of Southwestern grasslands, serving as predators, prey, scavengers, and parasites. In addition, many plant-feeding species attack plants and stored foods, while others pollinate flowers and eat plant pests.

Beetles appear to be relatively diverse in Southwestern grasslands. For example, the tiger beetle (Cicindelidae) is found worldwide, but in North America, the Rocky Mountains and the Great Plains contain the highest numbers of species, 15 to 20, compared with 10 for New England and 15 for the Middle Atlantic States (Parmenter and others 1995, Pearson and Cassola 1992). Ford (2001) found an average of eight beetle species per hectare on unburned, ungrazed shortgrass steppe in the southern Great Plains of New Mexico. Burned areas contained an average of 11 beetle species per hectare (Ford 2001). Darkling beetles (Tenebrionidae) are more diverse in Western arid lands than elsewhere in North America and are major detritivores in the Southwest (Crawford 1990).

African dung beetles have been recently introduced into grasslands throughout the United States to speed the decomposition of livestock fecal pats, primarily with the goal of reducing numbers of nuisance and disease-transmitting insects that breed in the dung (Dymock 1993; Hoebeke and Beucke 1997). While the grasslands of the Central and Southwestern United States contain fecal pats, the overall habitat conditions are too hot and dry to support these species (Loftin and others 2000).

Ants originated in tropical areas and spread into temperate habitats. Many of the species found in the Western United States are not unique to the region (Holldobler and Wilson 1990, Parmenter and others 1995). Ants are the dominant arthropod predators (on other arthropods and on plant seeds) in some ecosystems (Holldobler and Wilson 1990). In some areas of the Chihuahuan Desert there may be as many as 4,000 ant colonies per hectare. Arid regions of the Southwest often contain a diverse ant fauna, with 23 to 60 species (MacKay 1991). For example, 23 to 50 species occur in the Chihuahuan Desert (Loftin and others 2000, MacKay 1991). Among the more conspicuous species are harvester ants in the genera *Pogonomyrmex* and *Aphaenogaster*. These species have large long-lived colonies (a decade or more) and contribute to soil mixing and aeration (MacKay 1991) and to seed dispersal (Loftin and others 2000).

Although termites occur primarily in the tropics, the species that are successful in temperate regions are of great importance in the recycling of nutrients in dead grass and wood (Loftin and others 2000, MacKay 1991, Schaefer and Whitford 1981). To avoid desiccation, temperate species are primarily subterranean, and their activity periods coincide with warm temperatures and an area's rainy season (Loftin and others 2000, MacKay and others 1989,).

Termites have low species richness (up to a dozen species in the Southwestern United States), but may be the greatest regional consumers of net primary production (MacKay 1991, Parmenter and others 1995). Bodine and Ueckert (1975) estimated that termites removed 20 to 50 percent of grass and plant litter from a grassland in Texas (Loftin and others 2000). Termites' positive contributions include nutrient cycling and nitrogen enrichment of the soils in arid and semiarid regions (Loftin and others 2000). Termites are known to consume cattle fecal pats (MacKay and others 1989).

Spiders form a major part of the arthropod fauna of the Southwest, but, as is the case with many other arthropod groups, the total number of species in the region is still unknown (Gertsh 1979). In a recent review of the status of arthropod systematics, Schaefer and Kosztarab (1991) estimate that most of the United States species of arachnids (and insects) that are still undescribed occur in the desert and montane Southwest and Great Basin areas (Parmenter and others 1995). Spiders serve as both predator and prey in Southwestern grassland ecosystems.

Mycorrhizal Fungi

The majority of plants in grassland ecosystems form some kind of mutualistic relationship with mycorrhizal fungi (Allen 1991, Miller 1987). In the relationship, soil resources are provided to the host plant root in exchange for energy-containing fixed-carbon compounds for the fungus. The fungus acts as an extension of the host root system. For example, 1 ml of soil may contain 2 to 4 cm of root, 1 to 2 cm of root hair, and 50 m of mycorrhizal fungal filaments, or hyphae (Allen 1991). Most external hyphae are concentrated near the plant root, but may extend 4 to 7 cm from the root surface (Read 1992, Rhodes and Gerdemann 1975). The small diameter of the hyphae allows the fungus to penetrate small pores in the soil and efficiently extract mineral nutrients such as nitrogen, phosphorus, potassium, calcium, sulfur, copper, and zinc (Allen 1991, Stribley 1987).

In addition to uptake of minerals, mycorrhizae have been shown to improve plant water relations and soil structure (Allen and Allen 1986, Ames and Bethlenfalvay 1987, Mathur and Vyas 2000, Miller and Jastrow 1994, Thomas and others 1993), and to reduce susceptibility to pathogens and nematodes (Habte and others 1999, Newsham and others 1995). The result is an increased ability for the host plant to survive and grow under stressful environmental conditions.

Mycorrhizal relationships are classified according to the morphology of the root/fungus interface and vary depending on the species of plants and fungi involved. In grasslands, the vast majority of mycorrhizas are of the arbuscular mycorrhizal type, formed with fungi of the order Glomales (Allen 1991, Miller 1987). Arbuscular mycorrhizae are so named for the internal fungal structure where nutrient exchange takes place. Initial fungal colonization can occur through germination of soil-borne spores, or through pieces of hyphae or infected root fragments. A hyphal network then develops between cells of the root cortex. Hyphae enter the cell wall of some cortical cells and branch dichotomously to form a profusely branched surface, known as an arbuscule, surrounded by the host cell membrane. Nutrient exchange takes place across this interface (Allen 1991, Bonfante-Fasolo and Scannerini 1992, Bowen 1987).

Other types of mycorrhizal associations may be found in isolated patches within the grasslands, or at the ecotone between grasslands and other vegetation types (Allen 1991, Trappe 1987). Ectomycorrhizal associations, formed between coniferous plants and basidiomycete or ascomycete fungi, are found at the periphery of montane grasslands. Ericaceous plants (for example, *Vaccinium* and *Gaultheria*), which form their own kind of ectendomycorrhizae, are found in the understory of coniferous forests and on rocky uplands within or adjacent to montane meadows. Willows present within the meadow may form either ecto- or arbuscular mycorrhizas. Orchids and saprophytes, found in a wide variety of habitats, also form a unique type of mycorrhizae.

The boundary between arid grasslands and other vegetation types is less distinct in terms of mycorrhizal associations. Desert shrubland plants, including *Larrea* and *Prosopis*, are also primarily arbuscular mycorrhizal (Staffeldt and Vogt 1975, Titus and others 2001, Virginia and others 1986). Juniper species, found at the upper boundary of arid grasslands, also form arbuscular mycorrhizae (Klopatek and Klopatek 1997, Lindsey 1984), facilitating their expansion into grass-dominated areas. Expansion of pinyon, which is ectomycorrhizal (Acsai 1989, Klopatek and Klopatek 1997), most likely depends upon the availability and dispersal of ectomycorrhizal inoculum.

Distribution and Occurrence

Arbuscular mycorrhizal fungi are found throughout the world, the same species often occurring on multiple continents. The fungi are grouped into three families and either five or six genera, based on wall characteristics of the soil-borne spores (Morton and Benny 1990). Associations between fungus and host plant appear to be nonspecific; a single root system may host associations with multiple species of fungi, and fungal networks may extend between adjacent mycotrophic plants. The fungi themselves are obligately mutualistic, and attempts to culture them separately have failed. Some species of fungi do appear to be habitat-specific, being associated with certain soil textures, nutrient levels, or extremes in pH (for example, *Glomus diaphanum* Morton & Walker and *G. spurcum* Walker & Pfeiffer), while others are common in a variety of habitats throughout the world (for example, *Glomus etunicatum* Becker & Gerdemann) (J. Morton, personal communication, Tatsuhiro and others 2000). Widespread or common fungal species may comprise different physiological ecotypes, however (Allen and others 1995).

Plant species differ in their ability to form mycorrhizal associations and in the amount of benefit derived. The degree of dependence on mycorrhizal fungi is correlated with the fineness of the plant root system and the number and length of root hairs per unit length (Baylis 1975, Hetrick and others 1992). Colonizing annuals in advanced families, such as the chenopod, mustard, and amaranth families, rarely form mycorrhizal associations and are referred to as "nonmycotrophic" (Allen 1991). Reeds and sedges in inundated soils also rarely form mycorrhizae; however, associations may form later in the season as soils dry out (Allen and others 1987, Miller and Sharitz 2000, Rickerl and others 1994).

Most grassland species are facultative mycotrophs and can survive with or without the association. They range from species that show no positive response to inoculation with mycorrhizal fungi, to those that show dramatic increases in growth and reproduction. The degree of response for any given plant depends on a number of factors, including the fungal species involved (fungal symbionts differ in their ability to acquire resources), soil texture and nutrient levels, environmental growing conditions such as temperature, precipitation, and light, and biotic factors such as competition and herbivory (Abbott and Gazey 1994, Cade-Menun and others 1991, Frey and Ellis 1997, Hetrick and Bloom 1984, Hetrick and others 1986, Koide 1991, van der Heijden and others 1998).

In general, C_4 or warm-season grasses are more dependent on mycorrhizae than are C_3 or cool-season grasses (Hetrick and others 1990, Van Auken and Bloom 1998, Wilson and Hartnett 1998). Warm-season grasses alter their root morphology in response to colonization (Hetrick and others 1991, 1994), a trait associated with mycorrhizal dependency. Some grass species (such as *Andropogon gerardii* Vitman) are highly dependent on mycorrhizae for normal growth and reproduction (Hetrick and others 1989). There are some indications that perennial grass species may be more responsive to mycorrhizae than annuals (Boerner 1992). Grassland forbs and shrubs also vary in mycorrhizal dependence, with genera such as *Linum*, *Sphaeralcea*, and *Artemisia* consistently showing a positive response to mycorrhizal fungi (Lindsey 1984, R. Pendleton, data on file, Rocky Mountain Research Station, Albuquerque, NM). Few Southwestern grassland species could be considered truly obligately mycorrhizal, that is, unable to grow and reproduce without the symbiosis.

Community Interactions

Considerable recent work supports the idea that mycorrhizae are involved in regulating plant community interactions by (1) increasing the fitness of mycotrophic plant species, (2) affecting the outcome of competitive interactions among species, and (3) connecting plant root systems via hyphal networks.

Most studies on the beneficial effects of mycorrhizae measure plant growth or biomass as the dependent variable. Plant fitness, however, is measured by the number and quality of offspring produced. Production of biomass is important only as it contributes to the plant's ability to survive and reproduce. Therefore, although plant growth effects in field studies are often negligible, the true benefit of mycorrhizal fungi may rest in the enhanced ability of a plant to survive "ecological crunch" periods of acute stress (Allen and Allen 1986, Fitter 1986, 1989, Trent and others 1993). Such periods may include the vulnerable seedling stage, periods of short-term drought, or episodic outbreaks of insects or disease. The presence of mycorrhizal fungi has also been shown to affect the reproductive capacity of a plant, influencing the timing of reproduction, quantity of seed produced, and quality (size and competitive ability) of the resultant offspring (Koide and Lu 1992, Lewis and Koide 1990, Shumway and Koide 1995).

The presence of mycorrhizal fungi can alter the outcome of interspecies competition among plants. This has been demonstrated most conclusively in competition experiments between mycotrophic and nonmycotrophic plants. In the presence of mycorrhizal fungi, the mycotrophic grasses *Pascopyrum smithii* and *Bouteloua gracilis* were able to outcompete non-mycotrophic *Salsola tragus* L., an exotic invasive of disturbed soils. In their absence, the reverse was true (Allen and Allen 1984). Similar changes in competitive outcome have been obtained using other grass-exotic weed, grass-grass, and grass-legume combinations that differed in their dependence on mycorrhizae (Benjamin and Allen 1987, Crush 1974, Fitter 1977, Goodwin 1992, Hall 1978, Hartnett and others 1993, Hetrick and others 1989, West 1996). Multispecies experiments have shown that the presence or absence of mycorrhizal fungi can alter species composition and diversity in grassland ecosystems (Grime and others 1987, Hartnett and Wilson 1999, van der Heijden and others 1998, Wilson and Hartnett 1997).

Species composition and diversity of the fungal community itself may affect plant community structure (van der Heijden and others 1998). Mycorrhizal fungal species differ in their ability to take up nutrients and promote growth (Haas and Krikun 1985, Jakobsen and others 1992, Stahl and Smith 1984, van der Heijden and others 1998, Wilson and Tommerup 1992). Fungal species are known to have a patchy distribution and may occur in specific microhabitats (Allen and MacMahon 1985, Johnson 1993). Therefore, individual plants may be colonized by the same fungal species, by different fungal species, or by multiple species of fungi (Rosendahl and others 1990, van der Heijden and others 1998).

The number and proportion of fungal symbionts can vary from plant to plant even within a relatively homogenous vegetation type. Differences among plants in their response to colonization, as well as in the complement of mycorrhizal fungi present on the root system, will differentially affect their ability to compete for soil resources (Streitwolf-Engel and others 1997, van der Heijden and others 1998).

Another factor affecting plant-plant interaction is that of interplant connections through a shared mycelial network. Studies have documented carbon and phosphorus transfer between plants connected by arbuscular mycorrhizal hyphae (Chiariello and others 1982, Francis and Read 1984, Newman and Ritz 1986, Whittingham and Read 1982). The ecological significance of these fungal connections is not well understood, however. Interplant connections may promote the transfer of nutrients from larger "source" plants to subordinate "sink" plants, thus allowing less competitive species to coexist with dominants (Allen 1991, Grime and others 1987). This scenario was invoked by Grime and others (1987) to explain why the addition of mycorrhizal fungi increased plant diversity in a microcosm experiment. Source to sink resource transfer may be of particular benefit to seedlings establishing in existing vegetation (Fitter 1989, Francis and Read 1984) and to plants growing in nutrient-poor soil patches (Allen and Allen 1990).

In contrast to the above theory, existing data from greenhouse competition experiments using mycotrophic species show an increase in competition in the presence of mycorrhizal fungi, rather than a cooperative interaction (Caldwell and others 1985, Hetrick and others 1989, Zobel and Moora 1995). There is great difficulty, however, in extrapolating data from greenhouse experiments with two species to actual field conditions, where the complexity of interaction is much greater. Zobel and Moora (1995) suggested that competition experiments might not include the critical stages of plant development (such as seedling and flowering stages) where source-sink transfer is thought to be most important.

Variation in precipitation patterns or patchiness in resource availability, factors rarely included in greenhouse experiments, can greatly affect mycorrhizal functioning and plant community composition (Allen 1991, Duke and others 1994, Hartnett and Wilson 1999, Pendleton and Smith 1983). Temporal and spatial niche partitioning due to differences in microhabitat requirements and in plant phenology limit the amount of actual competition experienced by a plant (Hetrick and others 1989). Certainly, extensive research is required before the role of mycorrhizae in regulating natural grassland communities can be well understood.

Disturbance and Succession

In grasslands, most arbuscular mycorrhizal (AM) hyphae and spores occur in the top 20 cm of the soil profile where the concentration of plant roots is the greatest (Allen 1991). Any disturbance resulting in the redistribution of these soils can affect the number and dispersion of mycorrhizal roots, hyphae, and spores. In most cases, soil disturbance leads to a reduction in the number of mycorrhizal propagules (Allen and others 1987, Jasper and others 1989, Moorman and Reeves 1979, Powell 1980). Small-scale natural disturbance such as gopher activity can create patches of low inoculum density as subsurface soils are brought to the surface (Koide and Mooney 1987). Inoculum density increases as hyphae from adjacent mycorrhizal plants slowly expand into sterile areas (Allen 1991). Spores may also be carried into these areas by crickets, grasshoppers, and other insects (Trappe 1981).

Large-scale disturbances such as surface mining, road construction, and watershed erosion can create extensive areas in which the amount of mycorrhizal inoculum is greatly reduced or eliminated (see for example Allen and Allen 1980). Mycorrhizal inoculum is introduced into these areas largely through wind-blown spores (Allen 1991). Consequently, the unaided restoration of mycorrhizal-dependent vegetation may require a long time (Miller 1987).

Changes in soil inoculum potential can affect a plant's ability to colonize disturbed areas. Many researchers have noted an increase in the abundance of mycotrophic species along a successional sequence (Allen 1991, Janos 1980, Miller 1987). Allen and Allen (1990) have proposed a successional model in which biomes are classified according to nutrient and moisture gradients. The degree of dependence on mycorrhizae increases with precipitation and decreases with enhanced soil fertility.

In arid grasslands, as in arid shrubland, early seral plant species are predicted to be largely nonmycotrophic, whereas late seral species are facultative in their dependence. These predictions are supported by numerous research reports. For example, Allen and Allen (1980) found five of seven annuals growing on strip-mined and disked prairie sites were nonmycorrhizal. Nonmycotrophic annuals predominated for up to 10 years on mined sites composed of sterile subsoil. Similarly, Pendleton and Smith (1983) found flat semiarid disturbed sites to be dominated by nonmycorrhizal species. Cover of mycorrhizal species increased with water availability. Soil fumigation can delay succession, suggesting that soil biotic communities, including mycorrhizal fungi, have a significant impact on successional dynamics (Stevenson and others 2000).

Little is known about successional patterns in mesic and dry alpine grasslands. The successional model of Allen and Allen (1980) would predict that all seral stages comprise facultative species, with perhaps some obligate species in late seral stages. Allen and others (1987) found all surveyed species colonizing disturbed alpine ecosystems to be mycotrophic. No relationship between age since disturbance and degree of colonization or spore number was apparent.

Species of the fungal symbiont also appear to change with succession (Allen 1991). Fungal communities of late successional grasses apparently differ from those of early successional grass species (Johnson and others 1992). Others report an increase in diversity of VA fungal species with increasing seral stage (Allen and others 1987, Brundrett 1996, Nicolson and Johnston 1979).

Grazing

The response of mycorrhizae to aboveground herbivore grazing varies widely depending on the plant and fungal species involved, on the type and intensity of grazing, and on a number of abiotic factors. Some studies have shown a decrease in mycorrhizal colonization of forage grasses as a result of livestock grazing or defoliation (Allsopp 1998, Bethlenfalvay and Dakessian 1984, Bethlenfalvay and others 1985, Wallace 1981). Others report either no effect or an increase in mycorrhizal activity (Allen and others 1989, Davidson and Christensen 1977, Tisdall and Oades 1979, Wallace 1987). In some instances, colonization by mycorrhizal fungi has been shown to enhance tillering and a prostrate growth habit, thereby increasing the plant's ability to tolerate grazing (Bethlenfalvay and Dakessian 1984, Miller 1987, Wallace and others 1982). Allsopp (1998) found that the response to defoliation by three mycorrhizal forage grasses depended on the grazing tolerance of the host plant species.

The response of the mycorrhiza to grazing is likely related to the carbohydrate storage and photosynthetic capacity of the host plant at the time defoliation occurs. Regulation of the association seems to be a function of the host plant (Koide 1993). And, although generally considered beneficial to the host plant, the fungi can be parasitic if the net cost of the symbiosis exceeds the net benefit (Johnson and others 1997, Koide 1993). Plants benefit from the association when photosynthate is readily available or when the association increases the plant's ability to photosynthesize (Bethlenfalvay and Pacovsky 1983). Therefore, when soil resources are more limiting than photosynthetic leaf area, herbivory may have little effect on mycorrhizal functioning (Allen 1991).

Under conditions that severely limit the amount of carbon a plant is able to fix and store—such as extremely low nutrient availability, insufficient light, or prolonged drought—the fungus may constitute an excessive carbon drain on the plant (Johnson and others 1997). Herbivory under these conditions would likely be accompanied by a reduction in mycorrhizal function. For example, in a pinyon-juniper ecosystem, defoliation caused a decrease in mycorrhizal function for plants growing in nutrient- and water-stressed volcanic fields, but not for plants growing in a nearby sandy loam (Gehring and Whitham 1995).

Prescribed burn on the Kiowa National Grassland, Union County, New Mexico. (Photo by Mike Friggens)

Fire

Response of mycorrhizal fungi to fire is variable, with some studies reporting a decrease in mycorrhizal function, while others report no effect or an increase in function (Allen 1991, Eom and others 1999). Of those studies reporting a temporary decrease in mycorrhizal function, whether in soil propagule numbers or percent root colonization, most had recovered within 1 to 2 years (Dhillion and others 1988, Gurr and Wicklow-Howard 1994, Pendleton and Smith 1983, Rashid and others 1997). Recovery rates depended in part on fire intensity and on soil conditions at the time of burning (Klopatek and others 1987, Wicklow-Howard 1986). Klopatek and others (1987) reported moderate decreases in propagule numbers when soil temperatures reached 50 to 60 °C. Significantly greater propagule loss occurred at soil temperatures greater than 60 °C. Colonization of plant roots also decreased following fire; however, the decrease was less in wetter soils.

The observed response of mycorrhizal fungi to fire may be a reflection of fire effects on the plants themselves, rather than a direct effect upon the fungi (Dhillion and others 1988). Soil provides good insulation. Riechert and Reeder (1971) reported no increase in soil temperature at a depth of 1 cm, despite surface temperatures of 200 °C. Dhillion and co-workers (1988) attributed temporary decrease in root colonization to a fire-induced stimulation of root production that temporarily outstripped colonization by the fungi. Much of the research on fire effects on mycorrhizal fungi has been done in tallgrass prairie, however. Additional research on arid grasslands is needed. Recovery of vegetation in arid grasslands is highly dependent on postfire precipitation patterns and generally occurs at a slower rate than in more mesic grasslands. The response of mycorrhizal fungi to fire is most likely reflective of the vegetation response, generally proceeding at a slower pace in arid areas and depending on the same suite of factors that affect vegetation response.

The interaction between fire, watershed stability, and mycorrhizal fungi needs further research. However, one of the most important factors affecting mycorrhizal recovery following fire is that of soil erosion. In areas where postfire erosion is significant (such as steep slopes or high-intensity fires), loss of topsoil may delay the recovery of mycorrhizal function (O'Dea and others 2000). Eroding topsoil carries with it a large proportion of mycorrhizal propagules. Seedlings planted on eroded soils had significantly less colonization, growth, and survival than those with additions of captured eroded soil (Amaranthus and Trappe 1993).

Atmospheric Change

Disturbance in the form of increasing atmospheric CO_2 concentrations and anthropogenic nitrogen deposition may also significantly affect mycorrhizal functioning within grassland communities (Egerton-Warburton and Allen 2000, Rillig and others 1999). Researchers are just beginning to explore these possibilities. McLendon and Redente (1991) reported a delay in successional change of plant species on fertilized plots. Early seral colonizers, nonmycotrophic *Salsola tragus* and *Bassia scoparia* (L.) A.J. Scott, still retained dominance of N-fertilized plots 2 years after control plots had changed to perennial (and mycotrophic) grasses, shrubs, and forbs.

Exotic Weeds

Noxious weeds compose a serious threat to the structure, organization, and function of ecological systems (Olsen 1999). Weeds prefer disturbed areas where resource availability is increased (Davis and others 2000). Of course, these are also areas where the mycorrhizal inoculum potential of the soil has been reduced. A healthy soil microflora can, in some cases, differentially enhance survival and production of native species over that of exotic weeds.

As mentioned previously, mycorrhizae generally enhance the competitive ability of those plants that are most responsive to the fungi. Many of the world's most aggressive weeds are either nonmycotrophic or facultatively mycotrophic (Trappe 1987). Invading annuals on arid grasslands are often nonmycotrophic weeds of the chenopod, mustard, and amaranth families (Allen 1991). Competition experiments between these species and native grasses have demonstrated enhanced competitive ability of native grass species in the presence of mycorrhizal fungi. Mycorrhizae contribute, therefore, to the resistance of these native communities to invasions of exotic weeds (Goodwin 1992).

Exotic grasses, on the other hand, are facultatively mycotrophic. Large areas in the Western United States are now dominated by exotic grasses such as cheatgrass (*Bromus tectorum* L.) in the Great Basin, Lehmann lovegrass (*Eragrostis lehmanniana* Nees) in the Southwest, and Mediterranean annual grasses in southern California (Allen 1995, McClaran 1995, Stylinski and Allen 1999). The role of mycorrhizal fungi in regulating competitive interactions between these exotic species and native vegetation is uncertain. It is likely, however, that competition between exotic and native grass species will be little altered by mycorrhizal fungi (Goodwin 1992).

In monoculture, cheatgrass shows no positive response to mycorrhizal fungi, although colonization levels can be quite high (Allen 1984, Benjamin and Allen 1987, Schwab and Loomis 1987). Cheatgrass may benefit, however, when the relative density of mycorrhizal competitors is high (Goodwin 1992, Schwab and Loomis 1987). Similarly, Nelson and Allen (1993) reported that the addition of mycorrhizae did not enhance competitive ability of purple nodding tussock grass (*Nassella pulchra* (A.S. Hitchc.) Barkworth) in competition with annual slender oats (*Avena barbata* Pott ex Link).

Unfortunately, in some cases, invasive nonnative woody or herbaceous species are more responsive to mycorrhizae than are native grasses. In these instances, the presence of mycorrhizal fungi may actually enhance the competitive ability of the invading species. For example, Marler and others (1999) report that AM fungi strongly enhance the competitive ability of spotted knapweed (*Centaurea biebersteinii* DC.) grown in competition with Idaho fescue (*Festuca idahoensis* Elmer). Clearly, the problems stemming from invasive weeds will increasingly challenge the resourcefulness, skill, and ecological knowledge of all managers and researchers concerned with the health of our Southwestern landscapes.

Ecosystem Restoration

The establishment of mycorrhizal associations is an important consideration in the design of successful revegetation and restoration efforts of grasslands. Miller and Jastrow (1992) list eight site conditions under which management for mycorrhizae may be particularly important, nearly all of which are applicable to Southwestern grasslands. Mycorrhizae may aid in restoration attempts by enhancing plant survival, increasing soil stability, and through their effect on competitive interactions of successional plant species. Mycorrhizae are also important contributors to stable soil structure and the redevelopment of nutrient cycles (Miller and Jastrow 1992).

As discussed above, anthropogenic activities that remove, compact, or otherwise disturb soil can severely reduce or eliminate mycorrhizal propagules. This, in turn, affects the ability of mycotrophic plant species to colonize the site. In addition to aiding plant survival and establishment, the reintroduction of mycorrhizae may allow late or mid-seral species to perform better than nonmycotrophic early seral species, thereby speeding up the rate of succession (Allen and Allen 1988, Reeves and others 1979). Because of this, mycotrophy could be an important factor in determining of seed mixes for grassland rehabilitation.

Severely disturbed sites such as mine spoils are particularly difficult sites for plant establishment, combining adverse growing conditions with a lack of viable mycorrhizal inoculum. Inoculation with VA fungi can greatly improve survival and growth of desirable reclamation plants, particularly shrubs, grown on mine spoil material and other severely disturbed soils (Aldon 1975, Lindsey and others 1977, Smith and others 1998, Stahl and others 1988, Williams and Allen 1983). Transplants may then form islands of inoculum that can spread through root and hyphal growth, as well as wind and animal dispersal (E.B. Allen 1984, M.F. Allen 1991). Reintroduction of mycorrhizae to severely disturbed sites may be enhanced through the use of soil amendments or redistribution of stockpiled topsoil, provided topsoil has been managed properly (Allen and Allen 1980, Allen 1984, Johnson 1998, Miller 1987, Pendleton 1981, Zak and Parkinson 1983).

Where severe soil disturbance makes it impractical to seed with mycorrhizal-dependent late-seral species, seeding with facultative midseral species such as cool-season grasses may help to increase soil inoculum to the point where more dependent warm-season grasses and shrubs can establish successfully (Noyd and others 1995). Johnson (1998) suggests that seeding of facultative mycotrophic species and manipulation of edaphic conditions to optimize mycotrophy may be a more cost effective method of rehabilitation than large-scale inoculation. The precise combination of techniques needed for successful rehabilitation will depend on soil conditions and the severity and scale of the disturbance, as well as management objectives (Allen 1995).

Biological Soil Crusts

Biological soil crusts are known by many names, including microphytic crusts, microfloral crusts, cryptogamic crusts, and cryptobiotic crusts (St. Clair and Johansen 1993). As the name implies, they are biological in nature, being composed of cyanobacteria, green algae, lichens, and mosses. Other bacteria and fungi (including mycorrhizal fungi) may also be present (Belnap and others 2001). It is important to distinguish biological soil crusts from inorganic soil crusts formed by chemical and physical means. Inorganic soil crusts reduce water infiltration and may hinder seedling emergence. In contrast, biological soil crusts play an important role in arid and semiarid lands by stabilizing soil surfaces, increasing soil fertility through the fixation of atmospheric nitrogen, and improving seedling establishment (Belnap and others 2001, Johnston 1997, St. Clair and Johansen 1993).

Biological soil crusts are formed by the interweaving of cyanobacterial and fungal strands within the upper soil surface. Gelatinous sheath material and other polysaccharides extruded by these organisms further help bind the soil particles together. The resulting crust may be anywhere from 1 mm to 10 cm thick (Belnap and Gardner 1993, Johnston 1997). The dominant photosynthetic organisms—cyanobacteria and green algae—require sunlight, and most of the living cells are found 0.2 to 0.5 mm below the surface (Belnap and others 2001). The polysaccharide sheaths of cyanobacteria expand when wet, pushing the interior bacterial filaments out across the soil surface. New sheath material is produced by these filaments as they dry out. Old sheath material gradually becomes buried and, although no longer photosynthetic, continues to contribute to the thickness and water-holding capacity of the crust (Belnap and Gardner 1993).

Biological soil crusts are highly variable in appearance and may be grouped according to their morphology. They may be smooth or rough, flat or pedicelled, greenish or black, depending on the temperature regime of the area and the types of organisms involved (Belnap and others 2001). Simple algal and cyanobacterial crusts may be flat. Topography of these crusts increases as the more complex lichens and mosses colonize the site. Increased topography of crusts is also associated with frost heaving (Johnston 1997). Crust morphology in hot desert regions, including the Chihuahuan and Sonoran Deserts, is flat to slightly rough or rugose. Cool desert regions such as the Great Basin and Colorado Plateau are dominated by rolling or pinnacled crusts (Belnap and others 2001). Different morphotypes differ in their ecological function with regard to water retention and erodability (Eldridge and Rosentreter 1999).

Soil crust composed of algae, lichens, and mosses. (Photo by Rosemary Pendleton)

Distribution and Occurrence

Biological soil crusts are found throughout the world, occurring on every continent and in multiple vegetation types. Although most well known from arid and semiarid regions, they also occur on shallow alpine soils and other areas where vascular plant cover is low. In the Southwestern United States, cyanobacteria, lichens, and mosses are the most important components of soil crusts (Ladyman and Muldavin 1996). Many of the most common organisms, such as *Microcoleus* spp., *Nostoc* spp., *Collema* spp., *Psora decipiens* (Hedwig) Hoffm., and *Catapyrenium lachneum* (Ach.) R. Sant., are cosmopolitan in nature, occurring in many geographic areas. Other species are endemic or have a much narrower geographic range (Belnap and others 2001, St. Clair and others 1993).

Regional trends in the timing and amount of precipitation greatly affect species composition of the crustal community. Regions in which precipitation falls mainly during the winter months have a diverse lichen community and a cyanobacterial community dominated primarily by *Microcoleus* (Belnap and others 2001). Great Basin areas are dominated by low-growing soil lichens such as *Collema tenax* (Swartz) Ach., *Catapyrenium lachneum*, and *Caloplaca tominii* (Savicz) Ahlner (St. Clair and others 1993). In contrast, crusts of the Great Plains are dominated by detached foliose (leaflike) or fruiticose lichens (Belnap and others 2001). Crusts of the Colorado Plateau are dominated by the cyanobacteria *Microcoleus*, along with the lichen *Collema*. Upland areas of the Colorado Plateau have many species of the genus *Psora* (St. Clair and others 1993). In regions characterized by summer rainfall, the lichen community is small and the cyanobacterial community diverse. Common cyanobacterial genera in the Sonoran Desert include *Nostoc*, *Schizothrix*, and *Scytonema* (Belnap and others 2001).

On a local level, species composition is largely influenced by soil texture, soil chemistry, slope, and aspect (Belnap and others 2001, Ladyman and Muldavin 1996). In general, silty loams support a greater diversity of crust organisms than do sandy soils (Belnap and others 2001). Green algae are more common on acidic soils, while blue-green algae (cyanobacteria) are more common on alkaline soils (Ladyman and Muldavin 1996). Gypsiferous and calcareous soils often have extensive crusts with a high diversity of organisms (St. Clair and others 1993). Alpine sites have many unique lichen taxa, as well as some that are common to the region (Belnap and others 2001). In the arid Southwest, crust development is often noticeably greater on north-facing slopes and near ridge tops (Brotherson and Masslich 1985, Pendleton, personal observation).

Ecological Function

Biological soil crusts contribute to a variety of ecological functions, including soil stabilization, nitrogen fixation, nutrient availability, and vascular plant establishment. Of primary importance is the ability of biological crusts to reduce wind and water erosion of soil surfaces (Belnap 1993, Belnap and others 2001, Eldridge 1993, Johnston 1997, Ladyman and Muldavin 1996). Within the crust, filamentous cyanobacteria and green algae intertwine with soil particles, stabilizing soil surfaces. Fungal strands, or hyphae, further bind soil particles, as do the rhizoids of mosses. Polysaccharides exuded by cyanobacteria, algae, and some fungi (see mycorrhizal section) cement soil particles together into stable aggregates. The increased surface topography of later successional crusts also serves to protect the soil surface from wind and water erosion (Belnap and others 2001, Johnston 1997).

Several studies have shown that crusted soils require much higher wind velocities before soil movement will occur (Belnap and Gillette 1998, Mackenzie and Pearson 1979, Williams and others 1995). Eldridge (1993) found a significant increase in splash erosion when biological soil crust cover was less than 50 percent. Loss of soil fines—particles associated with soil nutrients—also increased with decreasing crust cover. Moss-dominated and some lichen-dominated crusts increase infiltration of surface water, thereby decreasing erosion potential (Ladyman and Muldavin 1996 and references therein). Other studies report reduced infiltration, particularly in crusts dominated by cyanobacteria (Ladyman and Muldavin 1996 and references therein). However, even when infiltration is not increased, sediment loss is still reduced when compared with disturbed soils where the crust layer has been completely lost.

Biological soil crusts contribute to the fertility of grassland soils by increasing soil nitrogen and carbon content. This may be particularly important in arid areas of the Southwest where vegetation is sparse. Carbon contributions are greater where crusts are dominated by lichens and mosses, whereas nitrogen gains are greater where cyanobacteria and cyanolichens predominate (Belnap and others 2001). Carbon inputs to the soil come from active secretion of fixed-carbon compounds, as well as through the destruction of cell membranes during wet-dry cycles and upon death of the organism (Ladyman and Muldavin 1996). The amount of biomass contributed may be substantial. Beymer and Klopatek (1991) estimated carbon contributions of up to 350 kg/ha/year by undisturbed soil crusts in northern Arizona. Harper and Pendleton (1993) reported higher levels of soil organic matter in crusted areas of southern Utah.

Crust organisms are metabolically active only when wet. However, even small amounts of liquid moisture can trigger a rapid induction of metabolic processes. Available evidence suggests that positive carbon gains require a prolonged wet period (Jeffries and others 1993). In the Colorado Plateau, most growth of biological soil crusts occurs during the spring (Belnap and others 2001). Similar studies in areas of high summer rainfall would help define seasonal growth patterns for these regions. Because organisms differ in the temperature and moisture contents needed for maximum photosynthetic efficiency, regional shifts in the timing and amount of precipitation received may result in a compositional change in the crust community.

The cyanobacterial component of soil crusts, whether free-living or part of a lichen symbiosis, is capable of fixing atmospheric nitrogen. This process takes place only under anaerobic conditions. Many cyanobacteria have specialized cells for this purpose, called heterocysts. Nonheterocystic cyanobacteria, such as *Microcoleus* create anaerobic conditions by the layering of their filaments beneath the soil surface (Belnap and others 2001). Estimates of the amount of nitrogen fixed by soil crusts range from 2 to 365 kg/ha/year (Belnap and others 2001, Johnston 1997, Ladyman and Muldavin 1996, Rychert and others 1978). Fixation rates depend on temperature and precipitation patterns, as well as the species composition of the soil crust.

Arid regions are generally low in nitrogen content compared to other regions, and arid regions have few nitrogen-fixing plant species (Farnsworth and others 1976, Wullstein 1989). Studies using stable isotopes demonstrate that much of the nitrogen used by higher plants in arid and semiarid regions was originally fixed by soil crust organisms (Belnap 1995, Evans and Ehleringer 1993). Other studies report that plants growing on crusted soils have higher tissue concentrations of nitrogen than plants growing in nearby disturbed areas (Belnap and Harper 1995, Harper and Pendleton 1993).

The presence of biological soil crusts can significantly affect germination and growth of vascular plant species. Small seeds can lodge in the cracks of the roughened crust surface. The dark surface of the soil crust can raise soil temperatures 5 ^{o}C or more, promoting earlier spring germination and growth (Harper and Marble 1988, Harper and Pendleton 1993). The greatest benefit may be to shallow-rooted annuals and to deeper-rooted perennials during the critical establishment phase following germination. Many studies have reported increased survival and nutrient content of young seedlings growing in crusted soils (Belnap and Harper 1995, Harper and Marble 1988, Harper and Pendleton 1993, Pendleton and Warren 1995).

Nutrients enhanced by the presence of soil crusts include nitrogen, mentioned previously, and also P, K, Ca, Mg, and Zn. Enhanced nutrient content may be due to the tendency of biological crusts to trap nutrient-rich soil fines (Belnap and Gardner 1993). Cyanobacterial sheaths also secrete chelating agents that increase the availability of essential nutrients (see references in Belnap and others 2001, Harper and Pendleton 1993, Ladyman and Muldavin 1996). Biological crusts interact with other soil microorganisms in promoting the establishment and growth of vascular plant species. Harper and Pendleton (1993) found increased colonization by several rhizosymbionts (mycorrhizal fungi, *Rhizobium*, and rhizosheath organisms) in plants growing on crusted soils.

There is no credible evidence that cover of vascular plant species and biological soil crusts are negatively related, as was suggested by Savory (1988). Cover of biological soil crusts has been positively correlated with diversity of plant species (Beymer and Klopatek 1992, Harper and Marble 1988). Numerous other studies report either a positive correlation or no correlation between plant cover and the cover of biological soil crusts (Belnap and others 2001).

Ladyman and others (1994) examined vegetative cover on three undisturbed mesas in New Mexico and found that the mesa with the highest crust cover also had the highest grass cover. Specifically, blue grama grass (*Bouteloua gracilis*) was positively associated with soil crust cover. In another comparison between two grassland communities, Kleiner and Harper (1977) found no competition between soil crust organisms and vascular plant species. Rather, grass cover and crust cover appeared to be independent of one another. Ladyman and Muldavin (1996) suggest that specific associations of vascular plants and crust organisms that appear to be negative, positive, or neutral, may be more a reflection of their particular habitat requirements rather than actual competition. Greenhouse experiments, however, seem to indicate some competition between soil microorganisms and plants, particularly when crust organisms are themselves becoming established (Harper and Pendleton 1993, Pendleton, unpublished data).

Disturbance and Succession

Disturbance of the soil crust through trampling, vehicular travel, or soil movement affects the functioning of biological soil crusts through changes in cover, composition, or by affecting rates of carbon and nitrogen fixation (Belnap and others 2001, Ladyman and Muldavin 1996). The impact of the disturbance depends on the severity and type of disturbance, the frequency of disturbance, soil texture, and climatic conditions. Disturbance that compacts the soil crust but leaves crust material in place is less severe than disturbance that removes or kills crust organisms. Where the crust is destroyed, the loss of soil structure results in an unstable surface that is highly susceptible to erosion. Crusts formed on sandy soils are more easily damaged when dry, whereas crusts formed on clay soils are more vulnerable when wet. Lichens and mosses are more susceptible to disturbance than are cyanobacteria and green algae (Belnap and others 2001).

Repeated disturbance can result in a less complex (early successional) crust structure, or in the complete loss of the soil crust (Belnap and others 2001, Harper and Marble 1988). Estimates of recovery rates for soil crusts vary widely. Under optimal conditions visual recovery may occur within 1 to 5 years (Belnap 1993, Cole 1990, Johansen and Rushforth 1985). Recovery in terms of chlorophyll content, community composition, and organism density takes much longer. Belnap (1993) estimated recovery rates for Colorado Plateau crusts of 40 years for chlorophyll *a* content, 45 to 80 years for the lichen component, and 250 years or more for the moss component. Clearly, full recovery from severe disturbance can take a long time.

Crust development follows a definite successional pattern, with various organisms classed as either early or late successional species. Initial colonization of a site is usually accomplished by either cyanobacteria or green algae, depending on the acidity of the soils (Belnap and others 2001). A common colonizer of Southwestern soils is the large filamentous cyanobacteria, *Microcoleus*. These filaments help stabilize the soil surface, a function especially important for sandy soils, and allow colonization by other cyanobacteria and algae.

In hot deserts with summer monsoonal rainfall, different cyanobacteria, such as *Schizothrix* and *Nostoc* spp., may be more common (Johnston 1997). Once soils are stabilized, colonization by gelatinous nitrogen-fixing lichens, such as *Collema tenax*, can occur (Belnap and others 2001). Other early successional Southwestern species include the lichens *Cladonia chlorophaea* (Flk.) Spreng., *Endocarpon pusillum* Hedwig, and *Peltigera refuscens* (Weiss) Humb. Mid to late successional species include other lichen taxa (such as *Psora* spp.), mosses (*Grimmia* and *Tortula* spp.), and liverworts (*Riccia* spp.; Ladyman and Muldavin 1996). Because growth of crustal organisms can take place only when wet, succession occurs at a faster rate in years of high precipitation, on north-facing slopes, and on fine-textured soils that retain moisture longer (Belnap and others 2001, Ladyman and Muldavin 1996).

Monitoring of biological soil crusts has been suggested as a means of assessing ecosystem health. More recently, Eldridge and Rosentreter (1999) proposed a system of monitoring using morphological groups rather than species. Crust morphology determines its functioning in terms of water retention, erodability, and resistance to disturbance (Belnap and others 2001, Eldridge and Rosentreter 1999). This system provides an easier and more accurate method for nonspecialists to use in monitoring the effects of different management practices on ecosystem health. Specific techniques for monitoring biological soil crusts are covered in Belnap and others (2001).

Grazing

Different regions of the Southwest have different grazing histories, soils, and precipitation patterns. With the exception of the short-grass steppe, most regions historically experienced minimal disturbance by large herds of grazing mammals and may therefore be more susceptible to damage from grazing animals (Belnap and others 2001, St. Clair and others 1993). The effect of livestock grazing on biological soil crusts is due primarily to trampling, or hoof action. Trampling breaks up the sheath and filament structure that binds the soil particles, leading to increased erosion and loss of soil fertility.

The destruction of the crust through hoof action, while advocated by some (Savory 1988), does not result in increased plant cover as had been surmised (Ladyman and Muldavin 1996 and references therein). The magnitude of the disturbance is proportional to the grazing pressure and is also dependent on the seasonality of grazing. As discussed previously, biological crusts on most soils of the Southwest are more tolerant of disturbance when soils are moist. On these soils, late fall/early winter grazing is least likely to cause damage to crusts (Anderson and others 1982, Belnap and others 2001, Marble and Harper 1989). Several researchers have suggested that periodic rest from grazing would benefit crust organisms and grasses alike (Brotherson and others 1983, Johansen 1986). Belnap and others (2001) recommend light to moderate grazing in the early to mid wet season. Low-elevation grazing during the winter months followed by grazing of high-elevation sites during the summer would more closely mimic grazing patterns of native ungulates.

Fire

Fire has historically been a common occurrence in grassland ecosystems. Evidence from other vegetation types suggests that fire can damage soil crusts, depending on the fire's intensity, frequency, and timing, and depending on soil moisture content during and immediately following the fire (Belnap and others 2001, Ladyman and Muldavin 1996). High intensity fires, followed by drought, can result in substantial loss of the crust community, particularly lichens and mosses. Recovery may take many years. Unburned patches within the burned matrix can, however, provide propagules for reestablishment of soil crust organisms (Belnap and others 2001). Low intensity fires may allow for rapid regrowth of algae and cyanobacteria (Johansen and others 1993, Johnston 1997). Johansen and others (1993) theorized that recovery of the crust might be partially dependent on recovery of vascular plant cover. This suggests that recovery in fire-adapted grasslands may proceed at a faster pace. Adequate moisture during the postfire recovery period will further speed the recovery process.

Ford (2000) studied the effect of fire season on biological crusts in a short grass steppe. In the short term, growing-season fire appeared to reduce the impact of fire on microbiotic crusts ability to fix nitrogen, compared to dormant-season fire, due to the differences in the nature of the fire. Growing-season fires occurred when vegetation was green and moisture was high, whereas dormant-season fire occurred when moisture was low and fine fuel (vegetation) was dry. Again, adequate moisture appears to be the key component in crust recovery following fire and abundant precipitation will override any fire effects (Ford, personal communication).

Pollution

Many lichens are sensitive to common air pollutants and have been used as indicator species in other regions (Nash and Wirth 1988, St. Clair and others 1993). However, studies on Southwestern soil lichens found near pollution sources such as power plants have found no change in crust composition or cover (Belnap and others 2001, Ladyman and Muldavin 1996). It is thought that the low stature of soil lichens and the alkalinity of Western soils may ameliorate the effects of acidic sulfur emissions. Human-induced increases in atmospheric nitrogen deposition do have a negative effect on soil crusts, inhibiting natural fixation rates by soil microorganisms (Harper and Marble 1988). Fixation rates of cyanobacteria and cyanolichen crusts declined in the presence of power plant effluents (Belnap and others 2001). The effects of increasing atmospheric CO_2 levels have not yet been studied.

Exotic weeds

Little is known about the effects of exotic weed introductions on soil crust functioning. In areas where native bunchgrasses have been replaced by annual brome (such as *Bromus tectorum*), the soil crust consists of only a few species of annual mosses and cyanobacteria instead of the complex perennial crust community characteristic of undisturbed sites

(Kaltenecker 1997). This change in crust composition may be at least partially due to the drastic change in fire frequency that accompanies annual brome invasions. Similar studies are needed to determine the effects of lovegrass and other exotic weed invasions on composition and functioning of biological soil crusts in the Southwest.

Restoration

The use of soil inoculants may hasten recovery of soil crusts on severely disturbed sites. This would be particularly helpful for large-scale disturbances where nearby sources of naturally dispersed inocula are not available. Initial trials have been promising. St. Clair and others (1986) used a slurry of soil crust and water to inoculate pinyon-juniper and salt desert shrub communities following severe fire. Within 6 months, inoculated soils had greater numbers of algae and lichens. Belnap (1993) used crumbled dry crust material to inoculate scalped plots on four sites in the Colorado Plateau. Recovery of inoculated sites was significantly greater in terms of chlorophyll *a* content, crust cover, and species diversity 2 to 5 years following disturbance. These experiments relied on preexisting crust material taken from other locations. Johansen and others (1994) have developed techniques to culture nitrogen-fixing cyanobacterial species. Initial tests used intact or ground alginate pellets as the inoculum source (Buttars and others 1994, 1998, Johansen and others 1994). Currently, development of a more economically feasible delivery system is under way (Johansen, personal communication).

Invasive Weeds

For centuries weeds have been the focus of study for farmers and agronomists. Alien weeds have arrived in North American from a number of sources. Many species were introduced as contaminants of agricultural seed or as unnoticed hitchhikers that immigrated with people, household goods, and livestock to North America and subsequently moved with settlement across the continent. Other species were originally introduced as ornamental plants and subsequently escaped cultivation to become established in the native landscape. A third group of species are those intentionally introduced with the hope that they would fulfill a specific management objective, but that have since proliferated to the point where they now pose a serious problem (Brock 1998, Cousens and Mortimer 1995).

The invasion of alien species has been likened to a biological wildfire that is rapidly spreading at a rate of 200 acres/hour in the West (Lee 1999, Mitchell 2000). As the number of exotic species and acreage affected has risen, so has the concern of environmentalists and managers. Exotic weed invasions now command the attention of scientists and conservationists from a variety of backgrounds (Cousens and Mortimer 1995).

At the Federal level, President Clinton (1999) established a cabinet-level Invasive Species Council, which is charged with providing leadership in the management of invasive species. Executive Order 13112 outlines a far-reaching program designed to "prevent the introduction of invasive species and provide for their control and to minimize the economic, ecological, and human health impacts that invasive species cause." The executive order was the impetus for the current Forest Service weed management program. As Sheley and Petroff (1999) stated, our "commitment to addressing the rapid invasion of noxious weeds has been revived and intensified."

Among the definitions of a weed are the tried-and-true "a weed is a plant out of place" and "a weed is any plant that interferes with the management objectives of a site" (Lee 1999). Lee (1999) defines an *invasive* as a plant that is not native to a particular ecosystem, and an *invasive weed* as one that has or will have a negative impact on the environment. The terms "invasive" and "weed" are used with both native and nonnative species. While a species may be native to North America, it may become invasive in regions where it was not historically present.

The definitions used in this section are as follows. A *noxious weed* is a legal term that identifies an undesirable plant as one that is regulated in some way by law (Sheley and Petroff 1999). Executive Order 13112 (Clinton 1999) standardizes the terminology used by Federal land managers. *Alien species* are species not native to a particular ecosystem. *Invasive species* are alien species whose introduction causes or is likely to cause economic or environmental harm or harm to human health.

Problems stemming from invasive plant species are many. Biological invasions disrupt natural ecosystems, posing negative consequences for both ecological and socioeconomic systems (Davis and others 2000, Mitchell 2000). Invasive species displace desirable native species, reduce the quality of wildlife habitat, damage sensitive riparian and watershed areas, and increase erosion (Lee 1999, Masters and Sheley 2001). As native vegetation becomes displaced, opportunities for land use decline and land values drop. Improvement requires enormous inputs of time and money from managers and management agencies (Lee 1999). Less visible, but perhaps more important, are the alterations in ecological processes that accompany vegetation change (Masters and Sheley 2001). Changes in vegetation result in changes in the soil microflora, affecting nutrient cycling and decomposition rates (Adams and Wall 2000) as discussed in a previous section.

One of the greatest challenges facing scientists and managers is the unpredictable nature of biological invasions—why certain species become problems and where these problems are likely to occur (Peterson and Vieglais 2001). Invasive species have a number of characteristic traits that have been identified (Mitchell 2000). They are generally fast growing, produce numerous small seeds, and are adapted to disturbance (Grime 1988). Some, however, appear to be extremely competitive and can invade under conditions of low stress and low disturbance (Mitchell 2000). The presence and spread of nonnative species is thought to be an indicator of ecosystem health (Mitchell 2000), but even communities in good condition can be susceptible to weed invasions (Sheley and Petroff 1999).

Several hypotheses have been proposed in an attempt to explain why invasions take place (see Masters and Sheley 2001 and references therein for a review). Invasions seldom occur as a moving front, but rather by the establishment of small satellite populations that are often some distance from previously known populations (Cousens and Mortimer 1995). Noble (1989) theorized that displacement of native vegetation could occur either because the invader is a superior competitor, or because the invader is adapted to novel conditions present at the time of the invasion. Climatic change (for example, global warming or increased CO_2 levels) may constitute such novel conditions and contribute to the spread of some invasive species (Brock 1998). Lonsdale (1999) lists three factors that affect the ability of an alien species to invade a new environment: (1) the number of propagules entering the new environment, (2) the physiology of the invading species, and (3) the invasibility of the environment.

Invasibility of a given environment depends on a multitude of factors, including degree of disturbance and the health and makeup of the resident community (Lonsdale 1999). Each community appears to have its own invasibility criteria. Davis and others (2000) recently proposed a general theory of invasibility. They hypothesize that resources required by an invading species are not uniformly available through time, but rather fluctuate. These fluctuations may be caused by meteorological events and/or site disturbances such as grazing, pests, or mechanical disturbance. Therefore, the degree to which a community is susceptible to invasion varies through time, and invasion events are episodic. A number of modeling studies have attempted to predict where invasions will occur geographically, some with a fair degree of success (Peterson and Vieglais 2001).

Management

No universal prescription exists for managing invasive weeds that grow on Forest Service grasslands. Management plans take into consideration the biology of the weed, the number of plants involved, and the condition of the plant community. A single weed that has not yet flowered can be eradicated using several methods. But if that individual plant is allowed to set seed, eradication becomes more complex and will likely require more than one treatment. Without treatment, those seeds will produce a population, and as the seed bank and number of plants increases, eradication becomes exponentially more difficult. At some point in the growth of the population, eradication becomes impossible and alternative management methods become necessary (Brock 1998).

To provide guidance for appropriate management and control strategies, weeds are assigned an A, B, or C classification (Lee 1999). These class assignments are based on the distribution and population size of an invasive weed within a State or other land management area. Class A species have limited distribution within a management unit. Preventing new outbreaks and eliminating existing populations is the primary focus of management plans for class A weeds. Invasive weeds that are not present within a management unit but are found in adjacent areas, and therefore pose an invasive threat, are also listed as class A weeds. Class B weeds have well-established populations, but these populations are found only in limited areas. Containment within the current population location and preventing the establishment of new populations is the management focus for class B weeds. Class C weeds are widespread throughout a State or management area. Class C weeds are candidates for long-term management and suppression programs (Lee 1999).

The 1970s saw an increase in concern about the presence, expansion, and difficulty in controlling noxious and invasive weeds. These concerns—coupled with the increased costs of weed control, concerns about herbicide use, and general complexity involved in trying to control weeds on public and private lands—led to a new concept for the management of weeds. Termed Integrated Weed Management (IWM), this concept incorporates multiple management techniques into an integrated and well-planned strategy for management and control of weeds (Sheley and others 1999).

Walker and Buchanan (1982) define IWM as the application of technologies in a mutually supportive manner and selected, integrated, and implemented with consideration of economic, ecological, and sociological consequences. IWM involves several components: (1) prevention, (2) early detection and eradication of new weed populations, (3) containment and treatment of established invasive weed populations, and (4) re-vegetation and site rehabilitation. A successful IWM plan includes education, a constant sustained effort, evaluation of results, and improvement of management strategies as the plan is implemented (Lee 1999, Masters and Sheley 2001, Sheley and others 1999). Prior to the development of an IWM plan, survey and mapping of the management area must be done to identify existing invasive weed populations.

Prevention has two aspects. The first is to limit the movement of plant propagules to new locations. Some measures that limit propagule migration include the cleaning of seeds from equipment and vehicles, use of weed-free livestock feed, and efforts to prevent seed movement by other human activities. The second is the ongoing monitoring of the landscape so that new satellite populations are identified quickly and eradicated.

Management options for the treatment of existing weed populations fall into four basic categories: (1) chemical, (2) biological, (3) mechanical, and (4) cultural (Lee 1999, Masters and Sheley 2001, Sheley and others 1999). Chemical treatment involves the use of herbicides. A broad range of herbicides—registered for use on rangeland and grasslands—act upon various parts of the plant and at different life cycle stages of growth. It is extremely important to use the appropriate herbicide, one that has the maximum impact on the target species and minimizes impact on nontarget species (see Bussan and others 2001, Masters and Sheley 2001 for a review of herbicides and their uses).

Biological treatment uses living organisms to reduce weed populations. Alien plant species generally lack their complement of population-controlling insects and pathogens (Mitchell 2000). Biological control specialists collect potential control agents (insects or pathogens) from the country where the weed originated. After extensive testing, these are released into infested areas with the goal of suppressing the alien weed population. Eradication is not the goal of biological control, but rather the reduction of the population to more acceptable levels (Masters and Sheley 2001).

Mechanical (also called physical) treatment involves removal of aboveground plant parts and/or disruption of the root system such that the plant is killed or severely injured (Masters and Sheley 2001). Methods include hand pulling, tilling, plowing, and mowing. As with all forms of treatment, the method used must be appropriate to the biology of the target species. For example, tilling a weed that reproduces vegetatively will spread the infestation further, whereas tilling an annual weed prior to seed set may provide good control.

Cultural practices include all methods to promote the growth of desired vegetation. Healthy native plant communities are less susceptible to weed invasion. Cultural practices include fertilization, alteration of grazing practices, reseeding, revegetation, and other practices that promote the growth of desirable plant species (Masters and Sheley 2001). Cultural practices go hand in hand with other forms of treatment. When invasive weeds are removed from a community, steps can be taken to facilitate the establishment of desirable vegetation and prevent weeds from filling the recently vacated niche in the plant community (Masters and Sheley 2001).

Fire, both wildfire and prescribed, can impact weed populations either positively or negatively. Although fire can reduce the population of some weed species, the disturbance caused by fire provides opportunities for weed population expansion (Crawford and others 2001). The effect of fire on invasive species populations in the Southwest is the subject of ongoing research.

USDA Forest Service Southwestern Region

The Southwestern Region (Region 3) of the USDA Forest Service has a variety of grassland communities—from National Grasslands to high elevation montane grassland and meadow communities, with a number of grassland community types in between. All grassland communities in the Region are subject to invasion by weeds. Presently, the primary weed management focus of Region 3 is the preparation of National Environmental Policy Act (NEPA) documents, which are required prior to the formulation of weed management plans (D. Parker, personal communication).

The Region currently lists 38 species on its noxious weed list (table 3-1); 23 are classified as class A, five as class B, and 10 as class C. The total of 15 B and C species is a relatively low number compared to the number of well-established invasive weed species in States adjacent to the Region (R. Lee, personal communication). This fact, coupled with the list of 23 class A species, illustrates that there is still an opportunity to prevent many species of weeds from becoming well established in the Region. Prevention is the least expensive and most efficient form of weed management. Cox (2001, also available at http://web.nmsu.edu/~kallred/herbweb/newpage3.htm) provides a detailed list of the alien plant species of New Mexico. In addition, the Cooperative Extension Services of Arizona and New Mexico maintain current State lists of invasive species that can be accessed online.

Summary

Distributions of grasslands are regulated by soils and climate and modified by disturbance. A shifting equilibrium typically exists between grasslands, deserts, and shrublands of the Southwest, such that changes in the severity or frequency of disturbance

Table 3-1. Noxious weed list, USDA Forest Service Southwestern Region (Region 3). Life cycle classification; A=annual, B=biennial, P=perennial. Class refers to Region 3 management priority classification (see text).

Scientific Name	Common Name	Origin	Life Cycle	Class
Aegilops cylindrica Host	Jointed Goatgrass	So. Europe	A	C
Alhagi pseudalhagi Medicus.	Camelthorn	Asia	A	C
Asphodelous fistulosus L.	Onion Weed	Mexico	P	A
Cannabis sativa L.	Marijuana	Asia	A	C
Cardaria chalepensis (L.) Hand. –Maz.	Lens-podded Hoary Cress	Eurasia	P	A
Cardaria draba (L.) Desv.	Whitetop/Hoary Cress	Eurasia	P	A
Cardaria pubescens (C.A. Mey.) Jarmolenko	Globe-potted Hoary Cress	Eurasia	P	A
Carduus nutans L.	Musk Thistle	So. Europe	B	B
Centaurea calcitrapa L.	Purple Starthistle	Europe	A	A
Centaurea diffusa Lam.	Diffused Knapweed	Eurasia	P	A
Centaurea maculosa auct. non Lam.[1]	Spotted Knapweed	Eurasia	P	A
Centaurea melitensis L.	Malta Starthistle	Europe	A	B
Centaurea repens L. [2]	Russian Knapweed	Eurasia	P	A
Centaurea solstitialis L.	Yellow Starthistle	Europe	A	A
Cirsium arvense (L.) Scop.	Canada Thistle	Eurasia	P	A
Cirsium vulgare (Savi) Ten.	Bull Thistle	Eurasia	B	C
Convolvulus arvensis L.	Field Bindweed	Europe	P	C
Dipsacus sylvestris Huds.[3]	Teasel	Eurasia	B	A
Drymaria arenarioides Will.	Alfombrilla	Mexico	P	A
Eichhornia azurea (Sw.) Kunth	Anchored Waterhyacinth	Brazil	P	A
Euphorbia esula L.	Leafy Spurge	Eurasia	P	A
Halogeton glomeratus (Bieb.) C.A. Mey.	Halogeton	Asia	A	B
Hydrilla verticillata (L.F.) Royle	Hydrilla	So. Africa	P	A
Hyoscyamus niger L.	Black Henbane	Europe	B	A
Isatis tinctoria L.	Dyer's Woad	Europe	B	A
Kochia scoparia (L.) Schrad.[4]	Kochia	Asia	A	A
Linaria dalmatica (L.) P. Mill.	Dalmation Toadflax	Europe	P	A
Linaria vulgaris P. Mill.	Yellow Toadflax	Eurasia	P	A
Lythrum salicaria L.	Purple Loosestrife	Europe	P	A
Onopordum acanthium L.	Scotch Thistle	Europe	B	A
Peganum harmala L.	African Rue	No. Africa	P	B
Rorippa austriaca (Crantz) Bess.	Austrian Field Cress	Eurasia	P	B
Salsola iberica (Sennen & Pau) Botsch. ex Czerepanov[5]	Russian Thistle	Russia	A	C
Sonchus arvensis L.	Perennial Sowthistle	Eurasia	P	A
Sorghum halepense (L.) Pers.	Johnson Grass	Mediterranean	P	C
Tamarix ramosissima	Salt Cedar	Eurasia	P	C
Tribulus terrestris L.	Puncture-vine	So. Europe	A	C
Verbascum thapsus L.	Mullein	Asia	B	C

[1]= *C. biebersteinii* DC.
[2]= *Acroptilon repens* (L.) DC.
[3]= *D. fullonum* L.
[4]= *Bassia scoparia* (L.) A.J. Scott
[5]= *S. tragus* L.

events (such as grazing, fire, or drought) can cause a change from one vegetation type to another. Humans have manipulated grassland vegetation for thousands of years through the use of fire, livestock grazing, and other means. It is therefore difficult to separate human influence from that of climate and other factors in the formation and maintenance of these ecosystems.

Southwestern grasslands are generally characterized by low to intermediate rainfall, a long dry season, seasonal extremes of temperature, dominance of grasses, and large grazing mammals and burrowing animals. Because some Southwestern grasslands developed under grazing by large herbivores, they are generally tolerant of grazing. Other Southwestern grasslands are thought not to have had a long evolutionary history of grazing and might be more susceptible to grazing disturbance.

Soils vary considerably for grasslands in the Southwestern Region because of different climate, vegetation, topography, and parent materials. Small organisms that exist in grassland soils include bacteria, fungi, algae, and nematodes. These soil organisms have a profound effect on essential ecosystem processes such as decomposition, nutrient cycling, and maintenance of soil fertility. Arthropods also perform vital ecological functions in grassland ecosystems. Some of their contributions include soil aeration, seed dispersal, and plant pollination, in addition to facilitating the decomposition of organic debris. Invertebrates also provide an important prey base for wildlife.

The majority of plants in Southwestern grasslands form some kind of mutualistic relationship with mycorrhizal fungi. These range from plant species that show a negative response to inoculation with mycorrhizal fungi, to those that show dramatic increases in growth and reproduction. Because of this differential response, the quantity and type of mycorrhizal fungi present in the soil can affect plant community interactions by influencing the relative fitness and competitive ability of mycotrophic plant species. Biological soil crusts also play an important role in the Southwest. They stabilize soil surfaces, increase soil fertility through the fixation of atmospheric nitrogen, and improve seedling establishment. The cover of biological soil crusts has been positively correlated with diversity of plant species. There is no credible evidence that plant cover and biological soil crusts are negatively related.

Biological invasions disrupt natural ecosystems, posing negative consequences for both ecological and socioeconomic systems. They displace desirable native species, reduce the quality of wildlife habitat, damage sensitive riparian and watershed areas, and increase erosion. As native vegetation becomes displaced, alterations in ecological processes occur including changes in vegetation, soil microflora, nutrient cycling, and decomposition rates.

No universal prescription exists for the management of invasive weeds growing on Forest Service grasslands. Weed management plans currently being written by the Forests of the Southwestern Region will provide guidelines for addressing the complex problem of invasive weeds.

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Chapter 4:
Biodiversity, Functional Processes, and the Ecological Consequences of Fragmentation in Southwestern Grasslands

What is Biodiversity?

Concern over accelerating extinction rates and loss of species diversity on a global scale was the subject of E.O. Wilson's seminal volume *Biodiversity* (Wilson 1988). This work essentially transformed the term "biodiversity" into a household word as a short-hand for species diversity—or more simply, the full array and variety of living organisms on Earth. But the term biodiversity means much more than the complement of plant and animal species that one expects to find in some given area. The term now encompasses not only the diversity of species, but their genetic structure, the interaction of the biotic and abiotic components of the environment at the ecosystem level, and at an even higher level the array of communities and ecosystem processes and functions that make up the landscape or regional level of biological diversity. Such a detailed exploration of biodiversity is beyond the scope of this section.

In the interest of (relative) brevity, the following discussion will focus on the more limited definition of biodiversity, concentrating on the variety of plant and animal species of the New Mexico and Arizona grasslands.

A few points of clarification on terminology: The most common indices of species diversity (such as Simpson index, Shannon-Weiner index) are a function of two parameters: (1) species richness, or quite simply the number of species that occur in a defined area; and (2) species evenness, a measure of the extent to which the individuals of the different species present are equally abundant. Diversity is thus a measure of species richness weighted by relative abundance. Technically the most highly diverse communities are those with the greatest species richness, each component species being equally abundant within the community. In reality, such a situation is unlikely to occur, as many species (such as top predators) are naturally less numerous in the community. Furthermore, from a management standpoint it is often the more rare species in the community that are of interest, rather than the common species that tend to dominate weighted indices of diversity. The term "biodiversity," then, as used by many biologists, most often refers to the more simple measure of species richness, and that is how the term is used in this discussion. Biodiversity as used here refers to *natural* or *native* biodiversity. The distinction is important because the introduction of exotic plants or animals may technically increase diversity (at least in the short term) by adding to the overall species richness of a given area. However, as discussed later, introduced species usually interfere with normal community and ecosystem functions at some level and often eventually replace the native species, thereby leading to a net loss of native biodiversity.

Why is Biodiversity Important?

There are two main schools of thought on this subject. One theory is commonly referred to as the "rivet hypothesis" (Ehrlich and Ehrlich 1981). This hypothesis proposes that each species plays some small but important role in the ecosystem – like the rivets that collectively hold an airplane together. After some number of species (rivets) are lost, a critical threshold is crossed and the system (plane) rapidly begins to disintegrate. The fundamental assumption is that greater diversity (more rivets) results in increased stability of the system.

The other theory may be called that of "functional redundancy." This idea holds that communities comprise a few functional groups of ecologically equivalent species, to the effect that several species may be lost with little impact because there are several "back-up" species in place that are capable of carrying out the same function in the ecosystem (Walker 1992).

Although sources of great debate, there is not a great deal of hard data to back up either theory. Studies in grasslands to date, however, tend to support the rivet hypothesis. Grassland plots with greater species diversity were found to be more resistant to drought and to recover more quickly than less diverse plots (Tilman and Downing 1994). Another grassland experiment showed that plant productivity and soil nitrogen utilization both increased significantly as a function of plant species diversity, leading the authors to conclude that "the establishment and functioning of these grassland ecosystems depended on their species richness" (Tilman and others 1996:720).

How, then, to manage for biodiversity in grassland ecosystems? With so little evidence, the assumption of functional redundancy seems somewhat reckless. Even if there is some degree of ecological overlap between species, we have no idea if the "backup" system is as efficient as the primary one (Odum 1992). The most prudent course of action in Southwestern grasslands – or any ecosystem—is to ensure the ecological integrity of the system by managing for the conservation of maximal native biodiversity, or, in the oft-quoted words of Aldo Leopold, "to save every cog and wheel"—or, to see the context of his remarks (Leopold 1953):

> The last word in ignorance is the man who says of an animal or plant: "What good is it?" If the land mechanism as a whole is good, then every part is good, whether we understand it or not. If the biota, in the course of aeons, has built something we like but do not understand, then who but a fool would discard seemingly useless parts? To keep every cog and wheel is the first precaution of intelligent tinkering.

Biodiversity in the Southwest

Although at first glance it might seem that the climatic extremes of the desert Southwest would prevent many species from being successful in this environment, in fact the opposite is true. The highly variable precipitation, temperature extremes, and elevational gradients created by the basin and range topography have provided numerous opportunities for adaptation and evolution. Small-scale variations in soils, aspect, and moisture all affect microclimate and resource conditions, and this variety of conditions in turn offers a diverse array of niches available for exploitation, resulting in increased floral and faunal species richness. Species diversity in the Southwest has been further enhanced by its biogeographic history. The flora and fauna of distinctive regions historically isolated during the Pleistocene (for example, Great Plains, Chihuahuan Desert, Sonoran Desert, Mojave Desert, and Great Basin Desert) have converged following glacial retreats, leading to the characterization of the desert Southwest as a "biological melting pot" (Parmenter and others 1995). For Southwestern grasslands in particular, biodiversity is further enhanced by the complex, mosaic nature of their distribution. Grasslands in this region are interspersed with shrubs, woodlands, and riparian areas, leading to high species diversity due to the presence not only of grassland specialists, but also more generalist species from adjacent habitats that may utilize the grasslands for other purposes (Parmenter and Van Devender 1995). Highly patchy interspersed habitats, such as sand dunes, playa lakes, and lava flows, add to biodiversity by the presence of specialized and/or endemic plant and animal species distinctive to these areas (such as fringe-toed lizards in sand dunes) (Parmenter and others 1995). Brown and Kodric-Brown (1995) argue that biodiversity along the Arizona and New Mexico borderlands may be among the greatest on the continent due to the biogeographic confluence in this region.

Underlying the considerable biodiversity of the Southwest at the level of species richness is a high degree of genetic variability. Rangeland plants as a group exhibit high levels of genetic diversity, probably due to a combination of adaptation to diverse ecological conditions and frequent hybridization between interfertile species (Nevo and Beiles 1989, Wayne and Bazzaz 1991). Modern grassland plants have survived historical cycles of great climatic variation; high levels of genetic variability have given these plants the ability to adapt and persist throughout such oscillations (Tausch and others 1993). Genetic diversity is also high in Southwestern animals, both vertebrates and invertebrates. Pocket gophers (Geomydiae), for example, are common mammals in Southwestern grasslands. Although the species often are difficult to distinguish visually, they are some of the most genetically variable mammals known; species may differ not only in terms of the alleles represented, but in the numbers of chromosomes carried as well (Parmenter and others 1995, Patton and Sherwood 1983). Despite the fact that Southwestern mammals are relatively well known, modern laboratory techniques continue to identify new species on the basis of genetic differences between populations. Within the last few decades several new species have been identified in New Mexico, including a grasshopper mouse (*Onychomys arenicola*), a meadow-jumping mouse (*Zapus hudsonius*), and a deer mouse (*Peromyscus gratis*) (Hafner and others 1981, Modi and Lee 1984). Similar patterns of high genetic variability have been documented in Southwestern invertebrate species as well, including fruit flies and grasshoppers (Dobzhansky 1944, Rentz and Weissman 1980).

Plant Diversity

Plant diversity in the Southwest is high relative to that elsewhere in the country (Brown 1982), due in large part to the factors discussed above (environmental variability, biogeographic history, and genetic mixing), as well as the convergence of temperate and subtropical species along the border with Mexico. In addition to representation of plants from different regions, the Southwest supports a high number of endemic species. In the grasslands of the Chihuahuan Desert, for example, two-thirds of the grass species may be considered endemics (Burquez and others 1998). The presence of not only grasses, but forbs, shrubs, and occasional trees as well, adds to the diversity of the grassland community (Burgess 1995, McClaran 1995). The richness of plant species present is considered to be one of the most important indicators of overall rangeland health (West 1993). Diversity in plant communities reflects much more than just the variety of species present; the array of cover values, patchiness, and densities of plants all contribute to vegetative diversity (Moir and Bonham 1995). Species composition is affected not only by factors such as soil characteristics, precipitation, and topography, but also by the frequency and intensity of disturbances such as fire or livestock (Burgess 1995). In general the identity of the plant species present tends to remain constant over time (not accounting for introductions of exotics), but the relative abundances of these component species may change dramatically, thereby leading to alterations in the physical structure of the grassland system (Westoby and others 1989). The vegetative architecture of the ecosystem is a particularly important consideration, as it is one of the key characteristics influencing the use of the habitat by both invertebrate and vertebrate animals (Parmenter and others 1995).

About 10,000 species of grasses exist worldwide, making the grass family Poaceae the fourth most speciose family in the plant kingdom (behind the asters, legumes, and orchids; Smith 1993, Watson 1990). If one considers grasses in terms of their contribution to range quality, Stubbendieck and others (1986) suggest 94 species of grasses in their list of the 200 most important range plants in North America. Some of the more common native grasses of the Southwest include several species of gramas (such as black grama [*Bouteloua eriopoda*], blue grama [*B. gracilis*], hairy grama [*B. hirsuta*]), bush muhly (*Muhlenbergia porteri*), mesa dropseed (*Sporobolus flexuosus*), tobosa (*Hilaria mutica*), and Arizona cottontop (*Digitaria californica*). Some native grasses have nearly been extirpated from their former ranges, such as the giant sacaton (*Sporobolus wrightii*), a bunchgrass that once grew 1 to 2 m high across the floodplains of the Southwest. In this climate characterized by low rainfall, high rates of evapotranspiration and shallow soils, more than 95 percent of the grass production in desert grasslands is from C_4 species (Sims and others 1978). Desert grasslands are typically composed of a mixture of perennial and some annual grasses, most of the dominants being perennial caespitose bunchgrasses such as blue grama interspersed with suffretescent grasses such as bush muhly. Some of the common desert grasses are sod forming, such as curly mesquite grass (*Hilaria belangeri*) (Burgess 1995). These may be interspersed with small trees or shrubs such as mesquite (*Prosopis* spp.), creosote (*Larrea tridentata*), sages (*Artemisia* spp.), saltbush (*Atriplex* spp.), or rabbitbrush (*Chrysothamnus* spp.), subshrubs such as snakeweed (*Gutierrezia* spp.), and succulents or cacti such as sotol (*Dasylirion* spp.) or prickly pear (*Opuntia* spp.). The diversity of plant life forms that typify desert grasslands today is different from the vast monotypic "grasslands" one associates with the prairies of the Midwest—so much so, that Burgess (1995:58) proposes that a more appropriate name for Southwestern desert grasslands would be "Apacherian mixed shrub savanna" (historical conditions may have differed, however, as will be discussed below).

Herbaceous plants provide much of the ecological and botanical diversity in Southwestern grasslands; legumes and asters are particularly prominent members of many grassland communities. Members of the legume genus *Astragalus* are common, amongst others, and numerous members of the sunflower family may be present (such as *Aster*, *Antennaria*, *Wyethia*, *Chrysopsis* spp.). Other familiar forbs in grassland communities include Rocky mountain beeplant (*Cleome serrulata*), various species of flax (*Linum* spp.), and penstemons (*Penstemon* spp.). Cacti and succulents are particularly distinctive features of desert grasslands. In New Mexico and Arizona, nearly 150 species of cacti contribute to the plant diversity of the Southwestern region. The prickly pear (*Opuntia* spp.) has been proposed as a "keystone resource species" in Southwestern grasslands, a species that provides resources during bottlenecks of availability, due to the dependence of a variety of animals on prickly pears for both food and water during times of drought (West and Whitford 1995). Cacti also demonstrate the importance of microclimate variation in providing for increased species diversity. The rare grama grass cactus (*Toumeya papyracantha*), for example, depends upon the favorable microclimate provided by black grama grass to survive (Fletcher and Moir 1992, as cited in Moir and Bonham 1995).

The variable climate and topography of the Southwest contribute to the plant diversity in the grasslands of this region. The plant species composition and distribution of Southwestern grasslands depends heavily upon soil depth and texture, which in turn control the water retention potential of the soil (Burgess 1995). The conservation of desert soils is particularly critical, as soils in arid Southwestern grasslands tend to be shallow, and more than 30 percent of the available nitrogen and organic matter is in the top 10 cm of soil (Charley 1977). This top layer is also highly susceptible to erosion, and loss of these soils may lead to a decrease in floral diversity and associated faunal diversity (Noss and Cooperrider 1994). Soil characteristics are also greatly influenced by the activities of invertebrate animals. Soil invertebrates play a critical role in decomposition of organic matter, soil development, and alteration of the physical characteristics of the soil leading to increased water holding capacity, thereby influencing the associated plant community as well (Abbott 1989, Hutson 1989, Whitford and others 1995). The importance of subterranean termites, for example, has been demonstrated by the experimental elimination of this group on study plots, resulting in dramatic changes in both the species composition and productivity of grasses, forbs, and shrubs (Whitford 1991). Herbivorous vertebrates also have a strong influence on plant community composition, structure, and productivity, particularly fossorial rodents such as kangaroo rats and prairie dogs. The activities of these burrowing rodents aerate the soil, modify soil nutrient levels, and enhance moisture retention (Parmenter and Van Devender 1995). As moisture availability is probably the single greatest factor limiting plant diversity in the Southwest (Burgess 1995), many of the vertebrate and invertebrate animals of grasslands are making a critical ecological contribution to maintaining the overall biodiversity of these systems through their impacts on soil characteristics.

Beyond aerating the soil, cycling nutrients, and creating pockets of moisture retention through their subterranean activities, grassland rodents such as kangaroo rats have yet another significant impact on plant diversity. Herbivores generally promote plant diversity by suppressing more vigorous species that might otherwise exclude other members of the community, thereby allowing less competitive species to persist in the system (Burgess 1995). In desert grasslands, the abundance of annual grasses and forbs increases in proximity to kangaroo rat mounds (Andersen and Kay 1999). Furthermore, kangaroo rats have been demonstrated to actually control vegetative species composition and structure through selective seed predation and soil disturbance. A Chihuahuan desert shrubland was converted to grassland following exclusion of kangaroo rats from the site; both annual and perennial grasses increased in density up to three-fold in the absence of these animals (Brown and Heske 1990).

Plant diversity is also strongly affected by the presence or absence of mycorrhizal fungi. Mycorrhizal fungi have been proposed as "critical link species" in Western grasslands (West and Whitford 1995). Critical link species are species that play an important role in ecosystem function, but are not necessarily considered keystone species (Westman 1990). Approximately 90 percent of vascular plants are believed to depend upon a mutualistic association with mycorrhizae for enhanced phosphorous uptake; plant establishment or growth may be severely inhibited in the absence of the appropriate fungus. Studies of sagebrush steppe invaded by the exotic cheatgrass (*Bromus tectorum*) found that the repeated fires carried by the cheatgrass led to the widespread elimination of soil mycorrhizae, thereby inhibiting the reestablishment of perennial grasses and shrubs (Wicklow-Howard 1989). Granivorous rodents again play an important role by dispersing mycorrhizal spores throughout the grassland system through seed transport (Parmenter and others 1995).

Invertebrate Diversity

When most people think of animal biodiversity, it is the larger, more conspicuous animals characteristic of the terrestrial ecosystem that come to mind—birds, mammals, reptiles, or amphibians. In virtually all ecological systems, however, it is the invertebrate animals that not only account for the vast majority of species diversity and animal biomass, but also make the greatest contribution to key ecosystem processes such as nutrient cycling. Grasslands are no exception to this rule. Referring to shortgrass prairie systems, Arenz and Joern (1996:91) dub invertebrates as "the most significant contributor to the diversity of the prairie system." The contribution of invertebrates to biodiversity is hardly surprising, as insects alone compose approximately 90 percent of all terrestrial animal species, and fewer than 10 percent of these have even been identified and named (Gaston 1991). What invertebrates lack in size they make up for in sheer numbers; Lauenroth and Milchunas (1992) estimate the total biomass of arthropods on North American grasslands exceeds that of vertebrates if domestic livestock are excluded.

The invertebrate inhabitants of grasslands begin at the microscopic level in the soils with protozoans and nematodes. Protozoans are found in most any grassland with sufficient soil moisture. Although this requirement might seem to exclude them from the grasslands of the desert Southwest, this is hardly the case. Just a tiny amount of moisture trapped between soil particles is sufficient to sustain them. One study of the semiarid shortgrass prairie of Colorado found more than 20,000 protozoans in each gram of dry soil (Elliott and Coleman 1977). Soil protozoans feed on bacteria, yeasts, algae, and fungal mycelia (Curry 1994), and play an important role in the transformation of soil organic matter (Elliott and Coleman 1977). Nematodes are a highly diverse group of roundworms that also play a key ecological role in grasslands. Although their distribution and activity is also restricted to some degree by moisture availability, at temperate grassland sites a square meter of soil will yield an average of 9 million individuals of various nematode species (Sohlenius 1980). In terms of ecosystem function, nematodes are considered by some to be the most important consumers of energy (Scott and others 1979), and their habit of feeding on plant roots strongly affects net primary productivity in grassland systems (Smolik and Lewis 1982).

Arthropods form a more familiar component of the grassland invertebrate fauna and may include whip scorpions, crickets, grasshoppers, beetles, flies, bees, wasps, cicadas, centipedes, spiders, ants, and termites. Some grassland arthropods are less conspicuous due to their nocturnal or subterranean habits, a common strategy for coping with the heat stress of life in the desert (examples include sun spiders, scorpions, termites) (Whitford and others 1995). About 90 percent of grassland arthropods reside in either the soil or litter, including those that nest belowground or spend a significant portion of their life stage belowground (Arenz and Joern 1996). The presence of ephemeral wetlands in grassland systems provides for a more diverse array of arthropods, including mosquitoes, freshwater shrimp, and water fleas. If water is present long enough, dragonflies or damselflies may be added to the invertebrate fauna (Loring and others 1988).

The species diversity of most groups of grassland arthropods appears to be higher in the Southwestern United States than in the rest of the country (Danks 1994, Parmenter and others 1995), although data on arthropod diversity is relatively limited (Kosztarab and Schaefer 1990). Grasshoppers are one of the best known and highly diverse groups of grassland arthropods. The species diversity of grasshoppers in the Southwest is greater than all other Western States, with the exception of California (Parmenter and others 1995). For example, 30 species of the subfamily Gomphocerinae occur in the Southwest, as opposed to only five to 20 species in the rest of North America (Otte 1981, as cited in Parmenter and others 1995). Darkling beetles (Tenebrionidae) also reach the peak of species diversity in Western arid lands and are one of the major detritivores in Southwestern ecosystems (Crawford 1990). Other species rich arthropod groups in the Southwest include soil-dwelling mites and collembolans (Crawford 1990, Zak and Freckman 1991).

Termites (Isoptera) are considered keystone species in Southwestern grasslands (West and Whitford 1995). A keystone species is one whose ecological impact on the community is disproportionately large relative to its abundance (Power and others 1996). Although termites contribute relatively little to invertebrate diversity in the Southwestern grasslands (there are only about 12 species), they are widely considered to be one of the most important invertebrates in these grassland systems in terms of their contribution to ecosystem function. The biomass of subterranean termites in desert grasslands is estimated to exceed that of domestic livestock (Whitford and others 1995). The presence of termites has a strong influence on the abundance and species composition of soil microfauna, and they are important consumers of dead plant material and dung. This latter point is not to be taken lightly; dung decomposes so slowly in desert systems that without termites to decompose the dung from livestock and incorporate it into the soil, grazed desert grasslands would eventually be covered with dry dung, leading to reduced plant productivity and leading to an overall decrease in the carrying capacity of the grassland system (Whitford and others 1995). Termites also consume approximately 50 percent of all photosynthetically fixed carbon in desert grassland systems (Whitford and others 1995). Termites therefore play a critical role in carbon and nutrient cycling in desert soils, and their subterranean activities also have a strong effect on soil aeration and water infiltration, increasing the water storage capacity of the soil. The cumulative ecological impact of subterranean termites in grassland systems led Whitford and others (1995:181) to declare this group the "most abundant and functionally most important arthropods in desert grasslands."

Ants (Formicidae) are one of the more familiar arthropods of Southwestern grasslands. Ants play an important role in maintaining the plant diversity of desert grassland systems by preferentially harvesting the seeds of dominant plant species (Whitford and others 1995). The subterranean nests of ants are important in grassland systems for concentrating nutrients and allowing for increased water infiltration; these properties lead to high density vegetation surrounding ant nests in the desert grassland environment (Whitford and others 1995). Ant nests may persist in grassland habitats for up to 80 years (Whitford and others 1995).

The local species richness of invertebrates in grasslands is largely dependent upon the species composition, productivity, and habitat structure of the plant community (Arenz and Joern 1996, Lawton 1983, Strong and others 1984). Herbivorous arthropods tend to dominate the invertebrate community in grasslands (French 1979). Each plant species tends to have its own specialized set of invertebrate herbivores, which in turn support an array of invertebrate predators and parasites, and so on. Herbivores make up approximately 85 percent of the arthropod biomass in shortgrass steppe (Lauenroth and Milchunas 1992), and spiders are significant secondary consumers in these systems (Schmidt and Kucera 1975). Even though most of the invertebrates in grasslands are herbivorous, invertebrates are estimated to consume less than 10 percent of the live biomass in grassland systems (Chew 1974). As many arthropods are associated with the early and midstages of vegetational succession (Usher and Jefferson 1991), the diversity of invertebrates is often greatest in areas with a diverse mixture of plant species and physiognomies, in conjunction with natural disturbances (Samways 1994).

Despite their largely inconspicuous nature, invertebrates form a critical component of Southwestern grasslands through their contribution to decomposition and nutrient cycling, increasing soil porosity and water infiltration, regulating the growth of soil bacteria and fungi, and controlling the availability of mineral nutrients for plants (Whitford and others 1995).

Vertebrate Diversity

The American Southwest has been called "one of the most biologically diverse regions in the United States" when it comes to vertebrate animals (Parmenter and Van Devender 1995:196). Many groups of animals reach their highest levels of species richness in the country along the border with Mexico in southern Arizona, New Mexico, and Western Texas. The species richness of mammals in the Southwest is rivaled only by that of central California; bird species richness reaches its peak in the Southwest, southern Texas, and California, and the numbers of species of reptiles are higher only in eastern Texas (Parmenter and others 1995 and references therein). The only group of vertebrates that are not richly represented in the arid Southwest is, not surprisingly, aquatic organisms; the species richness of amphibians and fishes reaches its peak in the Southeastern States (Parmenter and others 1995).

Once again it is environmental heterogeneity created through a combination of elevational variability, climate dynamics, and the natural mosaic pattern of desert grassland habitats, in conjunction with the biogeographic history of the region, that accounts for this faunal richness. The greatest diversity of animals is usually within the more temperate conditions found at intermediate elevations. Animals at high elevations tend to be limited by cold temperatures, while those at low elevations are limited by aridity (Parmenter and others 1995). Although how the merging of formerly isolated faunal regions has contributed to the present diversity of Southwestern animal species was mentioned briefly above, a more concrete example of this blending offered by Parmenter and others (1995) may help to clarify how this process has acted to enhance vertebrate diversity in the Southwest (table 4-1).

The vertebrate diversity of grasslands is enhanced by the collective representation of taxa from many convergent habitats. Not only are grassland specialists represented, but numerous other animals that may

Table 4-1. An example of how the rich assemblage of terrestrial vertebrate species present today in Southwestern grasslands is in part the result of biogeographic history. Formerly distinct faunas derived from several major geographic regions now coexist in this area due to range expansions and the removal of physical barriers following glacial retreats at the end of the Pleistocene. A few of these species are listed below (adapted from Parmenter and others 1995).

Biogeographic region	Vertebrate species contributed to Southwestern grasslands
Great Plains	Western box turtle
	Great Plains skink
	Black-tailed prairie dog
	Northern grasshopper mouse
	Swainson's hawk
	Lark sparrow
Sierra Madre (Mexico)	Yarrow's spiny lizard
	Rock rattlesnake
	Pygmy mouse
	Montezuma quail
Chihuahuan Desert	Texas horned lizard
	Trans-Pecos rat snake
	Silky pocket mouse
	Banner-tailed kangaroo rat
	Scaled quail
	Cassin's sparrow
Sonoran Desert	Collared lizard
	Sidewinder
	Desert kangaroo rat
	Southern grasshopper mouse
	Gila woodpecker
	Bendire's thrasher
Mojave/Great Basin Desert	Short-horned lizard
	Chisel-toothed kangaroo rat
	Sagebrush vole
	Sage thrasher
	Sage sparrow

dwell primarily in the surrounding mosaic of desert scrub, pinyon juniper, or riparian areas, and that use the grasslands for foraging, may add to the species richness; this is particularly true of more mobile, generalized species of mammals or birds (Parmenter and others 1995). Up to 18 species of bats, for example, may be found in Southwestern grasslands. While six of these species are commonly found in grasslands, most of them will utilize grasslands only if their other habitat requirements may be met within a reasonable distance, specifically the presence of appropriate roost sites and water (Chung-MacCoubrey 1996). The interspersion of Southwestern grasslands with habitats such as pinyon-juniper woodlands meets these requirements.

Reptiles and Amphibians

Herptiles—reptiles and amphibians—are important components of the grassland vertebrate community. In the Southwest, reptiles in particular make a significant contribution to overall diversity. Approximately 44 species of reptiles are associated with desert grasslands, with fewer (18 species) present in high elevation mountain meadows (Parmenter and Van Devender 1995). Although the arid Southwest is a challenging environment for many amphibians, certain toads are relatively common in desert grasslands (such as the Western spadefoot toad [*Scaphiopus hammondi*]), and true frogs (such as the Chiricahua leopard frog [*Rana chiricahuensis*]) and tiger salamanders (*Ambystoma tigrinum*) may be found near permanent water sources. Common grassland/desert specialists include various species of box turtles, spadefoot toads, earless lizards, whiptails, horned lizards, bullsnakes, and rattlesnakes. Many species associated with grassland habitats require specific habitat features, such as rock outcroppings to serve as dens for wintering snakes (Collins 1982, Hammerson 1986). Many reptiles and amphibians join a variety of other animals in taking advantage of the beneficial microclimate provided by prairie dog burrows. At least 12 species of amphibians and 17 species of reptiles have been reported as regularly associated with black-tailed prairie dog towns (Reading and others 1989, Sharps and Uresk 1990). Spadefoot toads (family Pelobatidae) require playas, temporary pools that fill with water during the summer monsoons, for breeding (Corn and Peterson 1996). Most Southwestern amphibians have short aquatic larval stages to take advantage of the ephemeral water sources for breeding. In fact, permanent water sources may be detrimental to amphibian populations by attracting predators such as raccoons (*Procyon lotor*), or by encouraging the establishment of exotic species such as bullfrogs (*Rana catesbiana)* and centrarchid fishes. These predators have been implicated in the disappearance of several native species of amphibians in various areas of the country (Collins and others 1989, Hayes and Jennings 1986).

Both reptiles and amphibians play functional roles in food webs both as predators on invertebrates and small vertebrates and as prey for larger animals (although some herps, such as the desert tortoise, are herbivores). Amphibians in particular are an important avenue for nutrient transport between aquatic and terrestrial systems. Amphibian and reptile populations are also particularly sensitive indicators of environmental stresses and may thus serve as a warning signal of problems such as pesticide contamination (Beiswenger 1986, Blaustein 1994). Like other grassland animals, herps are sensitive to changes in habitat composition and structure, and herp species richness is generally greatest in relatively heterogeneous habitats. In dense grasslands, for example, moderate grazing that increases the patchiness of the grass density and adds some variety to its structure enhances the habitat for a variety of snakes, lizards, and toads.

One of the more fascinating groups of grassland reptiles is the genus *Cnemidophorus*, the whiptail lizards. There are 10 unisexual species of whiptails in the Southwest that are made up entirely of female individuals. These lizards reproduce by an asexual autofertilization method known as parthenogenesis. All-female whiptail lines are polyploid (have more than one set of chromosomes), indicating that they were originally formed through the hybridization of two sexual species. Each individual in the species is genetically identical to the original hybrid (Cole 1984, Parmenter and Van Devender 1995). The desert grasslands of the Southwestern United States are the evolutionary center for this highly unusual group of vertebrates; seven of the 10 unisexual species are restricted almost entirely to this habitat. The desert grasslands of Texas and New Mexico are home to the New Mexican (*Cnemidophorus neomexicanus)*, checkered (*C. tesselatus*), and Chihuahuan (*C. exanguis*) whiptails; desert grassland (*C. uniparens*), Sonoran (*C. sonorae*), and Gila spotted (*C. flagellicaudus*) whiptails are found in Arizona; and the plateau whiptail (*C. velox*) is found in the grasslands of the Great Basin. Furthermore, several whiptail species (New Mexican whiptail, desert grassland whiptail, and the little striped whiptail *C. inornatus*—a nonparthenogenetic species) are believed to be dependent on native stands of grasses within these habitats.

Birds

Although birds are some of the most abundant vertebrates found in Southwestern grasslands (Parmenter and Van Devender 1995), the bird community tends to be the most simplistic in terms of species richness (Knopf 1996). As one example, Knopf (1996) cites his results from a series of 112 point count surveys in the Pawnee National Grasslands in Colorado, in which just three species (horned lark [*Eremophila alpestris*], McCown's longspur [*Calcarius mccownii*], and lark bunting [*Calamospiza melanocorys*]) accounted for 87 percent of all individuals recorded; only 14 species of native birds were recorded in total. Bird species commonly found in the grasslands of the Southwest include the horned lark, lark bunting, meadowlarks (both Eastern [*Sturnella magna*] and Western [*Sturnella neglecta*]), scaled quail (*Callipepla squamata*), mountain plover (*Charadrius montanus*), burrowing owl (*Athene cunicularia*), short-eared owl (*Asio flammeus*), prairie falcon (*Falco mexicanus*), ferruginous hawk (*Buteo regalis*), and various sparrows (such as vesper [*Pooecetes gramineus*], lark [*Chondestes grammacus*], Cassin's [*Aimophila cassinii*], and Botteri's [*A. botterii*]) (Knopf 1996, Parmenter and Van Devender 1995). Even some shorebirds, such as long-billed curlews (*Numenius americanus*), utilize Southwestern grasslands for breeding. At high elevations, rosy finches (*Leucosticte* spp.) and white-tailed ptarmigan (*Lagopus leucurus*) may be found in alpine meadows of the Southwest (Parmenter and others 1995).

Many of the typical avian residents of Southwestern grasslands nest on the ground, as the lack of vertical structure in grasslands offers little other choice. The burrowing owl, a common denizen of Southwest grasslands, nests below the ground in abandoned prairie dog burrows. Most members of the grassland bird community are granivores, and the dynamics of this community are closely tied to levels of seed production. As most desert grasses set seed in late summer or early fall following the monsoons (McClaran 1995), the peak in resident grassland bird numbers usually occurs in late summer in coincidence with maximum seed production (Maurer 1985). Avian species richness increases during the winter months, when the grasslands of the Southwest support a great concentration of migratory species such as Baird's sparrow (*Ammodramus bairdii*), white-crowned sparrow (*Zonotrichia leucophrys*), grasshopper sparrow (*Ammodramus savannarum*), sage sparrow (*Amphispiza belli*), Sprague's pipit (*Anthus spragueii)*, McCown's longspur, and chestnut-collared longspur (*Calcarius ornatus*).

Birds respond strongly to changes in habitat architecture, and the diversity of the grassland bird community will increase in the presence of shrubs, trees, cacti, or even human structures (Grinnell 1922, Knopf and Scott 1990, Parmenter and others 1995, Szaro 1981). The addition of vertical structure provides a far greater range of avian habitats and thereby adds a whole new component to the bird community. The maximum species richness is probably found along riparian corridors or near permanent wetlands in grasslands, where large numbers of migrants and transients concentrate (Parmenter and Van Devender 1995); here, various species of warblers, vireos, and other decidedly nongrassland species may be found. It is important to recognize, however, that even if species diversity is technically enhanced by the addition of vertical structure such as woody plants or human developments, grassland specialists are usually lost in the process. The enhanced bird diversity witnessed in such cases is most likely provided by an increase in relatively common generalist species and might mask any concomitant population declines or extirpations of narrow endemics that may occur (Knopf 1992).

In pure grasslands without significant vertical structure provided by shrubs or trees, different bird species demonstrate preferences for an array of grass heights and various patterns of patchiness. Mountain plovers and McCown's longspurs, for example, occur in short grasslands, often those that have been subjected to "heavy grazing pressure to the point of excessive surface disturbance" (Knopf 1996:141 and references therein). Lark buntings will use areas of shortgrass prairie but require that tufts of taller grasses be interspersed in the landscape to provide nest concealment (Finch and others 1987). Baird's sparrow can be found across a wide range of grassland types and grazing intensities (Kantrud 1981), and Cassin's sparrow requires grasslands that provide at least 6 percent shrub cover and may be lightly grazed (Bock and Webb 1984).

Grassland birds are a source of great conservation concern, as this group of birds has shown consistently steep population declines over the past few decades, on the order of 25 to 65 percent—more than any other guild of North American bird species (Askins 1993, Knopf 1992, 1996). Formerly widespread and common species such as the lark bunting and Cassin's sparrow are showing statistically significant declines, and the mountain plover has been proposed for Federal listing as a threatened species. Although some theorize that declines in populations of neotropical migratory birds are due to loss of wintering habitat in the tropics (see Briggs and Criswell 1979, Lovejoy 1983, Terborgh 1980), most of the grassland birds in question are short-distance migrants that spend their winters in the grasslands of the Southwest and Mexico, suggesting that alteration of Southwestern grassland habitats may be contributing to the decreases witnessed (DeSante and George 1994).

Mammals

The Southwestern grasslands owe much of their high vertebrate diversity to mammals, and more particularly to rodents. In a comparison of mammals found in six habitat types in a single region of New Mexico, desert grasslands had the greatest species richness of any major ecosystem type with 56 species, ahead of desert scrub, pinyon-juniper woodland, montane forest, montane meadow, and riparian zone (Parmenter and Van Devender 1995). The high species richness of the desert grassland is primarily due to the diversity of rodents in this system, especially ground squirrels (Sciuridae), kangaroo rats (*Dipodomys* spp.), and mice (Muridae). Rodents tend to be the dominant mammals in all desert grasslands, and are well represented by grassland specialists, including the bannertail (*D. spectabilis*) and Ord (*D. ordii*) kangaroo rats, black-tailed prairie dog (*Cynomys ludovicianus*), and spotted ground squirrel (*Spermophilus spilosoma*) (Parmenter and Van Devender 1995). As one indication of just how diverse the rodent community of the Southwestern grasslands is, Parmenter and others (1995) point out that just one 20 ha area of Chihuahuan Desert has the same number of native rodent species as the entire States of Michigan and Pennsylvania combined, and that's allowing those States two introduced species and two semiaquatic species.

Many other mammal species are also characteristic of Southwestern grasslands, including, amongst others, pronghorn (*Antilocapra americana*), white-sided jackrabbits (*Lepus callotis*), swift fox (*Vulpes velox*), badgers (*Taxidea taxus*), coyotes (*Canis latrans*), mule deer (*Odocoileus hemionus*), several species of bats, and—in high elevation grasslands—pikas (*Ochotona princeps*). Historically, bison (*Bison bison*) were found in the shortgrass prairie regions of eastern New Mexico, but probably did not occur regularly in the arid grasslands farther west (Berger and Cunningham 1994, Kay 1994, Mack and Thompson 1982). Many of these mammals have a strong impact on the overall diversity of the grassland system through various types of disturbance. Vegetation structure and species composition, for example, are affected by selective feeding of herbivores and by soil disturbance. By selectively grazing on dominant species, herbivorous mammals allow subdominant plant species to compete and persist in the community (Risser and others 1981). Digging by badgers, prairie dogs, kangaroo rats, and gophers creates soil disturbances that allow for the establishment of annual forbs and grasses, and also increases the porosity and water-holding capacity of the soil (Benedict and others 1996 and references therein). Wallowing by bison in areas where they formerly occurred, and small scrapes created by pronghorn, serve a similar function (Benedict and others 1996, Parmenter and Van Devender 1995). These small-scale natural disturbances add unique microhabitats available for colonization by other species, increasing vegetative diversity, enhancing the mosaic nature of the habitat, and leading to increased faunal diversity as well (both invertebrate and vertebrate) (Benedict and others 1996, Collins and Barber 1985). Overall, Collins and Barber (1985) found that diversity in a mixed-grass system was enhanced by moderate levels of natural disturbance (*sensu* the "intermediate disturbance hypothesis" of Connell 1978) and that small-scale disturbances have an additive effect that further enhances diversity.

Grassland biodiversity is also strongly impacted by the presence of keystone species. The power that kangaroo rats exert over the structure and dynamics of their habitat has led to their designation as a keystone species in the grassland systems of the Southwest (Brown and Heske 1990, West and Whitford 1995). Kangaroo rats exert their influence largely through selective seed predation and soil disturbance. In a long-term study on an Arizona desert shrubland, Brown and Heske (1990) demonstrated that the removal of kangaroo rats resulted in dramatic increases in grass densities, as well as a shift toward large-seeded winter annual plant species. Small-seeded winter annuals decreased, herbaceous vegetation increased (including both grass and forbs), litter accumulation increased, seed-eating birds decreased, and several new species of rodents colonized the plots where kangaroo rats were absent. The other native rodents on the plots where kangaroo rats were removed were not able to prevent the conversion of the habitat from shrubland to grassland, along with the associated changes in the resident fauna, thus supporting the keystone role of the kangaroo rats in this system. Furthermore, kangaroo rat burrows provide favorable microclimates for a diverse array of both invertebrate and vertebrate animals. Western box turtles, Great Plains skinks, and massasaugas use kangaroo rat mounds for shelter, and several species of roaches, crickets, and beetles are found almost exclusively in these mounds (Hawkins and Nicoletto 1992).

The black-tailed prairie dog is a critically important keystone species in Southwestern grassland systems whose presence greatly enhances local biodiversity (Kotliar and others 1999, Miller and others 1994, Whicker and Detling 1988). The burrowing and feeding behaviors of prairie dogs have drastic effects on the structure, species composition, and nutritive value of surrounding vegetation, create open areas to add to the heterogeneity of the habitat, modify the physical characteristics of soils, affect energy and nutrient cycles, and provide valuable microclimates utilized as shelters by a multitude of both invertebrate and vertebrate animals (Benedict and others 1996, Miller and others 1990, 1994, Whicker and Detling 1988 and references therein). The activities of prairie dogs, feeding on and clipping vegetation in the area of their colony, stimulates fresh plant growth and enhances the nutritional content of the vegetation, leading herbivores such as pronghorn to preferentially feed on prairie dog towns (Coppock and others 1983). Prairie dog towns are considered to be centers of animal diversity due to the great numbers of species that converge on the colonies to either use the old burrows, forage on the surrounding vegetation, or feed on the prairie dogs themselves (Miller and others 1990, 1994). Nearly 170 species of vertebrates have been reported using prairie dog towns, although this number is undoubtedly excessive due to the inclusion of birds flying over (Benedict and others 1996). A more critical recent review suggests that there is sufficient evidence for the strong dependence of nine vertebrates upon prairie dog colonies, 20 species appear to use the colonies opportunistically, and 117 species may have some relationship with the colonies, but data to support any solid conclusions are lacking (Kotliar and others 1999). However, of those animals that are closely associated with prairie dogs, several are of great conservation concern, including the black-footed ferret (*Mustela nigripes*), swift fox, ferruginous hawk, burrowing owl, and mountain plover (Kotliar and others 1999, Samson and Knopf 1994). This association does not bode well as eradication programs have resulted in eliminating the black-tailed prairie dog from 98 percent of its former range, reducing its numbers to the point that the species is now under consideration to be listed as endangered (Miller and others 1994, U.S. Fish and Wildlife Service 2001).

Threats to Grassland Biodiversity

The native biodiversity of Southwestern grasslands has been greatly altered through human activities. Numerous animal species have been extirpated or greatly reduced through direct persecution, including the black-tailed prairie dog, Mexican wolf, bison, and grizzly bear (Benedict and others 1996; table 4-2). Others have been reduced presumably due to their dependence on a keystone species that has been removed from the system. For example, the black-footed ferret (endangered) and mountain plover (proposed for listing) are strongly dependent upon prairie dog colonies for survival (Kotliar and others 1999), and the

Table 4-2. Terrestrial vertebrates of Southwestern grasslands that are now extinct or have been extirpated from the region. The following species are not necessarily restricted to grassland habitats; although many are grassland specialists, this list also includes those species that rely heavily on grasslands in some parts of their range or as one component of a mosaic of habitats utilized. Sources for the information presented here include Arizona Game and Fish Department (1988), Association for Biodiversity Information (2001), and New Mexico Department of Game and Fish (2000).

Common name	Scientific name	Federal status	Extirpated or extinct	Primary cause of extinction or extirpation	Notes
Western boreal toad	*Bufo boreas boreas*	CW	Extirpated	Unknown	Believed extirpated; formerly occurred in alpine meadows
New Mexico sharp-tailed grouse	*Tympanuchus phasianellus hueyi*		Extinct	Habitat loss or degradation due to overgrazing, agriculture, succession	
Sage grouse	*Centrocercus urophasianus*		Extirpated from NM & AZ	Overhunting, habitat loss or degradation from overgrazing	
New Mexican banner-tailed kangaroo rat	*Dipodomys spectabilis baileyi*		Extirpated from AZ	Habitat degradation from overgrazing	Inhabited Great Basin desertscrub
Black-tailed prairie dog	*Cynomys ludovicianus*	CW	Extirpated from AZ	Direct human persecution	Some small populations persist in NM
Black-footed ferret	*Mustela nigripes*	E	Extirpated	Elimination of prairie dogs (primary prey)	Currently attempting reintroduction in NM
Bison	*Bison bison*		Extirpated from NM and AZ	Overhunting	Now exist on private ranches
Merriam's elk	*Cervus elaphus merriami*		Extinct	Overhunting	Native AZ elk
Mexican wolf	*Canis lupus baileyi*	E	Extirpated	Direct human persecution	Experimental populations reintroduced in NM and AZ
Intermountain wolf	*Canis lupus youngi*		Extinct	Direct human persecution	
Grizzly bear	*Ursus arctos*	T	Extirpated from NM and AZ	Direct human persecution	Persist in Northwestern States (for example, Montana), Canada, and Alaska

E = endangered, T = threatened, CW = candidate with "warranted but precluded" determination declines in their populations have been linked to the extirpation of prairie dogs (Miller and others 1994). Not only does the disappearance of the prairie dog have dire consequences for the species dependent upon it, but the prairie dogs themselves are now threatened with deleterious genetic consequences as a result of the fragmentation and isolation of their remaining populations (Pizzimenti 1981).

A far more subtle factor has been responsible for most extinctions at the local level in Southwestern grasslands. Changes in the structure and function of grassland habitats have probably been responsible for more losses of native diversity than any other cause (Stacey 1995). "While losses of biological diversity at the local level are often the least noticed," Stacey (1995:34) points out, "they are extremely important because they change the functional dynamics of the local community and because if local extinctions continue long enough the species will be lost over wide areas and may not recover without human intervention." Changes in grassland habitat structure and function may come about in many ways, but some of the most important sources of these changes in Southwestern grasslands have been the loss of fire as a natural cyclical event, the elimination of prairie dog colonies, heavy grazing by livestock, the introduction of nonnative grasses, and shrub encroachment (Parmenter and Van Devender 1995, Risser 1988). The division of formerly expansive rural landscapes into increasingly fragmented "ranchettes" is the latest recognized threat to grassland biodiversity (Brown and McDonald 1995, Maestas and others 2002), and will be discussed separately under the section on habitat fragmentation.

Loss of Natural Fire Cycles

Fire plays a key role in the maintenance of most grassland systems. Without periodic fires, woody plants begin to encroach into grasslands, converting them to shrublands or woodlands. The grasslands of the Southwest are no exception. Many researchers agree that historically fires were both common and extensive in the desert grasslands, and that these fires were instrumental in maintaining the integrity of these systems (Bahre 1991, Humphrey 1958, McPherson 1995, McPherson and Weltzin 2000).

The exception to this rule may be grasslands dominated by black grama. The extreme difficulty black grama exhibits in recovering from a burn indicates that this species is not fire-adapted and probably did not evolve under a history of frequent burning (Buffington and Herbel 1965, Dick-Peddie 1993). More recently, however, it has been proposed that the negative effects witnessed may have been attributable to a coincident period of drought rather than to fire (Curtin and others 2002). Precipitation has a considerable impact on grassland productivity following fire, both in terms of timing and quantity.

The natural frequency and extent of grassland fires in the Southwest are believed to have declined dramatically since Euro-American settlement of the region in the late 1800s (Bahre 1991, 1995, Humphrey 1958). A review of the role of fire in desert grasslands reveals that the natural frequency of fire in these systems was probably on the order of every 7 to 10 years (McPherson 1995 and references therein). Fires occurring on this cycle are believed to be sufficient to prevent the establishment of woody plants, by killing seeds on the surface and preventing woody plants from reaching the age where resprouting is possible (McPherson 1995). Although fires eliminate grass cover in the short term, in the long term, grasses are rejuvenated by the occurrence of fire and benefit from the elimination of woody plants. The timing of fires is also important. Fire in the early summer, when the growth of many perennials is just beginning, can negatively impact warm season grasses, whereas these same grasses are tolerant of fire during the dormant season (McPherson 1995). The level of soil moisture at the time of ignition is also a consideration; for some plant species, burning on dry soils may be damaging (W. Moir, personal communication 2003).

Although many factors contribute to fire regimes, perhaps the most important change that has resulted in decreased fire frequency and intensity in the Southwest is the lack of fine fuels to carry the fires (Humphrey 1958, McPherson 1995). Historically, the timing of this change corresponded with the widespread increase in livestock grazing in the Southwest after 1880. At this time, stocking rates reached record levels, and overgrazing was actually encouraged to reduce the fire hazard and encourage the growth of trees (Bahre 1991, Leopold 1924 as cited in McPherson 1995). In addition, the use of wooden posts for livestock fencing provided the incentive for quickly suppressing rangeland wildfires that would compromise the integrity of the fences (Sayre 2002). Today, fragmentation from roads and suburban developments serve as a kind of artificial firebreak to contain the spread of extensive wildfires (Bahre 1995, McPherson 1995). The continuing growth of residences on formerly undeveloped lands has also led to a demand for active fire suppression in these areas (Hansen and others 2002). These changes in the frequency and intensity of natural fire regimes have doubtless contributed to the widespread conversion of Southwestern grasslands to shrublands (Archer 1989, Brown 1982, Humphrey 1958), thereby radically altering the nature of the habitat for native grassland species.

Prairie Dog Eradication

In the Southwestern grasslands, the prairie dog is considered a "keystone" species—a species that has a large overall effect on a community or ecosystem disproportionate to its abundance (Kotliar and others 1999, Power and others 1996). The activities of these burrowing animals have a dramatic impact on the patch dynamics and ecosystem function of the grasslands that they inhabit. Prairie dog disturbances impact the physical and chemical properties of the soil, alter vegetational structure, affect plant species composition, and improve the nutrient value of plants growing in the vicinity of their colonies (O'Meilia and others 1982 and references therein, Whicker and Detling 1988). The increased nutritional value of forage on colonies may act to offset any decrease in biomass as a result of clipping by prairie dogs (Holland and Detling 1990, O'Meilia and others 1982 and references therein). Bison, elk, pronghorn, and livestock all preferentially graze on prairie dog colonies, presumably because of the increased value and palatability of the herbage there (Coppock and others 1983, Knowles 1986, Krueger 1986, Wydeven and Dahlgren 1985).

Active prairie dog towns contribute to increased biological diversity by supporting a different complement of species compared to areas unoccupied by prairie dogs (Agnew and others 1986, Mellink and Madrigal 1993, O'Meilia and others 1982). Furthermore, several vertebrate species are considered highly dependent upon prairie dogs either as prey or for the habitat provided by their colonies, including the endangered black-footed ferret. Other animals considered true prairie dog associates are the mountain plover, burrowing owl, ferruginous hawk, golden eagle (*Aquila chrysateos*), horned lark, swift fox, deer mouse (*Peromyscus maniculatus*), and northern grasshopper mouse (*Onychomys leucogaster*) (Kotliar and others 1999).

Once a dominant force in the grasslands of the Western United States, the ecological impact of the prairie dog on these systems has nearly been extinguished. Up until the early 1900s, prairie dog colonies were estimated to cover hundreds of millions of acres of shortgrass prairie and desert grasslands west of the Great Plains (Anderson and others 1986). Today prairie dogs are estimated to persist on a mere 2 percent of their former range (Anderson and others 1986, Miller and others 1994). One species, the Utah prairie dog (*Cynomys parvidens*), is endangered, and the black-tailed prairie dog, formerly the most abundant and widespread of the five species of prairie dogs in North America, is a candidate for listing (USFWS 2000). The population numbers of the black-tailed prairie dog are estimated to have been reduced by 98 percent, and the species may occupy as little as 0.5 percent of its original range (Mac and others 1998 as cited in USFWS 2000). Although the conversion of native prairie habitat to other land uses may have contributed to some degree, undoubtedly the greatest single factor in the loss of prairie dogs has been the concerted effort by both Federal and State government agencies to exterminate these animals for the benefit of the livestock industry (Mulhern and Knowles 1996, Parmenter and Van Devender 1995).

The campaign to eradicate prairie dogs from Western grasslands began in earnest following the release of a Department of Agriculture report suggesting that the presence of prairie dogs may reduce range productivity by 50 to 75 percent (Merriam 1902). The U.S. Biological Survey responded with a massive poisoning campaign under the auspices of its Predator and Rodent Control program. Aiming to reduce competition with livestock, millions of acres of prairie dog colonies were poisoned, and shooting of prairie dogs was encouraged across their range (Bell 1921, Mulhern and Knowles 1996, Parmenter and Van Devender 1995, Van Pelt 1999). Fear of sylvatic plague buoyed these efforts after the bacterium was discovered in black-tailed prairie dogs in Texas in the 1940s (Cully 1989, Mulhern and Knowles 1996). In some States, annual extermination of prairie dogs on State and privately owned lands was a legal requirement. Nebraska, for example, only recently repealed this mandate in 1995 (Van Pelt 1999).

The black-tailed prairie dog and the Gunnison's prairie dog (*C. gunnisoni*) are the two species that inhabit the grasslands of Arizona and New Mexico. Described as occurring in "immense colonies" in Arizona in 1885 (Mearns 1907), the Arizona prairie dog (*C.l. arizonensis*), a subspecies of the black-tailed prairie dog, was largely extirpated from that State by 1938; one small single colony survived until 1960 (Van Pelt 1999). In New Mexico, the range of the black-tailed prairie dog has been reduced by at least 25 percent (Hubbard and Schmitt 1984). In the Animas Valley, for example, biologists from the Museum of Southwestern Biology did not observe one single prairie dog between the years 1955 and 1972 (Findley 1987). Yet in 1908 Vernon Bailey had described this same area as an almost continuous prairie dog town for its length and breadth, estimating that over 6 million prairie dogs inhabited the valley (Bailey 1932).

Notwithstanding the drastic declines already witnessed in prairie dog numbers and the evidence of a cascade effect on other species, prairie dogs today are still widely considered to be vermin and enjoy little in the way of legal safeguards from any states (Van Pelt 1999). This is in spite of more recent evidence that the level of competition between prairie dogs and livestock is more likely on the order of 4 to 7 percent (Uresk and Paulson 1988 as cited in Miller and others 1994) and that there is no significant difference in the market weight of steers whether they graze in conjunction with prairie dogs or not (O'Meilia and others 1982;

although it should be noted that the statistically insignificant weight difference did result in an economic loss). Following their comprehensive review, Kotliar and others (1999:186) concluded that prairie dogs are "crucial to the structure and function of native prairie systems." Not only are scientists today stressing the importance of preserving remaining prairie dog colonies to maintain biodiversity (for example, Miller and others 1994), some are going further and calling for the reintroduction of prairie dogs to restore ecosystem function (for example, Manzano-Fischer and others 1999).

Overgrazing by Livestock

Livestock grazing is the predominant land use in the Western States. More than 70 percent of the land area in the West (11 states, from Montana, Wyoming, Colorado, and New Mexico westward) is grazed by livestock, predominantly cattle, including wilderness areas, wildlife refuges, National Forests, and some National Parks (Fleischner 1994 and references therein). Some argue that overly heavy levels of livestock grazing are one of the greatest sources of habitat degradation in the West (for example, Noss and Cooperrider 1994), leading to widespread declines in the native wildlife of North American grasslands (Fleischner 1994). Others point out that most studies of grazing effects have suffered from poor experimental design (for example, Brown and McDonald 1995, Jones 2000), or have found the impact of grazing to be relatively negligible on factors such as native species richness (Stohlgren and others 1999). The issue of livestock grazing in the West is highly contentious. In all cases, it is important to remember that the impact of grazing will vary greatly depending upon any number of variables, including the season of use, stocking rate, environmental conditions, and the evolutionary history of grazing in the region (Fleischner 1994, Jones 2000, Milchunas and Lauenroth 1993), and may also differ according to geographic scale (Stohlgren and others 1999).

This discussion will focus primarily on how poorly managed grazing of livestock may impact grassland habitats in the Southwest.

Grazing by livestock has the potential to alter grassland habitats in many ways. Depending on the intensity and length of the grazing regime and environmental conditions, livestock activities may significantly alter plant species composition, extent of vegetative cover, and physical structure of the habitat (Bock and others 1984). As discussed above, any changes in these vegetative parameters exert a strong influence on the associated fauna, so that changes in plant diversity and structure result in changes in animal diversity. Most frequently, overgrazed sites result in a loss of specialized native fauna and may or may not exhibit an increase in more widespread, generalist species (Bock and others 1984, Bock and Webb 1984, Jones 1981). In an Arizona grassland, for example, heavily grazed pastures had an abundance of birds that are commonly found in disturbed areas, such as horned larks and scaled quail, while grassland specialists such as Cassin's, Botteri's, and grasshopper sparrows were the dominant species in ungrazed plots (Bock and Bock 1988). In an extensive review of the grazing literature, Jones (2000) found that a majority of the studies for which there were sufficient data reported a decrease in both rodent species richness and diversity in response to grazing.

Livestock grazing can have more indirect effects on the environment as well. Soil disturbances created by trampling and digging produce microsites ripe for the invasion of weedy plant species, and cattle tend to import propagules of nonnative plant species on their coats or through their feces (Hobbs and Huenneke 1992). The combination of selective grazing by livestock on more palatable species and the opportunities for invasion by exotic species through soil disturbance and increased nutrient input from dung results in the decline of native perennial grasses and an increase in nonnative annuals (Mack 1981, 1989, Moore 1970). Livestock grazing may also contribute to shrub encroachment by eliminating the grasses and reducing competition for the seedlings of woody plants (Humphrey 1958). Grazing had largely negative impacts on numerous soil and vegetation variables examined in the review by Jones (2000), including increased soil loss to erosion, decreased infiltration rates, and decreased litter cover. Although the results of her review suggest that grazing has an overall negative impact on arid ecosystems in North America, Jones also points out that it was not possible to control for important factors such as stocking rates, grazing intensity, or timing in her comparison.

Some would argue that moderate levels of grazing may benefit Southwestern grasslands because maximum biodiversity is achieved under intermediate levels of disturbance (Connell 1978). In a test of the intermediate disturbance hypothesis, Collins and Barber (1985) found that grassland vegetation diversity was high on light to moderately grazed mixed-grass prairie (as opposed to undisturbed or most severely disturbed treatments). They concluded that diversity in such systems may be increased by moderate levels of natural disturbance, and furthermore that such disturbances have additive effects that further increase diversity. Today, some practitioners promote the use of properly controlled livestock grazing as a key component of sustainable ecosystem management in Southwestern grasslands (Savory and Butterfield 1999).

The question of whether grazing is a natural disturbance in the grasslands of the Southwest has been the subject of some debate. In the shortgrass prairie, blue grama and buffalo grass *Buchloe dactyloides* coevolved with the bison and are apparently adapted to heavy grazing pressure; these grasses thrive under such conditions by reproducing both sexually and by tillering (Knopf 1994). By contrast, in more recent history the desert grasslands of the Southwest have been devoid of large herds of grazing ungulates. Most evidence points to an absence of large herds of bison west of the Rockies (Berger and Cunningham 1994, Durrant 1970, Gustafson 1972 as cited in Mack and Thompson 1982, Kay 1994). Although Southwestern grasses undoubtedly coevolved with grazing due to the presence of herbivorous megafauna in the Pleistocene, these grasses have now been released from selection for grazing defenses for at least 10,000 to 12,000 years (Jones 2000 and references therein). In their extensive review, Milchunas and Lauenroth (1993) report that the sensitivity of grasslands to grazing increases with increased aridity and/or the lack of an evolutionary history of grazing. If one accepts their results, it is hard to avoid the conclusion that the grasslands of the Southwest must be especially sensitive to potential grazing impacts.

The bunchgrasses of the arid Southwest are indeed highly susceptible to grazing by ungulates and respond in a manner quite distinctive from the grasses of the shortgrass prairie (Daubenmire 1970, Dyer 1979, Tisdale 1961). Whereas grazed areas in the shortgrass prairie tend to be recolonized by predominantly native plants (Mack and Thompson 1982 and references therein), the morphological and physiological features of bunch grasses render them incapable of recovering quickly from grazing. Continuous grazing in desert grasslands leads to changes in species composition, where bunch grasses are replaced by sod-forming grasses or annuals (Brown 1982), or invaded by Eurasian weeds (Mack and Thompson 1982; see also Milchunas and others 1988). Furthermore, the soils of these grasslands that evolved with few native grazers are protected by a cryptogamic crust of mosses, lichens, and liverworts; this crust can be permanently destroyed by the trampling of large ungulates, producing sites for the establishment of exotic species (Daubenmire 1970, Jones 2000, Mack and Thompson 1982). Uncontrolled livestock grazing also endangers riparian systems in grasslands, one of the greatest sources of local diversity, because livestock will eat the palatable woody species such as cottonwoods and willows, not only removing the bulk of the riparian plant community but also destabilizing the banks and potentially leading to a lowering of the water table (Kovalchik and Elmore 1992).

Grazing impacts in the Southwest remain a highly controversial and confusing issue. As Jones (2000) points out, the poor experimental design employed in the majority of grazing studies has left us with a dearth of solid information about the impacts of grazing on arid rangelands. Furthermore, results are contradictory. Although Jones (2000) found that grazing had negative impacts in the majority of studies reviewed, Stohlgren and others (1999) suggest that factors such as soil fertility or water availability may overshadow the impacts of grazing on variables such as native plant species richness. There is little question that the astronomical stocking rates of livestock in the late 1800s did great environmental damage to the grasslands of the American Southwest; what is not well known is how current grazing practices are impacting the system (Curtin and others 2002). From a management standpoint, one important point to keep in mind is that restoration of degraded rangelands will require much more than merely removing cattle. Ecosystem function must be restored, which demands the incorporation of dynamic processes such as fire and precipitation to affect grassland condition, and furthermore may require mechanical removal or chemical treatment to turn the clock back on woody invaders (Curtin and others 2002, McPherson and Weltzin 2000). Today there is a generally greater awareness of the importance of maintaining ecosystem function for long-term sustainability, and a growing emphasis on the proper management of livestock grazing to ensure the ecological integrity of Southwestern grasslands (Brown and McDonald 1995, Curtin 2002, Savory and Butterfield 1999, Sayre and Ruyle 2001), as witnessed by the recent evolution of sustainable ranching organizations such as the Malpai Borderlands Group and The Quivira Coalition. More well-designed scientific studies of various grazing practices and their effects on the biodiversity of Southwestern grasslands are clearly needed to eliminate the confusion surrounding this issue and to develop sound management guidelines.

Exotic Grasses

A mounting problem in the West is the spread of exotic grasses. Grasses such as cheatgrass and crested wheatgrass (*Agropyron cristatum*) may be intentionally introduced as livestock forage or invade following disturbance, soon displacing native grasses (Mack 1981, Marlette and Anderson 1986). Although exotics such as lovegrasses (*Eragrostis* spp.) are planted as cattle forage, these grasses actually increase in response to grazing, as the livestock tend to preferentially forage on the native grasses and reduce competition for the lovegrasses (Bahre 1995 and references therein).

From a biodiversity standpoint, one of the problems with at least some exotic grasses is that they do not appear to provide adequate habitat for native grassland species. In Arizona, grasslands that have been seeded with Lehmann and Boer lovegrass (*Eragrostis lehmanniana* and *E. curvula* var. *conferta*) have been described as "biologically sterile" (Bock and others 1986:462). Twenty-six native species (10 plants, five birds, three rodents, and eight grasshoppers) were found to be significantly more abundant in native grasslands; only three native species (one bird, one rodent, and one grasshopper) were more common in the grasslands dominated by the African lovegrasses. Bock and others (1986:462) explain: "Indigenous animals appear to have evolved specific dependencies on the native flora and/or its associated fauna, insofar as most find the exotic grasslands far less inhabitable."

The increase of exotic grasses in the Southwest may have further ramifications as well, as they alter the natural fire regimes and lead to further ecological changes in the system (Anable and others 1992, Cox and others 1990). Some exotics such as Lehmann lovegrass increase after fire, and such grasses provide more fine fuel to carry fires than native species of grasses (Cox and others 1984). This may result in a positive feedback loop, in which introduced grasses play a beneficial role in terms of increasing fire frequency, yet simultaneously have the negative consequences of extending the coverage of the less-desirable exotic grasses as well as increasing the intensity of fire (Anable and others 1992).

Shrub Encroachment

The landscape of the Southwest has been inexorably altered over the past century by an extensive invasion of woody plants into areas that were formerly grasslands. Numerous authors have documented this transition from grassland to shrubland (for example, Bahre 1991, Buffington and Herbel 1965, Glendening 1952, Humphrey 1987) using a variety of techniques including early survey records (York and Dick-Peddie 1969) and photo points (Hastings and Turner 1965). The evidence suggests that although about 75 percent of southern New Mexico was covered in grasslands prior to the late 1800s, by the late 1960s only 5 percent grassland coverage remained (York and Dick-Peddie 1969). In another study of the Chihuahuan Desert, an estimated 25 to 50 percent of the area that is currently covered by shrublands was actually grassland less than 200 years ago (Dinerstein and others 2000). Such a conversion represents a significant loss of habitat for both plant and animal species that are grassland specialists.

Many factors appear to have played a role in this transformation. The conversion of grasslands to shrublands is a common result of overgrazing (Risser 1988). As livestock preferentially consume the more palatable species, initially the perennial grasses, competition is reduced and unpalatable woody species have the opportunity to become established (Humphrey 1958). Furthermore, heavy grazing reduces the fuel loads provided by grasses to the point that fire frequency and intensity become reduced, thereby removing the natural source of control for woody shrubs in grassland systems (Archer 1989).

Increases in woody plants such as mesquite following active fire suppression were recognized early on by Griffiths (1910). Since that time, the critical role of periodic fires in restricting woody plant establishment has been clearly demonstrated (for example, McPherson 1995 and references therein). Humphrey (1958:37) argued that the grasslands of the Southwest are a "fire-caused subclimax," but many other factors—such as soil type and herbivory by native animals—are now believed to interact with fire to maintain the grasslands of the Southwest (Curtin and others 2002). Although fire alone is not considered sufficient to prevent shrub encroachment and maintain the grassland condition indefinitely (McPherson 1995), it is a critically important element, and human alteration of natural fire cycles through suppression efforts has undoubtedly facilitated the spread of woody plants into these grassland systems.

Another explanation for the shift from grassland to shrubland in the Southwest is climate change. It has long been recognized that shrubs will increase in grassland systems in response to drought (for example, Schlesinger and others 1990). However, Brown and others (1997) found evidence for shrub increases not in response to drought, but rather in response to increased levels of winter rainfall in recent years. Furthermore, through the observation of livestock exclosures they were able to document that these increases in woody plants occurred in spite of protection from grazing. The authors argue that under conditions of high winter precipitation, the establishment of cool-season C_3 woody shrubs is favored over that of the warm-season C_4 grasses that normally dominate the landscape (Brown and others 1997 and references therein).

Prairie dogs are believed to be instrumental in retarding the growth of woody invaders such as mesquite (Koford 1958, Weltzin and others 1997), and some authors have suggested that the elimination of prairie dogs may be partially responsible for the widespread encroachment of mesquite into Southwestern grasslands observed in recent years (Parmenter and Van Devender 1995). Other authors have also found that small mammals play an important role in maintaining grassland systems by restricting the establishment of woody plants (for example, Curtin and others 2000).

Whatever the mechanism(s), there is little doubt that the continuing expansion of woody plants and cacti into Southwestern grasslands is one of the greatest sources of habitat degradation or loss threatening grassland specialists today.

Table 4-3. Threatened and endangered terrestrial animal species of Southwestern grasslands. The following species are not necessarily restricted to grassland habitats; although many are grassland specialists, this list also includes those species that rely heavily on grasslands in some parts of their range or as one component of a mosaic of habitats utilized. Sources for the information presented here include Arizona Game and Fish Department (1988), Association for Biodiversity Information (2001), New Mexico Department of Game and Fish (2000), and U.S. Fish and Wildlife Service (2001).

Common name/ Scientific name	Federal status	New Mexico status	Arizona status	NM	AZ	Primary threats	Types of grassland habitat utilized	Note
Amphibians								
Great Plains narrowmouth toad *Gastrophryne olivacea*		E	C	•	•	Water developments, water table draw-down, overgrazing, road development	Flooded desert grasslands	
Northern leopard frog *Rana pipiens*			C		•	unknown	High elevation wet meadows	
Plains leopard frog *Rana blairi*			E	•	•	Habitat loss, bullfrog predation	Low elevation wetlands	
Northern casque-headed frog *Pternohyla fodiens*			C		•	Vegetation clearing, overgrazing, water table draw-down, roads	Desert grasslands and scrub	AKA lowland burrowing treefrog
Reptiles								
Bunchgrass lizard *Sceloporus slevini*		T		•	•	Habitat loss and degradation due to overgrazing, drought, shrub invasion	Dense grass cover at mid-elevations; alkali sacaton in NM	Now restricted primarily to montane grasslands
Desert tortoise *Gopherus agassizii*	T		C	•	•	Habitat fragmentation, habitat loss, overgrazing, off-road vehicles	Semi-desert grassland	
Massasauga *Sistrurus catenatus*			E	•	•	Agricultural development, road kills	Desert grasslands	
Arizona ridgenose rattlesnake *Crotalus willardi willardi*			C		•	Mining, woodcutting, road development, collecting	Ecotonal montane grasslands	
New Mexico ridgenose rattlesnake *Crotalus willardi obscurus*	T	E		•		Collecting, habitat degradation due to overgrazing	Ecotonal montane grasslands	
Birds								
Crested caracara *Caracara cheriway*	T		C		•	Human disturbance at nest sites	Desert grasslands	Small breeding population in Arizona (Pima Co.)
Northern aplomado falcon *Falco femoralis septentrionalis*	E	E	E	•		Collecting, pesticides, loss of grasslands to shrub invasion	Desert grasslands, savannah	Now primarily in Mexico
Ferruginous hawk *Buteo regalis*			T	•	•	Loss of prairie dogs, human disturbance at nest sites	Desert, plains grasslands & shrubsteppe	
California condor *Gymnogyps californianus*	E		E		•	Direct and indirect human persecution (shooting, lead poisoning, and so forth)	Forages in grasslands	Experimental population reintroduced in Arizona
Lesser prairie chicken *Tympanuchus pallidicinctus*	C			•		Habitat loss, fragmentation due to overgrazing, agriculture	Shortgrass steppe (in NM, shinnery oak-bluestem association)	
Masked bobwhite *Colinus virginianus ridgwayi*	E				•	Habitat loss and degradation due to overgrazing	Desert grasslands and scrub	
Mountain plover *Charadrius montanus*	PT			•	•	Loss of prairie dog towns (early declines due to market hunting)	Shortgrass steppe	

White-tailed ptarmigan *Lagopus leucuras altipetens*		E		•		Grazing of tundra habitats, human disturbance	Alpine tundra, montane grassland	
Gray vireo *Vireo vicinior*		T		•	•	Unknown – possibly clearing of shrubs	Juniper grassland	
(Arizona) grasshopper sparrow *Ammodramus savannarum ammolegus*		T		•	•	Habitat loss and degradation due to overgrazing	Arid grasslands, desert scrub	
Baird's sparrow *Ammodramus bairdii*		T	T	•	•	Urban development, overgrazing on wintering grounds; loss of native prairies on breeding grounds	Shortgrass steppe, plains, desert grasslands	Wintering migrant in Southwest
Bobolink *Dolichonyx oryzivorus*			E	•	•	Agricultural and urban development, overgrazing	Plains grasslands, agricultural fields	Local breeder in Arizona (Apache and Navajo counties)
Sprague's pipit *Anthus spragueii*			C	•	•	Agricultural and urban development, overgrazing	Winters in Sonoita and San Rafael grasslands (Cochise and Santa Cruz Co., Arizona)	Wintering migrant in Southwest
Mammals								
Black-tailed prairie dog *Cynomys ludovicianus*	C		E	•		Direct human persecution, canine distemper	Shortgrass steppe, desert and plains grasslands	Extirpated in AZ; remnant populations in NM
Hualapai Mogollon vole *Microtus mogollonensis hualpaiensis*	E		E		•	Loss of habitat due to overgrazing, recreational development	High elevation grassy areas near springs assoc. with ponderosa pine/mixed conifer forest	Listed as Hualapai mexican vole; distribution highly restricted
Navajo Mexican vole *Microtus mogollensis navaho*			T		•	Loss of habitat due to overgrazing, mining, recreational development	Mid-elevation montane grasslands	
(Arizona) montane vole *Microtus montanus arizonensis*		E		•	•	Loss of habitat due to water diversions, livestock impacts	High elevation mesic grasslands and marshes	
Least shrew *Cryptotis parva*		T		•		Loss of habitat due to water diversions, livestock impacts, agriculture	Mesic grasslands and marshes	
Meadow jumping mouse *Zapus hudsonius*		T	T	•	•	Loss of habitat due to overgrazing, urban encroachment	Montane meadows	
White-sided jackrabbit *Lepus callotis*		T		•		Loss and degradation of habitat due to overgrazing, agriculture	Desert grasslands	Found in Hidalgo County, NM (and Mexico)
Black-footed ferret *Mustela nigripes*	E		E	•	•	Elimination of prairie dogs (primary prey source)	Desert and plains grasslands in association with prairie dog towns	Extirpated; experimental reintroductions in NM and AZ
Mexican gray wolf *Canis lupus baileyi*	E	E	E	•	•	Direct human persecution	Desert grasslands	Extirpated; experimental reintroductions in NM & AZ
Sonoran pronghorn *Antilocapra americana sonoriensis*	E		E		•	Loss of habitat due to grazing, agriculture; in Mexico, poaching	Desert grasslands, galleta grasslands of SW Arizona	
Chihuahuan pronghorn *Antilocapra americana mexicana*			T		•	Overhunting, habitat loss and fragmentation	Southern Arizona grasslands	Extirpated; present populations reintroduced
(Desert) bighorn sheep *Ovis canadensis mexicana*		E		•	•	Overhunting, habitat loss and fragmentation	Desert grasslands, montane grasslands	Reintroduction programs under way

E = endangered, T = threatened, P = proposed, C = candidate

Summary of Threats to Biodiversity in Southwest Grasslands

Unfortunately the native biodiversity of Southwestern grasslands is threatened by multiple sources, as are the native flora and fauna of all major ecosystems today. The alteration of natural fire cycles, inappropriate grazing regimes, eradication of keystone species, exotic grasses, habitat loss to shrub encroachment—these are just a few of the many factors believed responsible for the plant and animal species of Southwestern grasslands that have declined to the point of being listed as threatened or endangered (tables 4-3 and 4-4). Human activities such as urban development, mining, water diversions, and collecting have all contributed to declines in biodiversity, as has the purposeful elimination of certain species in several cases. Given the vast array of potential factors impacting the biodiversity of Southwestern grasslands, the discussion here of threats is not meant to be comprehensive, but only to touch on some of the major sources of declines in native species richness. Habitat fragmentation, a major potential threat to myriad grassland species in the Southwest, is discussed separately in the following section.

Ecological Consequences of Habitat Fragmentation

Introduction and Theoretical Background

Human use of the environment has led to a condition in which large areas of formerly continuous landscapes

Table 4-4. Threatened and endangered plants of Southwestern grasslands. The following species are not necessarily restricted to grassland habitats; although many occur primarily in grasslands, this list also includes those species that are found in grasslands in some parts of their range or as one component of a mosaic of habitats utilized. Sources for the information presented here include Association for Biodiversity Information (2001), New Mexico Native Plants Protection Advisory Committee (1984), New Mexico Rare Plant Technical Council (2001), and U.S. Fish and Wildlife Service (2001).

Common name/ *Scientific name*	Federal status	NM	AZ	Primary threats	Grassland habitat	Notes
Arizona agave *Agave arizonica*	E		•	Collecting, overgrazing	Juniper grasslands 1100-2750 m	Endemic to central AZ
Cochise pincushion cactus *Coryphantha robbinsorum*	T		•	Collecting, pesticides, mining	Limestone hills in desert grasslands 1280 m	Only two populations, one in SE AZ and one in Mexico
Pima pineapple cactus *Coryphantha scheeri* var. *robustispina*	E		•	Collecting, livestock impacts, ORVs, habitat loss due to development	Open, flat alluvial basins of semi-desert grasslands and Sonoran desert-scrub 700-1400 m	Pima and Santa Cruz Co., Arizona and Sonora, Mexico
Kuenzler's hedgehog cactus *Echinocereus fendleri* var. *kuenzleri*	E	•		Collecting, livestock impacts	Great Plains grassland 1600 – 2000 m	Southcentral New Mexico
Fickeisen pincushion cactus *Pediocactus peeblesianus* var. *fickeiseniae*	C		•	Collecting, livestock impacts, ORVs	Limestone soils in Great Plains grasslands ~1500m	Coconino and Mohave Co., Arizona
San Francisco Peaks groundsel *Senecio franciscanus*	T		•	Recreational: off-trail hiking and climbing	Alpine tundra 3350-3750 m	Isolated mountain endemic
Sacramento Mountains Thistle *Cirsium vinaceum*	T	•		Water development, livestock impacts	Wet meadows ~2440 m	Endemic; persists only in areas too steep for livestock
Canelo hills ladies' tresses *Spiranthes delitescens*	E		•	Water diversions, livestock impacts, exotic species, mining	Permanently wet meadows (cienegas) ~1525 m	Limited to four cienegas in southern Arizona
Gypsum wild-buckwheat *Eriogonum gypsophilum*	T	•		Gypsum mining, recreational development	Open gypsum in grama grasslands ~1500m	Isolated population in Eddy County, NM

E = endangered, T = threatened, P = proposed, C = candidate have become increasingly fragmented and isolated. Urban development, agriculture, power lines, road construction, and other such activities have accelerated over the past century, subdividing the natural world into disjunct remnants of native ecosystems embedded in a matrix of anthropogenic land uses (Saunders and others 1991). The negative ecological impacts of such fragmentation on natural systems has led many conservation biologists to identify habitat fragmentation as one of the greatest threats to biodiversity today (Harris 1984, Noss and Cooperrider 1994, Wilcox and Murphy 1985).

Under the traditional definition, there are two fundamental components to habitat fragmentation. First, the activity that leads to fragmentation usually leads to an outright loss of some area of the original habitat; this component can be considered habitat loss or destruction. The second component is fragmentation per se, in which the remaining natural areas are relegated to patches of reduced size isolated from one another across the landscape (Wilcove and others 1986). A common analogy is that these fragments now exist essentially as habitat islands in a sea of altered or degraded lands; thus, this effect is also referred to as insularization (Wilcox 1980). The introduction of "edge effects" might be considered a third fundamental component of fragmentation. Edge effects are manifested in the form of altered physical processes and biotic interactions along habitat edges. The amount of edge habitat may increase dramatically through the process of fragmentation because reducing the size of the habitat patches results in a proportional increase in the amount of edge (Janzen 1983, Williamson 1975, Yahner 1988); altering the shape of fragments may also have this effect (Diamond 1975, Wilson and Willis 1975).

Many of the negative impacts of fragmentation are based on the principles of island biogeography, a classic model in conservation biology that predicts the number of species that will be found on an island as a function of species colonization and extinction rates, the size of the island, and its degree of isolation (MacArthur and Wilson 1967; fig. 4-1). Large islands near a potential source of immigrants have the greatest rates of colonization because individuals can traverse a short distance more easily, and the size of the island makes it more likely that dispersing individuals will happen upon it. Small islands far from the source have the least chance of intercepting potential colonists; it is less likely that individuals will be capable of traveling the greater distance required, and the small size of the island makes it less likely to be discovered. Balanced against the effect of colonization is that of extinction. Large islands support large populations of different

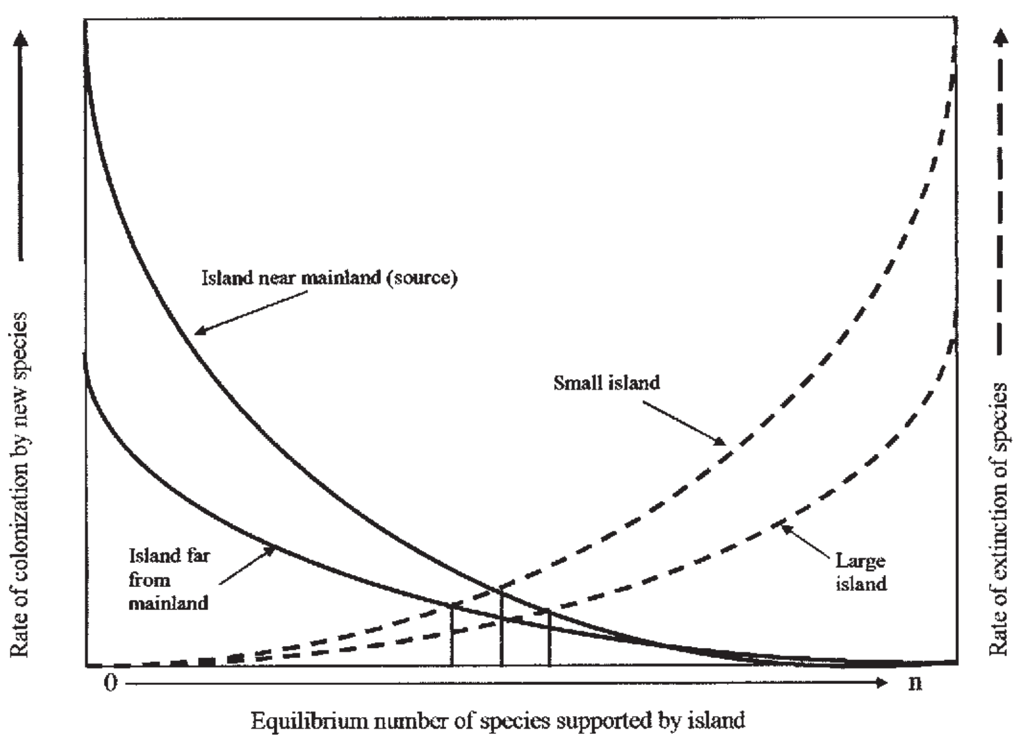

Figure 4-1. Graphic representation of island biogeography theory. Large islands located near a potential source of colonists should support a greater equilibrium number of species due to high immigration rates and low extinction rates. Small, isolated islands are predicted to have the least number of species due to lower colonization rates and greater extinction rates (after MacArthur and Wilson 1967).

species, and as large populations are more resilient in the face of potential extinction events, extinction rates should be relatively low. Small islands would support small populations of different species, and as small populations are particularly vulnerable to extinction (Harris 1984, Saunders and others 1991), extinction rates would be high. Based on the interaction between extinction rates and colonization rates, the model thus predicts that large islands located near a potential source of immigrants will support the greatest equilibrium number of species, while small, isolated islands will support the lowest number of species.

Although the theory of island biogeography was based on the species-area relationships observed on oceanic islands, this model has been widely applied to habitat fragments ("islands") in continental terrestrial systems as the basis for conservation planning (for example, Shafer 1990) and has largely formed the foundation of scientific inquiry into the effects of habitat fragmentation. Closely aligned with the theory of island biogeography and often applied to fragmented systems as well is the idea of metapopulation biology. The metapopulation concept holds that a population may be composed of a number of scattered subpopulations that are subject to repeated colonizations and extinctions, but that as a whole generally persists at some equilibrium level of abundance over time (Levins 1969, 1970). Metapopulations are typically characterized by one or more core or source populations and several fluctuating satellite populations. Satellite populations may occasionally go extinct when conditions are not favorable, but are replaced by new colonists dispersing from the core population when conditions improve (Bleich and others 1990). The viability of a metapopulation thus depends on the persistence of the core subpopulation and the ability of dispersing individuals to balance local extinction events by successfully recolonizing vacant patches (Gilpin and Hanski 1991); such recolonization events have been deemed the "rescue effect" (Brown and Kodric-Brown 1977).

Much of applied conservation biology rests on the precepts of island biogeography theory, the metapopulation concept, and the avoidance of edge effects. Basic principles of conservation design in fragmented landscapes include maximizing the size of habitat fragments to preserve species diversity and reduce extinction risk, minimizing the distance between fragments to facilitate dispersal, and controlling the shape of fragments to minimize the amount of edge relative to core habitat (Diamond 1975, Shafer 1990). In recent years, the major hypotheses stemming from the application of these theories (for example, that small fragments will support fewer species than large fragments) have been tested repeatedly and with largely variable results (discussed below; also see Bierregaard and others 1992, Debinski and Holt 2000 for a review). The vast majority of studies on the effects of fragmentation have centered on forested landscapes, particularly those in the tropics (for example, Lovejoy and others 1984) and the Central or Eastern United States (for example, Askins and others 1990, Robbins 1980, Robinson and others 1995, Whitcomb and others 1981, Wilcove and Robinson 1990). Few studies have focused on the impacts of fragmentation in North American grasslands (for example, Collinge 1998, 2000, Quinn and Robinson 1987, Robinson and Quinn 1988), and many of these have concentrated primarily on birds of the tallgrass prairie in the Midwest (for example, Johnson and Temple 1990, Winter and Faaborg 1999). As the effects of fragmentation in Western grasslands have gone largely unstudied, the discussion of the ecological impacts of habitat fragmentation that follows here is of necessity derived from studies conducted primarily in other ecosystems. On an ecological level grasslands suffer many of the same consequences of habitat fragmentation as do forested areas, although the contrast between the natural and altered conditions may not appear as abrupt. Data from grassland systems are utilized whenever they are available.

Edge Effects

Habitat edges exhibit a marked contrast in the structure and species composition of the vegetation between two adjacent elements in the landscape. Although natural edges occur in nature, as when blowdowns create openings in forests, the most common edge in a fragmented landscape is the product of human activity—an "induced" edge (Yahner 1988). Such edges have been associated with numerous negative impacts on the organisms originally inhabiting the remaining fragment, including increased levels of parasitism and predation, changes in species composition, and physical alterations in environmental conditions; these impacts are collectively known as edge effects (Lovejoy and others 1986, Yahner 1988). In forest systems, it is the removal of trees that results in fragmentation and the creation of habitat edges. Clearing for logging, development, agriculture, road construction, and other purposes all contribute to this process. In grassland ecosystems, it is just the opposite: the *introduction* of trees or shrubs is one of the primary causes of fragmentation and edge effects. Such seemingly innocuous human constructs as treelines planted for windbreaks or fencerows, stringers of trees along irrigation ditches, and trees along roadsides create long, linear stretches of edge habitat that can negatively impact the native species of the surrounding grasslands (O'Leary and Nyberg 2000).

Trees and shrubs create vertical structure in the grassland landscape, providing cover and perches for predators and leading to increased levels of predation along the edge habitat created by the interface between the grassland and the treeline (Burger and others 1994, Gates and Gysel 1978, Johnson and Temple 1990, Møller 1989, Ratti and Reese 1988, Winter and others 2000). Nest predators such as jays, raccoons, skunks and opossums hunt preferentially along the perimeter of agricultural fields or not far from a wooded edge, and prairie raccoons are known to use shelterbelts as travel lanes (Bider 1968, Fritzell 1978, Gates and Gysel 1978, Wilcove 1985). Structures such as fences, telephone poles, or rooftops provide perches for predators as well, and any human developments also tend to serve as a source of "urban predators" such as cats (Wilcove 1985). Perches that provide a good view to scan for potential host nests are also considered a critical habitat feature for brown-headed cowbirds, a nest parasite (Norman and Robertson 1975). In grassland systems, the introduction of trees, shrubs, or human structures provides these lookout sites, leading to significant increases in parasitism levels along such edges and resulting in reduced nest productivity or even nest failure in grassland breeding birds (Best 1978, Johnson and Temple 1986, 1990, Wray and others 1982). These effects have been found to extend into grasslands up to 75 m in from a woody edge (Burger and others 1994, Helzer 1996, Paton 1994), and many grassland breeding birds appear to avoid nesting or foraging within this zone (Delisle and Savidge 1996, Johnson and Temple 1990). In addition, grassland birds that do not tend to fly toward shrubs for cover when disturbed have been found to actively avoid woody edges, and the density of these birds tends to decrease as the amount of woody cover increases (Lima and Valone 1991). Such impacts are of particular concern because most species of grassland breeding birds have been exhibiting consistent and often striking population declines over the past few decades (Herkert 1994, Peterjohn and Sauer 1999, Samson and Knopf 1994), and many of these declines are believed to be linked with the loss and fragmentation of native grassland habitats (Herkert 1994, Johnson and Temple 1986, 1990, Peterjohn and Sauer 1999, Samson 1980, Vickery and others 1994).

The creation of edges, whether through increased woody vegetation or clearing, opens up avenues for incursion by opportunistic "edge" species and invasive exotics. The disturbance of native plant communities facilitates invasion by weedy and/or exotic plants, and such disturbance events typically accompany the activities that lead to fragmentation, such as road construction (Ewel 1986, Hobbs 1989, 1991, Rejmanek 1989, Saunders and others 1991, Schowalter 1988). Such increases in edge species or habitat generalists have also been found in such diverse taxa as insects (for example, Suarez and others 1998, Webb and Hopkins 1984), frogs (Laurance and Bierregaard 1996) and birds (for example, Herkert 1994, Samson 1980). Fragmentation and edge effects have been found to have a dramatic impact on the diversity of native ant species, for example. Suarez and others (1998) found that habitat fragments were characterized by high numbers of introduced ant species along the edges, and that unfragmented control plots supported three times as many native ant species as did habitat fragments. Furthermore, there was a negative correlation between the number of native ant species and time since fragmentation, suggesting that the native ants were incapable of recolonizing patches in the fragmented landscape once local extinctions had occurred.

Changes in the vegetative structure or species composition of the plant community may effect some changes in the animal community as well, typically leading to increased numbers of opportunistic species or habitat generalists (Saunders and others 1991). In grassland communities, the introduction of woody vegetation is correlated with increased species diversity of birds and lizards due to greater representation by generalists or species that normally utilize shrubby vegetation, while grassland specialists that formerly occupied the area tend to be lost (Germano and Hungerford 1981, Saunders and others 1991, Schmiegelow and others 1997). Changes in the faunal composition of habitat fragments may also impact the remainder of the community. In California grasslands, there was a significant correlation between the loss of native mammal species richness and the numbers of exotic birds and mammals occupying habitat fragments (Smallwood 1994). The loss of native species, it is proposed, leads to unstable population dynamics and lowers the "biotic resistance" (Simberloff 1986) of the community, leaving it vulnerable to invasion by exotics. Edges allow for the infiltration of formerly inaccessible interior habitats by a diverse array of invasives, but while fragments tend to support increased numbers of exotic or opportunistic species, habitat specialists tend to consistently decline within these patches (for example, Harris and Scheck 1991, Herkert 1994, Robinson and Quinn 1988, Samson 1980, Suarez and others 1998, Verner 1986, Webb 1989).

Impacts on the physical environment and ecological processes—The reduction in area of the original habitat and concurrent increase in the amount of edge can provoke physical changes in the fragment microclimate. Studies of forest systems have found that habitat fragments experience increased solar radiation along edges, altering plant species composition and leading to higher soil temperatures, in turn potentially affecting nutrient cycling (Lovejoy and others 1986, Saunders and others 1991). Increased soil temperatures may impact the numbers and activities of soil-dwelling organisms involved in decomposition as well as decrease the moisture retention capacity of the soil (Klein 1989, Parker 1989, Saunders and others 1991). Whether increased solar radiation effects such changes in grassland systems is largely unknown, although one study reports that nutrient cycling was not affected in studies of fragmented old fields (Debinski and Holt 2000). Another physical edge effect in forest fragments is the increased penetration of wind, which may result in direct physical damage to the vegetation or act to increase evapotranspiration and hence desiccation (Lovejoy and others 1986). Although grassland systems probably do not experience the same impacts due to their naturally short stature, winds do have increased accessibility to grasslands along cleared edges, resulting in the potential for increased transport of seeds, insects, and disease organisms into grassland fragments (Hobbs and Atkins 1988, Saunders and others 1991). Fragmentation can lead to changes in water regimes, as cleared areas contribute to increased runoff and erosion and lowered absorption of water into the soil (Kapos 1989, Saunders and others 1991). Replacement of deep-rooted native perennial grasses with introduced annuals can also contribute to reduced evapotranspiration rates, increased runoff, and increased temperatures at the soil surface. Such changes in the moisture levels of soils and runoff patterns can lead to the creation of new substrates for invasion by exotic or weedy plant species, impact seedbed characteristics, and result in the displacement of organisms that are unable to survive the altered environmental conditions (Hobbs and Huenneke 1992, Jones 1981, Saunders and others 1991).

Fire regimes may also be affected by habitat fragmentation. Most grasslands are considered fire-dependent ecosystems, requiring frequent fires to set back succession and maintain the natural distribution, productivity, and diversity of the grassland (McPherson 1995). As discussed above, habitat fragments are vulnerable to invasion by exotic species, and increased numbers of exotic grasses may seriously disrupt normal fire cycles. Introduced species such as lovegrasses (genus *Eragrostis*) are common throughout Southwestern rangelands (Loftin and others 2000). Sites dominated by lovegrasses may exhibit biomasses up to four times that of native grasslands, resulting in abnormally high fire frequencies and intensities that tend to kill the native plants but that lead to even greater abundances of the lovegrass (Anable and others 1992, Cox and others 1990). Fragments of native grasslands, on the other hand, may face the problem of decreased fire frequency. As fragments diminish in size, it becomes increasingly unlikely that they will be struck by lightning frequently enough to maintain the grasslands. A study of small prairie fragments in Wisconsin showed that the absence of fire over 32 to 52 years resulted in a loss of between 8 and 60 percent of the original plant species (Leach and Givnish 1996). Rare plants showed the greatest losses from these grassland fragments in the absence of fire. Roads and other agents of fragmentation may also act as firebreaks, restricting the spread of what would otherwise be extensive range fires. Finally, fragmentation due to human habitation also provides an incentive for active suppression of fires that could potentially threaten structures (Hansen and others 2002).

Area-Sensitive Species: Interaction of Edge Effects and Habitat Reduction

That species richness will decrease as a function of reduced geographic area is the most basic prediction of island biogeography theory. Studies of grassland birds show that this guild closely follows this prediction, as species richness is significantly correlated with the size of habitat fragments (Herkert 1994, Samson 1980). However, several species of grassland birds drop out of the community even in fragments that appear large enough to support them. These species simply will not utilize habitat fragments below a certain threshold size for nesting, even if the fragment is large enough to hold several average-sized territories and the habitat appears to be suitable; such species have been termed “area-sensitive” (see for example Herkert 1994, O’Leary and Nyberg 2000, Samson 1980, Vickery and others 1994; for forest birds, see Blake and Karr 1987, Robbins 1980, Robbins and others 1989, Winter and Faaborg 1999). The minimum area required by area-sensitive species varies widely: Eastern meadowlarks require only 5 ha, whereas Henslow’s sparrow will not nest in a fragment of less than 55 ha. Grasshopper sparrows and savannah sparrows fall toward the larger end of the range at 30 and 40 ha, respectively (Herkert 1994). Greater prairie chickens and upland sandpipers are well known for their avoidance of small grassland fragments and are found regularly breeding in fragments of 160 ha or more (Cannon and Christisen 1984, Samson 1980, Westemeier 1985). Furthermore, not only does nest density and nest success decrease with fragment size (Burger and others 1994, Johnson and Temple 1986, 1990, Samson 1980, Winter and Faaborg 1999; but see Delisle and Savidge 1996), but the simple distribution and density of several species of grassland birds is also positively correlated with fragment size (Helzer 1996, Herkert 1994, Winter and Faaborg 1999).

Exactly why these birds avoid small fragments, even when adequate suitable habitat appears to be available, is unclear. Most likely it is not the amount of available habitat per se that is important, but rather the amount of core habitat—that is, the amount

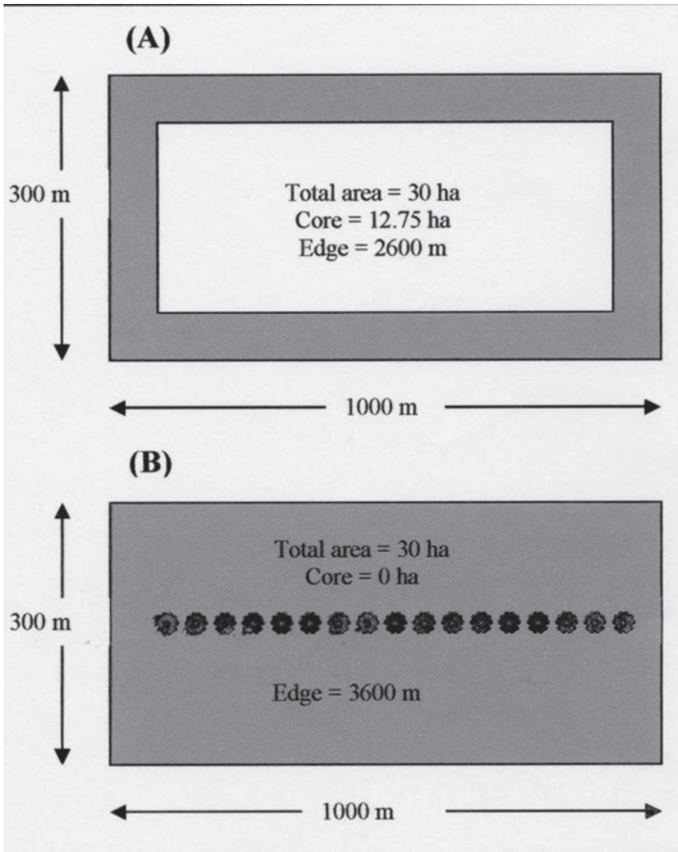

Figure 4-2. An example of how fragmentation and edge effects may render what appears to be an adequate area of quality habitat unsuitable for nesting or other activities for grassland birds. (A) Assuming edge effects extend a distance of 75 m into the interior from any edge (the shaded area), an area of 30 ha would offer 12.75 ha of potential core habitat. (B) Bisecting this area with a treeline (an induced edge) effectively places the entire area within the zone of edge effects and eliminates all potential core habitat, even though the total area is essentially unchanged.

of habitat that is far enough into the interior of the fragment to escape the edge effects of increased predation and parasitism—that is the critical factor (Brittingham and Temple 1983, Burger and others 1994, Gates and Gysel 1978, Johnson and Temple 1986, Winter and others 2000; see "Edge Effects" section above). Grassland birds clearly avoid nesting close to edges in habitat fragments (Delisle and Savidge 1996; Johnson and Temple 1990; O'Leary and Nyberg 2000; Warner 1994; Winter and others 2000); small fragments, particularly if they are somewhat linear in shape, may simply not provide any core area for nesting (fig. 4-2). Avoidance of edge for other activities, such as foraging, may also be a reaction to increased predator activity along edges (Andrén and Angelstam 1988, Fritzell 1978, Gates and Gysel 1978, Johnson and Temple 1986, 1990, Wilcove 1985, Yahner and Scott 1988). Several studies have found that the edge:area ratio of fragments has a greater influence on the presence and richness of grassland birds, and on the presence and success of nesting species, than does fragment area (Burger and others 1994, Helzer and Jelinski 1999, Temple 1986, Winter and others 2000). Area-sensitivity has been attributed only to birds thus far, but may possibly occur in other taxa as well.

Loss of Grassland Habitats and Fragmentation in the Southwest

In the Midwest, as little as 4 percent of the original native tallgrass prairie is estimated to remain; in some states, that figure may drop as low as 1 percent (Samson and Knopf 1994). Most of the Midwestern grasslands have been cleared for agriculture, particularly rowcropping for products such as wheat and corn. Destruction of Southwestern grasslands due to clearing for rowcrop agriculture has been relatively minimal, since such crops are few (for example, chile, cotton) and cover only a small portion of the land area of New Mexico and Arizona. Nonetheless, clearing for such purposes has contributed to the outright loss of native Southwestern grasslands, as has clearing for urban development (Bahre 1995). Grazing is the predominant use of Southwestern rangelands, and improper grazing practices can lead to loss of grasslands not through clearing per se, but through degradation of the grasslands to the point that they no longer function as suitable habitat for native species (for example, Bahre 1995, Bock and others 1984, Bock and Webb 1984, Noss and Cooperrider 1994). Uncontrolled heavy use of native arid grasslands by domestic livestock can lead to the loss of native grasses, the introduction of invasive exotic grasses and other weedy species, the destruction of cryptogamic crusts, altered grassland structure, and contribute to the conversion of grasslands to shrub-dominated desert scrub or pinyon-juniper (Bahre and Shelton 1993, Hobbs and Huenneke 1992, Humphrey 1958, Mack 1981, 1989, Martin 1975, Moore 1970, Wright and others 1979). Most grassland systems are maintained by periodic fires that set back succession, but a history of fire suppression has allowed the widespread encroachment of shrubs and trees into such systems (Humphrey 1958, McPherson 1995). The increased invasion of grasslands by exotic plants facilitated by grazing, road construction, and other forms of disturbance also contribute to altered fuel structure and fire regimes, leading to the eventual conversion of the native grassland to some other habitat type (Loftin and others 2000, MacDonald and others 1989, Panetta and Hopkins 1991, Saunders and others 1991). Whether lost through outright clearing, degradation, or gradual conversion, the end result of such habitat loss is twofold: first, there is an overall reduction in the area of extant native grassland, and second, those grasslands that do remain are relegated to disjunct fragments of relatively small size.

A new term for one particular source of fragmentation is becoming increasingly prevalent in the Western United States: exurban development (Knight 1999). Exurban development refers to low-density residential development that occurs beyond the limits of incorporated towns and cities. Expanses of land that were once devoted to agriculture or ranching are subdivided and sold for the development of "ranchettes" (single houses generally situated on from 10 to 40 acres of land) which contribute to this new trend of rural sprawl (Brown and McDonald 1995, Hansen and others 2002, Theobald 2000). Between 1994 and 1997, nearly 80 percent of the new home construction in the United States was in nonmetropolitan areas, and 57 percent of the houses were built on lots equal to or greater than 10 acres (Heimlich and Anderson 2001). This conversion of private ranching and farming lands to rural residential developments has been called "the most profound land use change in the New West" (Maestas and others 2002).

Although much of the land surrounding these homes remains in a relatively natural state, these low-density rural developments still introduce the negative effects of fragmentation into the environment with the associated predictable negative impacts on native biodiversity. The construction of buildings, roads, fences, and other structures associated with these rural subdivisions result in a dramatic increase in habitat fragmentation (Knight and others 1995). Knight (2003) reports that approximately one-fifth of the land area of a subdivided ranch is affected by houses and roads. The native species community composition changes in a predictable fashion, as specialized native species, such as dusky flycatchers, tend to be replaced by generalist, human-adapted species, such as black-billed magpies (Maestas and others 2002; see also Hansen and others 2002, Odell and Knight 2001). These changes are apparently little affected by the density of the housing development; that is, these effects are seen whether houses are densely clustered or spread more widely across the landscape (Odell and Knight 2001).

Predation and parasitism on native birds and mammals increases as residential development brings a concurrent increase in predators, both in the form of family pets and through associated increases in human-adapted species such as brown-headed cowbirds or jays (Hansen and others 2002, Maestas and others 2002). Nonnative plant species also tend to increase in association with exurban developments (Knight and others 1995, Maestas and others 2002), and natural disturbance regimes (such as fire) are disrupted (Bahre 1995, Hansen and others 2002). Furthermore, people tend to settle in the same areas that are most favored by wildlife, and outdoor recreationists moving into these rural areas both disturb and displace native wildlife (Hansen and others 2002). Although there has been little discussion of this issue until relatively recently, conservation biologists and land managers are becoming increasingly concerned about this newest threat to the biodiversity of the Western United States, apparently with good reason (see for example, Bahre 1991, 1995, Brown and McDonald 1995, Hansen and others 2002, Knight and others 2002, Odell and Knight 2001).

Changes in Species Richness and Species Composition

The reduced size and increased isolation of areas of native habitat have numerous theoretical repercussions for the native species that depend upon them, such as reduced species richness in the remaining fragments, interference with dispersal and colonization abilities, interruption of metapopulation dynamics, and increased risk of extinction (Meffe and others 1997, Noss and Cooperrider 1994, Wilcove and others 1986). Examination of these hypotheses in studies of habitat fragmentation has yielded mixed results. Although some habitats reduced in size do exhibit decreased species richness as predicted by island biogeography theory (Baur and Erhardt 1995, Bierregaard and others 1992, Collinge and Forman 1998), many either maintain the same number of species as prior to fragmentation, or actually exhibit an increase in species richness (Quinn and Robinson 1987, Simberloff and Abele 1982, Simberloff and Gottelli 1984). This is one of the key problems in applying island biogeography theory to continental systems: real islands are surrounded by a habitat matrix that is truly inhospitable to terrestrial species, whereas habitat "islands" are often encompassed by a matrix of habitats that, although hostile, may be habitable to some extent (Andrén 1994). In continental systems this matrix may serve as a source of potential colonists, allowing for the invasion of habitat fragments by weedy edge species, habitat generalists, or exotics (Doak and Mills 1994, Noss and Cooperrider 1994, Zimmerman and Bierregaard 1986). In such cases, fragmentation may actually result in an increase in species richness. However, the key point that is often overlooked is that while the overall number of species may rise, the *species composition* of the fragment may be irretrievably altered. Sensitive species of habitat interiors, endemic species, or habitat specialists may be lost, while numbers of common opportunistic species increase (for example, Harris and Scheck 1991, Lynch 1987, Noss 1983, Samson 1980, Verner 1986, Webb 1989, Webb and Hopkins 1984; see also discussion in the "Edge Effects" section above).

Although the number of species found in a habitat fragment may initially be high, one theory holds that over time the number of species the reduced fragment can now support will eventually drop—a process known as "relaxation" (Diamond 1972). A common criticism of contemporary studies of habitat fragmentation is that the time frame is too short—often on the order of just a few years—to document the relatively slow process of extinction following fragmentation (for example, Andrén 1994, Gonzalez 2000, Schmiegelow and others 1997, Tilman and others 1994). The process of species relaxation has been observed to some degree in birds (Schmiegelow and others 1997), small mammals, and insects (Debinski and Holt 2000), but perhaps the most thorough documentation of this phenomenon comes from a study of a microarthropod community in a bryophyte-based microlandscape (Gonzalez 2000). The reduced spatial and temporal scale of the dynamics in this community enabled the investigator to observe the effects of habitat fragmentation over many generations, which did in fact finally result in substantial numbers of local extinctions. Fragmentation thus has two effects on species richness operating on two time scales: first, immediately following fragmentation there is an "instantaneous sampling effect," in which species richness is a sample of the richness at a larger scale; second, there is the long-term process of community relaxation, resulting in a decrease in species richness to a new steady state value (Gonzalez 2000). This difference between the initial postfragmentation level of species richness and the eventual lowered steady state value has been termed "the extinction debt," because although the extinctions do not occur until many generations following fragmentation, they are bound to occur and are thus a debt that will come due in future years (Tilman and others 1994). The model upon which the extinction debt is based showed a 50 to 400 year or more time lag between habitat destruction and species extinctions; it also predicted that even those species initially most abundant in undisturbed habitat fragments can be the same species that are most susceptible to eventual extinction (Tilman and others 1994).

Vulnerability to Local Extinction

Habitat fragments may lose species for many reasons. Those species generally considered most susceptible to local extinctions are naturally rare species, species of habitat interiors or "area-sensitive species" (see discussion above), sedentary species, species with limited dispersal capabilities, species with specialized habitat requirements (especially if the resources required are patchy or unpredictable in occurrence) and animals with large home ranges or wide-ranging animals (Meffe and others 1997, Saunders and others 1991, Wilcove and others 1986). For naturally rare species, or those that occur at low densities in the environment, extinction due to fragmentation is largely a matter of chance. Being widely distributed across the landscape, the initial persistence of such a species would depend upon the likelihood that any remnant habitat patches just happen to capture some individuals of the population. The long-term maintenance of the larger population would depend upon the ability of these surviving individuals to successfully interact and reproduce in the fragmented landscape.

For other organisms, survival in a fragmented landscape may depend on the size of the remaining fragments. For each species, there is some "critical threshold" size of habitat area below which the species cannot persist. A generic threshold of 10 to 30 percent of the remaining habitat has been reported for birds and mammals (Andrén 1994), but the exact value of any such threshold ultimately depends upon the scale at which an organism interacts with its environment. In other words, it depends upon whether or not individuals of the species in question perceive the landscape as connected or fragmented (With and Crist 1995). For example, a wide-ranging species that is a habitat generalist, such as a robin, might essentially be able to experience a fragmented landscape as functionally connected, because the robin can easily utilize several disjunct fragments by flying between them and would be able to make use of the resources in most any fragment it happens upon. However, an animal with limited mobility and specialized habitat requirements such as a frog, might be incapable of crossing the surrounding landscape matrix and would therefore experience the same landscape as fragmented and restrictive. Furthermore, even if the frog managed to travel to another habitat patch, it would have to depend upon the presence of water in the new patch to persist there. The degree of fragmentation, then, as well as the value of a critical threshold, is a matter not only of the area of habitat remaining and its spatial arrangement, but also the habitat requirements and dispersal ability of the species in question (O'Neill and others 1988, Plotnik and Gardner 1993, With and Crist 1995). Even for relatively wide-ranging species, however, fragmentation can have significant impacts. Grassland raptors such as prairie falcons, ferruginous hawks, and rough-legged hawks have been found to decline in numbers if as little as 5 to 7 percent of the landscape becomes urbanized (Berry and others 1998).

Despite their excellent dispersal abilities, large animals are often the first to be lost from small fragments. For many of these, the remnant habitat patches may simply be smaller than their minimum home range or territory sizes. Some species of Midwestern raptors are thought to be declining because there are few tracts of habitat left that are extensive enough to meet their needs during the breeding season (Robinson 1991). Large carnivores typically maintain extensive home ranges; the home ranges of male mountain lions may exceed 400 km^2 (Seidensticker and others 1973). Mountain lions and other large carnivores such as grizzly bears are decreasing in numbers as the large tracts of habitat they require continue to shrink in size and become increasingly isolated from one another (Picton 1979, Wilcove and others 1986). Even many of our National Parks do not provide areas of habitat extensive enough to sustain populations of large animals over time without active management (Meffe and others 1997). Although larger parks are more likely to maintain their native animal communities, nonetheless nearly 30 species of mammals have experienced local extinctions from National Parks, including many smaller species such as rabbits (Newmark 1987, 1995). Overall, the current system of nature reserves in the world is considered to be too small to support viable populations of large carnivores and herbivores over the long term (Belovsky 1987, Grumbine 1990).

Animals with large area requirements face the problem of inadequate fragment size, but in addition these and all species face the problem of barriers to dispersal. Even for animals that have the ability to travel long distances, the terrain that must be traversed to move from one fragment to another is often so vast and hostile in nature that they stand little chance of surviving the trip. Roads are one potential barrier to dispersal and are a major cause of habitat fragmentation. One obvious consequence of roads is direct mortality. It is estimated that one million vertebrate animals are killed on roads in the United States every day (Lalo 1987). For the Florida panther, a wide-ranging species whose endangered status stems largely from habitat fragmentation, roadkill is the single greatest source of mortality (Meffe and others 1997). Roads also serve to block the movement of animals, both small and large, effectively isolating populations within habitat fragments. Many species of small mammals have been found to cross roadways only rarely, if ever (Adams and Geis 1983, Garland and Bradley 1984, Mader 1984, Oxley and others 1974). The same has been found for some carabid beetles (Mader 1984), and animals as large as black bears may find roads a barrier to movement (Brody and Pelton 1989). Even a 3-m dirt track was found to deter the movement of prairie voles and cotton rats in a Kansas grassland (Swihart and Slade 1984). Fencing of rangelands may also serve as a barrier to movement for large grassland species. Pronghorn, for example, normally travel across wide ranges but are restricted in their movements by their inability or reluctance to jump fences, potentially leading to death in cases where the animals are unable to escape particularly severe winter weather (White 1969, Wilson and Ruff 1999, Yoakum 1978). Although fences have now been designed to allow passage of pronghorn (Yoakum and others 1996 and references therein), they are not widely used, and recent studies demonstrate that fencing still serves as a barrier to natural pronghorn movements in the Southwest (van Riper and others 2001).

The inability of individuals to move freely between habitat patches may interrupt the stability of metapopulations, leading to their eventual decline and local extinctions. Key source populations may be eradicated in the process of fragmentation, or barriers such as roads, agricultural fields, or other inhospitable altered habitat may simply impede the dispersal of individuals to the point that the potential colonists required to shore up satellite populations are eliminated. Real world metapopulations in fragmented landscapes, such as that of the endangered bay checkerspot butterfly, closely follow the predictions of the theory of island biogeography: the probability of extinction of satellite populations increases with isolation from the source population and declines with increasing patch area (Thomas and Jones 1993).

Problem of Small, Isolated Populations

Long-lived species in particular may persist for many years following fragmentation due simply to the longevity of the individuals making up the population. Unless successful reproduction and recruitment is taking place, however, this species will disappear from the fragment as these individuals die out. For small populations in a fragmented landscape, the impediments to reproduction and recruitment are many. Simply by chance, the demographics of the population may not be conducive to successful reproduction; the age structure and sex ratio of the remaining few individuals are critical. A classic example of demographic misfortune is the dusky seaside sparrow: this endangered species was eventually reduced to a population of only six individuals, all of whom were male, thus dooming the species to extinction (Kale 1983). Successfully locating a mate is key to reproduction for most species, but fragmentation of the habitat may make it difficult for potential mates to find each other. Several studies have found a greater percentage of unmated male birds in small habitat fragments, indicating that females may not be able to locate them in isolated patches (Gibbs and Faaborg 1990, Robinson 1988, Simberloff and Gotelli 1984, Van Horn and others 1995, Villard and others 1993). Predation or parasitism may occur at greater levels in habitat fragments, thus reducing the reproductive success of individuals residing there (see discussion in the "Edge Effects" section). This last point underscores the importance of productivity data for estimating the viability of populations; numerous studies have shown that the abundance and/or density of individuals or nests are not reliable indicators of habitat quality (Maurer 1986, Van Horne 1983, Vickery and others 1992, Zimmerman 1992) or of nest success (Johnson and Temple 1990, Vickery and others 1992, Zimmerman 1984). Simply because large numbers of individuals of a particular species are found in a habitat fragment does not necessarily mean that the fragment is capable of supporting that species over the long term.

In addition to these problems, the small size and isolated nature of fragmented populations makes them vulnerable to other random processes. Natural catastrophes such as floods or fires may eliminate the few remaining individuals of a small population purely by chance. Random environmental changes may prove disastrous for such a population; a prolonged drought, for example, might lead to the extinction of a population of pupfish when the spring that they live in dries up. The genetic structure of populations may be profoundly affected by isolation. The interruption of gene flow among individuals in subpopulations may result in increased genetic drift, population bottlenecks, and inbreeding, all of which could lead to the fixation of deleterious alleles and decreased genetic diversity (Falconer 1981, Lerner 1954, Ralls and Ballou 1983, Wright 1977). Any of these factors—catastrophes, environmental variations, altered gene flow—as well as changes in demographic structure, might potentially lead to the extinction of a small, isolated population (Shaffer 1981). In reality, however, it is more likely a synergistic interaction between two or more of these factors that ultimately leads to the extinction of such populations in a process that is called an "extinction vortex" (Gilpin and Soulé 1986).

Of all these processes, the genetic consequences of isolation and interrupted gene flow in particular have received a great deal of attention by conservation biologists. In general, population bottlenecks, inbreeding, and the loss of genetic diversity are all believed to have a negative impact on the fitness of individuals through decreased fecundity and survivorship, a condition known as inbreeding depression (Falconer 1981, Lerner 1954, Ralls and Ballou 1983). The negative effects of inbreeding depression have been witnessed primarily in captive animal populations, but such impacts have also been documented in small wild populations that have become isolated, such as the lions of the Ngorongoro Crater in Africa that exhibit high levels of sperm abnormalities and low reproductive success (Packer and others 1991). However, there are also examples of small, isolated populations that have either retained relatively high levels of genetic variability (for example, one-horned rhinos; Dinerstein and McCracken 1990) and/or have survived severe population bottlenecks with no apparent problems stemming from inbreeding (for example, elephant seals; Bonnell and Selander 1974). Plants in particular seem to be resistant to the negative effects of inbreeding, most likely an adaptation to the limited dispersal ability of many species and self-fertilization (Barrett and Kohn 1991), although reductions in genetic diversity have been correlated with decreased fecundity in some plants found in isolated patches (Baur and Erhardt 1995). Although the impacts may be variable, the changes in gene flow and reduced number of individuals resulting from habitat fragmentation have the potential to significantly impact both the demographic and genetic structure of remnant populations (Fahrig and Merriam 1994). In the short-term, such alterations may be reflected in the reduced reproductive capacity and survivorship of individuals, possibly leading to localized extinctions for some species. On an evolutionary time scale, the reduced genetic variability stemming from processes such as fragmentation impedes the ability of individuals to respond to selection pressures, possibly leading to the extinction of the species (Frankel and Soulé 1981).

Corridors and Connectivity in Fragmented Landscapes

A central tenet of applied conservation biology has been the maintenance or reconstruction of habitat corridors to achieve connectivity between fragments (Meffe and others 1997, Preston 1962, Saunders and others 1991, Shafer 1990). In theory such corridors would essentially reconnect an otherwise fragmented landscape, facilitating the movement of individuals between patches, enabling continued gene flow, maintaining metapopulation dynamics, and reducing mortality of animals attempting to disperse through hostile terrain. In practice such corridors have produced mixed results. Most experiments have found that corridors do enhance movement, although this has been true primarily for small, less-mobile animals (Debinski and Holt 2000; but see Haas 1995). Small mammals such as chipmunks may use treelines to successfully colonize wooded patches in a fragmented forest landscape (Henderson and others 1985), and a few species of invertebrates have been found to preferentially utilize corridors in grassland fragments (Collinge 1998, 2000). In arid grasslands, movements of beetles in the genus *Eleodes* are strongly affected by vegetation structure (Crist and others 1992, Wiens and Milne 1989), and these beetles are one of the few species known to use corridors (Collinge 2000). Neotropical migratory birds, on the other hand, decrease in habitat fragments regardless of the degree of connectivity (Debinski and

Holt 2000). In the case of highly mobile organisms, the ability to move between fragments is most likely not the factor limiting populations. Corridors also do not appear to assist organisms with limited mobility; the habitat corridors provided by roadsides or ditches did not prove effective for colonization of habitat fragments by plants with short-range dispersal mechanisms (Van Dorp and others 1997). In general, many questions remain regarding the efficacy of corridors, particularly as they can serve not only for dispersal but also as conduits for predators, parasites, and disease transmission (see Noss 1987).

Impacts of Fragmentation on Grassland Plants

Due to their small area requirements, plants are often proposed to be relatively immune to habitat fragmentation (Noss and Cooperrider 1994). Some short-term studies have found that small fragments support just as many plant species as large fragments and that rare species appear to persist in such small fragments (Simberloff and Gottelli 1984). However, studies of fragmented grasslands in Europe using historical records have documented high levels of extinctions of plants when followed over the long term (Fischer and Stöcklin 1997). Although the total numbers of species at each site were essentially unchanged, there were significant increases in habitat generalists, while habitat specialists that formerly occupied the sites had disappeared. The interruption of plant-pollinator interactions, leading to reduced viability of plant populations in fragments, may be one factor contributing to local extinctions. Isolation of patches has been found to diminish both the abundance and species richness of bees, butterflies, and other pollinators (Debinski and Holt 2000, Jennersten 1988, Steffan-Dewenter and Tscharntke 1999). Fewer visits by pollinators can lead to reduced fecundity, viability, and decreased genetic diversity in plants isolated in habitat remnants (Aizen and Feinsinger 1994, Baur and Erhardt 1995, Jennersten 1988). The small population size and isolation of plants in habitat fragments may also result in inbreeding depression and loss of genetic diversity through founder effects, random genetic drift, and inbreeding (Rajmann and others 1994, Templeton and others 1990, van Treuren and others 1991, Young and others 1996, 1999). In some cases genetic variation continues to remain high in isolated plant populations, although even in these cases rare alleles may be lost from smaller fragments (Young and others 1999). In terms of negative genetic impacts, wind-pollinated species appear to be much more resilient to the effects of fragmentation (Fore and others 1992, Young and others 1993).

Summary of Habitat Fragmentation Effects

Not all organisms respond in the same manner to habitat fragmentation; persistence in habitat fragments, impacts on dispersal abilities, and use of corridors are highly species specific (Debinski and Holt 2000). Nor are all habitat fragments created equal; the size of the fragment, shape, amount of edge, and nature of the surrounding matrix will all influence the nature of the impacts on the individuals residing in the remnant patch (Helzer and Jelinski 1999). Taken as a whole, however, the majority of evidence from studies of habitat fragmentation indicates that the loss and isolation of natural habitats pose a strong threat to both regional and global biodiversity. Fragmentation greatly increases the risk of extinction for the native species of the original habitat through several mechanisms, including:

- Loss of habitat area for interior species.
- Barriers to dispersal, colonization, and maintenance of metapopulation dynamics.
- Random alteration of demographic and genetic structure resulting from isolation and small population size.
- "Edge effects" such as increased predation and parasitism, and invasion by exotic species or habitat generalists.
- Interference with biotic relationships, such as plant-pollinator interactions.
- Alteration of the physical environment, ecological processes, and natural disturbance regimes.

Strategies for counteracting the effects of habitat fragmentation include:

- Preventing or minimizing further fragmentation.
- Managing lands to restore natural disturbance regimes.
- Maintaining or restoring large natural areas to act as avenues for dispersal and genetic mixing of populations.
- Restoring habitat to increase the size of remaining habitat patches or buffer existing patches (Meffe and others 1997, Shaffer 1990).

Some basic design principles to counteract the effects of habitat fragmentation are summarized in figure 4-3. In an already fragmented landscape, it may be necessary to strive for protection of the largest possible area through strategies such as creating corridors by restoring connections between natural areas (Gatewood 1998). However, the importance of preserving extensive landscapes in order to maintain ecosystem processes cannot be overemphasized; wherever possible, such a strategy should be the first line of defense.

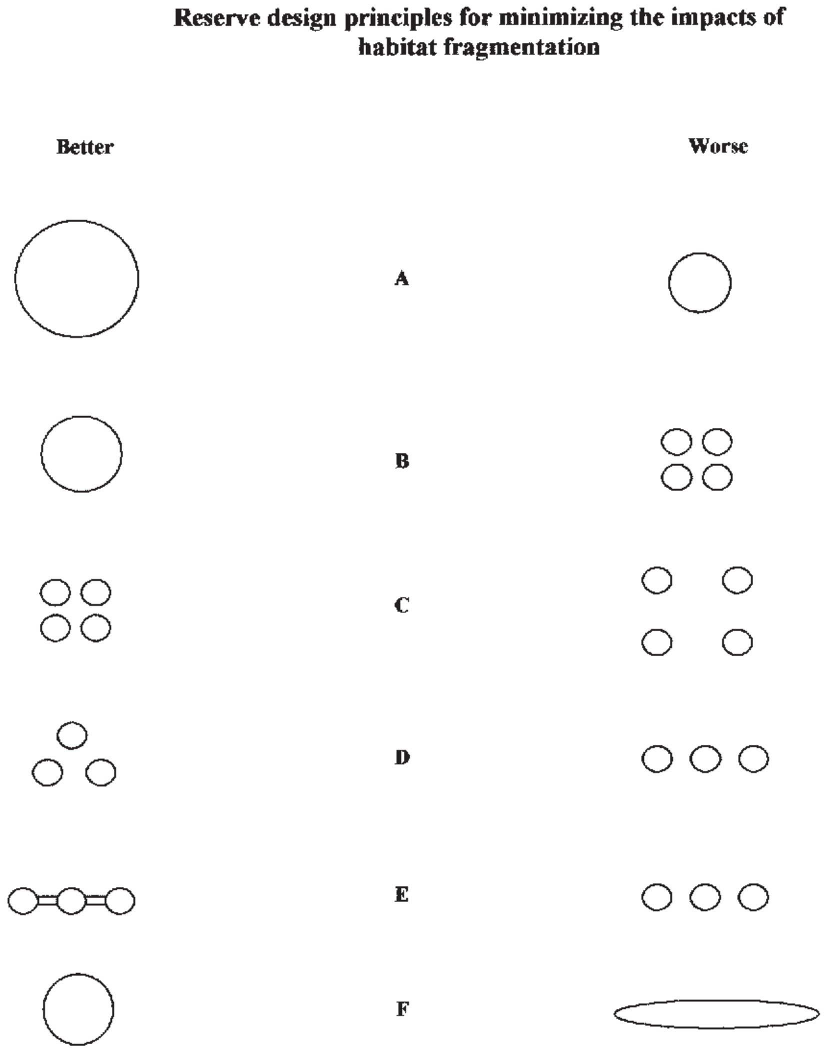

Figure 4-3. All areas of habitat are circular to minimize the amount of edge relative to area. These principles are based upon the theory that, in general: (A) large reserves are better than small to maximize number of species and individuals supported; (B) one large fragment is better than several small fragments, even if total area is equal; (C) fragments close together are better than fragments more widely separated to enhance movement between fragments; (D) fragments clustered are better than fragments strung out in a linear fashion to allow for easier movement between fragments; (E) fragments connected by corridors are better than fragments that are entirely disjunct to enhance movement between fragments; (F) fragments that are circular are better than fragments that are long and thin to minimize edge effects. Based on Diamond (1975). As discussed in the text, the basic theories behind these principles are not without controversy, yet they remain the primary foundation upon which contemporary conservation design is based.

The vast majority of information that we have on the consequences of habitat fragmentation is based on studies of forested ecosystems or tallgrass prairie. One of the greatest needs for research in the grasslands of the Southwest is the need to study the impacts of fragmentation in these arid grassland systems.

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Chapter 5:
Historic and Contemporary Land Use in Southwestern Grassland Ecosystems

Introduction

This chapter encompasses the lands of the Southwest as defined by Region 3 of the USDA Forest Service (USFS): Arizona, New Mexico, and portions of western Oklahoma and the Texas Panhandle. I examine human use and modification of the grasslands/rangelands of this region, with an emphasis on those areas managed by the Forest Service. Because the majority of publications serving as sources for this chapter use both "rangelands" and "grasslands" when referring to a variety of different grassland and rangeland vegetation types, this section does not distinguish between the terms. An exhaustive examination of all human uses and related topics and issues in Southwestern rangelands is well beyond the scope of this discussion; thus, a selective review of historical and contemporary topics is presented. The section begins with a review of continuous regional land use and modification from pre-Euro-American settlement (that is, American Indian times) to the present to serve as a background for understanding current land uses and land-use-related problems and issues. I then examine contemporary rangeland condition and the major human uses and activities affecting these lands, focusing on domestic livestock grazing, mineral extraction, and recreation. I conclude the chapter with a brief discussion of continuing and future trends in Southwestern rangelands and rangeland management.

American Indian (Pre-Euro-American Contact) Land Use

Paleoindian Period

The earliest undisputed evidence of human occupation of the Southwest began between 11,650 and 10,250 years ago. Archeological complexes from this period are referred to as Paleoindian and represent the remains of ancestral Native Americans from late Pleistocene times. These people were hunters and gatherers using both plant and animal food sources, under climatic conditions and vegetational patterns that were markedly different from those of today. Areas from which data have been obtained indicate that climatic conditions were generally wetter than present with less seasonal variation resulting in relatively mild winters and cool summers (discussed in Cordell 1997: 67-100).

Owing to the paucity of archeological remains from this period and to the fact that such remains are often deeply buried, interpretations of Paleoindian economic practices and resource use are sketchier and less well documented than those from later times. According to the most commonly accepted scenarios, relatively small groups of highly mobile hunters and gatherers exploited a variety of plants and animals, including Pleistocene megafauna such as mammoth (*Mammuthus columbi*), at least during the early portion of the period. In later Paleoindian times there was considerable reliance on bison (*Bison* spp.), including a now extinct larger form (*Bison antiquus*), as well as on modern fauna. In the eastern Southern Plains portions of the study area extending from present-day eastern New Mexico into present-day Texas and Oklahoma, human groups relied heavily on hunting bison supplemented by smaller game and plant foods. Studies have shown that bison numbers and range expanded and contracted throughout the Holocene with expansion and contraction of the grasslands, tied to climatic fluctuations and available moisture. Because the Southern Plains and Southwest are generally drier than more northern areas of the Plains, reliance on bison hunting may not have been a productive strategy at all times during this period. Sufficient numbers of animals may not have been present during periods of grassland contraction (Cordell 1997: 95; Reher 1977).

On the western margins of the Plains and in the foothills and mountains adjacent to the Plains, there was apparently a fairly equal reliance on hunting and gathering. The role and importance of bison hunting in these areas fluctuated to an even greater degree as bison range expanded and contracted. Mountain sheep (*Ovis canadensis*) and mule deer (*Odocoileus hemionus*) were also present and hunted in about equal numbers to bison. In the more western portions of the region, human groups were dependent upon plant foods and relatively small, nonmigratory game (Cordell 1997: 90-100).

Especially on the Plains and in the areas adjacent to the Plains, human population growth was tied to the availability of bison and the grasslands to support them. Thus, to understand important questions concerning the timing and effects of increasing regional populations, additional paleoenvironmental reconstruction studies are needed to determine the expansion, contraction, and spread of grasslands during this period. Little is also known concerning human manipulation of the physical environment in Paleoindian times (Cordell 1997: 90-100), although use of fire has been mentioned as a means of moving game during hunting drives (Bahre 1995: 232-235). Clearly, additional archeological and archeoenvironmental research is needed for this period to clarify Paleoindinan resource use and environmental manipulation strategies.

Archaic Period

Around 7,500 years ago (5500 B.C.) the Paleoindian period gave way to what is referred to as the Archaic, lasting until about A.D. 200. From about A.D. 200 until Euro-American contact, agricultural groups of varying intensity occupied the diverse environments of the Southwest. During this period from approximately A.D. 200 until the mid-1500s, Southwestern climate and vegetation assumed essentially modern forms (Cordell 1997: 101-102). There was a gradual climatic warming and drying from around 7,000 to 5,000 years ago (5000 to 3000 B.C.). This warm and dry period was followed by a gradual increase in moisture with a corresponding increase in woodland and forest vegetation (Hall 1985, Periman and Kelly 2000). A cooler period followed from approximately A.D. 1450 until 1900, which is referred to as the Little Ice Age (Kreutz and others 1997, Periman and Kelly 2000). As a whole, this entire period has been marked by climatic variability with both broad-scale and periodic climate fluctuations (Periman and Kelly 2000).

During the Archaic, human groups continued to rely on hunting and gathering but with a greater focus on food plants and locally available resources. Hunted game were all modern forms (Cordell 1997: 101-102). Although subsistence patterns varied throughout the Southwest, some generalizations can be made. In many areas, people relied on uplands with high topographic and vegetational diversity, which allowed use of a wider range of resources in a smaller geographic area (Gunnerson 1987, Tainter and Tainter 1996). Grasslands were also favored resource procurement locales of Archaic foragers.

Archeological information from various locations, such as from sand dune sites in the San Juan Basin and from grassland sites in southeastern Arizona, indicates heavy reliance on the seeds of weedy plants and grasses. These include Indian ricegrass (*Oryzopsis hymenoides*), dropseed (*Sporobolus* spp.), goosefoot (*Chenopodium* spp.), pigweed (*Amaranthus* spp.), mustard (*Descurainia* spp.), tickseed (*Corispermum* spp.), and mallow (*Sphaeralcea* spp.). Both goosefoot and pigweed favor disturbed ground and may have thrived and/or been semicultivated in areas where humans camped repeatedly (Bahre 1995: 233; Cordell 1997: 120-121). Small, medium, and large-sized game were taken. Faunal remains from cave sites in the highlands along the central New Mexico-Arizona border include bison (*Bison bison*), pronghorn antelope (*Antilocapra americana*), mountain sheep (*Ovis canadensis*), and deer (*Odocoileus* spp.). Rabbit (*Leporidae*) and deer were found at Ventana Cave in the low Sonoran Desert of southern Arizona (Cordell 1997: 122).

Southern Plains groups were apparently not heavily dependent upon bison during the Early Archaic but were concentrating on deer, small mammals, and plants. By later portions of the Archaic, however, bison increase in the area's archeological sites (Cojeen 2000, Drass and Turner 1989: 20-22; Simpson and others 1998). Bison may become a more abundant and predictable resource in the later Archaic owing to improved shortgrass range brought about by wetter conditions (Cojeen 2000, Creel and others 1990).

Ranging from about 1500 B.C. to 1000 B.C. in the low deserts, central mountains, and northern plateaus of the Southwest, human groups began to cultivate corn (*Zea mays*). However, they remained hunters and foragers with a mobile lifestyle for many hundreds of years thereafter, using corn and later beans (*Phaseolus* spp.) and squash (*Cucurbita* spp.) to varying degrees in their diets. Even with the considerable emphasis on agriculture found among many Southwestern groups immediately prior to Euro-American contact, wild foods always retained an important role in diet and survival (Cordell 1997: 147-151).

Puebloan and Other Agricultural Groups

Beginning around A.D. 200 and continuing until European contact and colonization in the mid-to-late 16th century, people became more sedentary and more committed to the production of agricultural crops. The widespread appearance of permanent dwellings, increasingly larger settlements, and ceramics are indicators of these trends. Subsistence patterns varied throughout the region depending upon both agricultural productivity and locally available wild resources (Cordell 1997: 221-224). Settlement locations also varied, most probably conditioned by both economic and defensive factors.

As dependence on agriculture increased, settlement location moved away from high eminences and upland areas to alluvial terraces and benches associated with major rivers and their tributaries, and to arroyos and intermittent watercourses in the Four Corners area, southern New Mexico, and southeastern Arizona. Agricultural occupations in south-central Arizona, however, were located on open flatlands without apparent defensive considerations from the beginning (Cordell 1997: 241). The Southwestern farming groups are ancestral to contemporary Native Americans in the region and included the Anasazi or Ancestral Puebloan of the Four Corners and Rio Grande areas; the Mogollon of southern New Mexico, southeastern Arizona, and northern Mexico; the Hohokam of south-central Arizona; and the Patayan of western Arizona (discussed in Periman and Kelly 2000).

In northeastern New Mexico, western Oklahoma, and the Texas Panhandle, groups of the Plains Village Tradition practiced hunting and gathering and grew corn along the creeks and watercourses of the region. Although agriculturally based, these groups are considered to have had a greater reliance upon hunting deer and bison than the Puebloan groups to the west (Fredine and others 1999). A mixed dependence on agriculture and bison hunting continued until around A.D. 1500 when agricultural groups were pushed south and west by incursions of nomadic Plains Apachean groups who were both raiding and trading with the Pueblos bordering the Plains (Fredine and others 1999, Glassow 1972). Migration of Plains agricultural groups as a result of increasingly drier climates has also been suggested, as has coalescing or amalgamating into preexisting nomadic cultures such as the Kiowa Apache (Baugh 1984, Cojeen 2000, Hughes 1991: 43; Simpson and others 1998). Continuity between prehistoric and protohistoric groups in the area continues to be a matter of debate (Cojeen 2000, Drass and Turner 1989: 28; Hughes 1991: 43).

As in earlier times, grasslands were used as sources of wild plant and animal foods. Bahre (1995: 234) concludes that although there is not a great deal of information concerning American Indian use of southeastern Arizona's grasslands, groups were known to collect vegetal foods from arborescent and succulent taxa in the Desert Grasslands and to harvest grass seeds in the Plains Grasslands (Huckell 1995). Especially favored were the seeds of big sacaton (*Sporobolus wrightii*) and browntop panicgrass (*Brachiaria fasciculate*) (Bahre 1995: 234; Doebley 1984). In describing animals hunted by Puebloan and other groups of the Middle Rio Grande, Scurlock indicates that bands of hunters journeyed to the Llano Estacado (plains of eastern New Mexico and western Texas) in the fall to hunt bison for both meat and hides (discussed in Scurlock 1998: 98).

In addition, bison ranged seasonally into the San Augustín Plains and the grasslands of northeastern Arizona in late prehistoric times. A herd was reported in the Chama Valley as late as 1690 (Callenbach 1996: 17-18; Scurlock 1998: 209). There was also considerable trade in raw materials and food stuffs among regional groups occupying the Puebloan areas and those areas farther to the east on the Plains proper (Scurlock 1998: 104-105). As mentioned previously, the resident Southern Plains groups focused heavily on both hunted and gathered grassland resources (Cojeen 2000, Fredine and others 1999, Simpson and others 1998).

Understanding Pre-Euro-American Contact Land Use

Understanding the role, extent, and importance of American Indian use and manipulation of Southwestern environments prior to Euro-American contact is critical for understanding contemporary land conditions, as well as the past "reference conditions" called for in studies of ecosystem management and restoration. Tainter and Tainter (1996: 28-29) recommend a combination of information derived from contemporary environmental sciences and social sciences with information derived from the historical sciences to elucidate past land use and management practices. Both ethnohistoric and archeological research into past environmental conditions and land use practices can provide valuable information. Periman and Kelly (2000: 27-28) discuss the role of archeological data in describing the impact of

Native American use on certain Southwestern riparian ecosystems in terms of reductions in tree cover from cutting roof timbers and firewood and from tree clearance for agricultural fields (Le Blanc 1985). Ongoing studies of changing landscape use in the Rio del Oso valley of northern New Mexico are providing both new information and new methodologies for determining past use and manipulation of riparian, woodland, and grassland ecosystems (Periman 2001).

These studies will make a timely contribution as information on prehistoric human use and manipulation of grassland ecosystems is sorely needed. Much of the available information (and more is needed) focuses on the use of fire by aboriginal peoples to manipulate their environment. According to Scurlock (1998: 201, 268-269), Native American groups in the Rio Grande Basin intentionally burned the grasslands periodically, which, along with lightning-caused fires, may have killed encroaching half shrubs and shrubs and stimulated vigorous growth of grasses. Although Baisan and Swetnam (1997) generally conclude that natural processes account for fire frequencies at most of their 60 study sites in Arizona and New Mexico, they suggest that fire frequencies at sites in the Manzanita and Sandia Mountains of the Rio Grande Valley during pre-Euro-American settlement times may have been influenced by human-caused ignitions. High fire frequencies prior to Euro-American settlement in these areas where lightning occurrence is low led them to the conclusion that American Indian groups were igniting fires for resource manipulation (Baisan and Swetnam 1997).

In discussions of southeastern Arizona, Bahre (1995: 234-235) concludes that the extent to which American Indian groups influenced grassland ecology is unknown but that the accidental or intentional introduction of fire to the grasslands may have contributed to their largely brush-free state at the time of Euro-American contact. However, the full role of fire in maintaining desert grasslands is unknown (Bahre 1991: 138-141; Dick-Peddie 1993: 106-107).

American Indians used fire to drive game, stimulate the growth of understory and food plants such as berries, clear areas for campsites and agriculture, and produce nutrient-rich ashes in fields (taken from a discussion in Scurlock 1998: 269). Indeed, Sullivan (1996: 145-156) argues that Western Puebloan groups on the Colorado Plateaus actively managed their habitats to increase production of wild resources such as Indian ricegrass, sunflower (*Helianthus* sp.), and goosefoot by controlled burning of the pinyon-juniper woodland. We look forward in the coming years to archeoenvironmental research that will help to clarify the past human role in maintenance and manipulation of pre-Euro-American contact North American ecosystems.

Contemporary American Indian Uses of National Forest Grassland Areas

Contemporary American Indian groups use Western grasslands, many of which occur on public land, in a variety of ways, including hunting, and gathering plant materials for food, medicines, and basketry. They also use these lands for grazing domesticated animals. The following brief review cannot do justice to the complex topic of Native American wild and domesticated plant use and its body of ethnobotanical literature. Especially important, detailed information on this topic can be found in the work and publications of Gary Nabhan (1985, 1989), founder of Native Seeds/Search. Nabhan has worked to gather and preserve seeds used in aboriginal agriculture and to document Native American gathering and farming practices in the Southwest in grasslands, as well as in the Sonoran Desert region.

In recent years, the Forest Service has collaborated with indigenous groups to learn from their traditional practices and to assist in assuring continued supplies of materials needed for economic pursuits and craft production. As examples, several forests in California have cooperated with Native American groups to identify and assure the continued propagation of plant materials such as the bear grass (*Nolina* spp.) used in basket making. Information on the use of bear grass and other grassland plants in the Southwest can be found in Bell and Castetter (1941).

The Hopi have worked with the Kaibab National Forest in northern Arizona to develop a comprehensive plan to manage pinyon-juniper woodlands and rangelands to protect cultural and spiritual values, while addressing concerns related to watersheds, soils, wildlife, recreation, range, and other resources (Thakali and Lesko 1998). The Apache, Navajo, and Hopi still collect plants from various meadows on the San Francisco Peaks, Coconino National Forest, Arizona, and the Hopi consider Bonito Park, by Sunset Crater, New Mexico, a Traditional Cultural Property (Pilles personal communication 2002).

The Tohono O'odham collect bear grass for basketry, and the San Carlos and White Mountain Apache gather medicinal herbs in grassland and mountain meadow areas of the Coronado National Forest in southern Arizona. The Mescalero Apache report collecting small amounts of coral beans (*Erythrina flabelliformis*) from the Chiricahua and Dragoon Mountains. The Coronado is currently working on draft memoranda of understanding with the Apache Tribes that would authorize collecting of small amounts of plants and acorns by Tribal members for personal use without permit. A potential conflict arises with the desire of some Tribal members to collect agave (*Agave* spp.) from the grasslands around Patagonia because agave is an important food source for the endangered long-nosed bat (*Leptonycteris sanborni*) in some areas (Farrell personal communication 2002). Discussions of the role and importance of agave as a food and fiber source to Native American groups are presented in Castetter and others (1938) and Gentry (1998), among others.

Prehistorically the Wichita and Affiliated Tribes occupied areas of western Oklahoma, northern Texas, and northeastern New Mexico that now comprise the Kiowa and Rita Blanca National Grasslands and the Black Kettle and McClellan Creek National Grasslands. In the 1700s and 1800s, horse-using groups such as the Cheyenne, Arapaho, Apache, Comanche, and Kiowa entered the area as a result of territorial dislocations caused by encroaching Anglo-American settlement. There is, however, little information concerning whether these groups still use the area for traditional practices, although they may consider it a traditional use area and/or important in their Tribal history (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000, Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999). Research to address these issues is planned for the near future (Benedict personal communication 2001).

Hispanic and Post-European Contact American Indian Land Use

The Spanish were the first Europeans to enter the Southwestern areas of present-day Texas, New Mexico, and Arizona beginning in the late 1520s with the shipwrecked Álvar Núñez Cabeza de Vaca. Over a period of about 6 years, Cabeza de Vaca and several companions made their way from the Texas coast through parts of New Mexico and probably Arizona to finally rejoin their countrymen on the west coast of Mexico (Stephens and Holmes 1989, Stout and Faulk 1974, Udall 1995, Weber 1992). Further expeditions followed in search of the fabled wealth reported by Cabeza de Vaca. In 1540, Francisco Vásquez de Coronado led an expedition that covered portions of New Mexico, Arizona, and the Southern Plains, probably passing through parts of the Texas Panhandle and into present-day western Kansas. However, the expedition route onto the Plains remains the subject of considerable debate (Bahre 1995, Bolton 1949, Fredine and others 1999, Trimble 1989, Wildeman and Brock 2000).

New Mexico

Significant Spanish influence began with colonization of the Rio Grande valley in 1598 by Juan de Oñate, initiating both the political and biological conquest of the region. Although the exact numbers are debated, Oñate brought with him soldiers, colonists, priests, and Mexican Indian servants, along with cattle, sheep, goats, and horses (Baxter 1987, Hammond and Rey 1953(1), Wildeman and Brock 2000). The settlers introduced domesticated plants and animals, as well as new technologies, which altered the flora, fauna, and landscape of the Southwest as has occurred throughout the New World (Crosby 1972, Melville 1994).

Throughout the 1600s, the Pueblo Indian populations of the Rio Grande declined as a result of diseases introduced by Euro-American contact, warfare, famine caused by a series of severe droughts, and destruction of food stores by raids from nomadic Indian groups. As the Native American population declined, the tribute and labor requirements of the Spanish colonists became more onerous. These conditions, along with forced relocations and intensive religious mission programs, led to the Pueblo Revolt of 1680. During this rebellion, the majority of Spanish were forced out of the Upper Rio Grande for 12 years. They returned in the period from 1692 through 1696 when Diego de Vargas initiated and completed the reconquest of New Mexico for the Spanish Crown (Simmons 1979).

Even though Hispano populations rose throughout the 1700s, the significant population declines of the Puebloan groups left a sufficient amount of land for both groups to farm and ranch along the rivers of the region. After the reconquest, the economic, political, and religious systems of New Mexico were significantly different from the earlier systems. The new generation of Spanish colonists were accomplished agriculturalists and stock raisers who worked their own land and maintained relatively cordial relations with the Pueblo Indian groups as both used the land in similar ways (Simmons 1979).

During both the Spanish Colonial and Mexican periods (1598 through 1846), land use and ownership throughout the Southwest were confirmed by land grants from the Spanish Crown or Mexican government. Although land grants were of several types, community grants to a group of settlers in common were of particular importance for later resource conflicts in the area (Eastman and others 1971, Harper and others 1943). Within community grants, settlers received individually owned building sites and agricultural plots of irrigated land near the ditch or stream. The irrigated plots were often quite small, averaging from 5 to 10 acres (Van Ness 1987). The villagers also used the grant grazing lands, timber lands, and pastures communally (Eastman and others 1971). Because kinsmen often worked their fields cooperatively and herded their animals together, they were able to manage on the small-sized, scattered agricultural plots.

Throughout the Colonial period, a subsistence economy based in small villages prevailed along the Rio Grande and its tributaries. Information on the community of Cañones (Kutsche and Van Ness 1981, Van Ness 1987) provides a good description of farming and ranching in the Hispano villages. Both animal and plant production were critical for the success of the mixed farming system, with sheep and goats the most important stock for food. Cattle were used to plow fields, thresh grain, transport produce, and manure fields. The community stock were individually owned but cooperatively grazed. They were moved into the higher elevation pastures during the spring and summer and returned to the village after the harvest to graze and manure the stubble fields (Kutsche and Van Ness 1981, Van Ness 1987).

Livestock numbers were modest for the first two centuries after colonization. Raids by nomadic Apache, Navajo, Ute, and Comanche limited range expansion, commerce, and trade (Clark 1987: 19-23; Van Ness 1987). Thus, economic production was for subsistence, not for competition in a commercial market. Sheep were more numerous than cattle and horses in the early years, primarily because of sale and loss of the latter to surrounding nomadic Indian groups (Gonzales 1969). By the early 1800s, the number of sheep began to increase as the Spanish population expanded eastward onto the plains around present-day Las Vegas, across the Sandia and Manzano Mountains, and westward from the Rio Grande Valley. This movement resulted from growing human and animal populations, a decline in raiding by the nomadic groups, and an expanded trade in wool and sheep during the Mexican period (1821 to 1846).

Sheep were the mainstay of the livestock economy of New Mexico until after U.S. takeover in 1848 (Beck 1962). Scurlock (1998: 116-117) reports 1 million sheep in New Mexico during the 1820s rising to 3 million by the mid-1800s with a reduction to approximately 377,000 by 1850. The reduction occurred as a result of Apache and Navajo raids and losses from drought, blizzards, and predators. During the first half of the 1800s, New Mexico supplied sheep to the West and to extensive mining operations in Mexico. Beck (1962) and Schickedanz (1980) state that annual sheep drives to Mexico often averaged 200,000 head between 1815 and 1830. Baxter (1987: 90-95) offers more conservative figures, listing some 240,000 sheep in the Albuquerque and Santa Fe areas in 1827, with a total of approximately 200,000 driven south to Mexico between 1835 and 1840. For example, about 80,000 were exported in 1835. In spite of the contradictory figures, sheep were an extremely important part of the economy throughout the period.

Sheep numbers also increased in the Navajo areas of northwestern New Mexico during the 1800s, so much so that some Navajos were described as wealthy by the time of the Mexican-American War (Brugge and Gerow 2000: 449-451). Scurlock (1998: 117) suggests that rising sheep numbers and Navajo practices of grazing outward from the hogan during the day and returning the animals to associated corrals at night may have initiated the first regional overgrazing west and north of the Puebloan and Hispanic settlements.

Herding sheep near Los Ojos, NM, in the 1990s. (Photo by Anne R. Baldwin)

Although concentrations of sheep and cattle near Hispano and Puebloan settlements created areas of resource overuse during Spanish Colonial times, herds were generally small, and there was abundant land (Baxter 1987: 23-24; Rothman 1989: 196-197; Scurlock 1995a: 15). Thus, relatively small populations of subsistence farmers successfully used the region's resources over the long period of Spanish control (Raish 2000a: 495). Overuse of favored areas intensified during the Mexican period as commercial sheep production increased (Scurlock 1995a: 15). By the end of the Mexican era, range deterioration was noted by American military personnel in some areas of the Rio Grande Valley, the Rio Puerco watershed, and on the Navajo lands higher up the Rio Puerco (discussed in Elson 1992, Scurlock 1998: 117). Nonetheless, the majority of farms and ranches remained small and subsistence oriented prior to U.S. conquest. On several visits to New Mexico in the 1830s, Josiah Gregg described the irrigation systems and subsistence orientation of agriculture in the small Hispano settlements, among his many other observations on the life and people of the area. He especially noted the fertility of the bottomlands in contrast to the barren condition of the unirrigated uplands (Gregg 1954: 104; Wozniak 1995: 34).

Arizona

To the west, Father Eusebio Kino is credited with establishing missions and bringing domesticated livestock and crops into southern Arizona beginning in the later 1600s (Bahre 1995: 235; Bolton 1919, Hadley and Sheridan 1995: 7-11), although domesticates may have reached the area earlier through trade with Spanish colonists to the east in the Rio Grande Valley (Bahre 1995: 235-236; Bolton 1952). At about this time, Apaches also entered the area warring with resident Piman groups and later with the Spanish. By 1740 Spanish settlers had begun ranching in the area, and by 1775 Tucson had been established.

Decreased hostilities with the Apaches after 1786 led more Hispanic ranchers and miners to enter the region, but their numbers remained small. According to Bahre (1995: 236-237), their activities appeared to have had little impact on the grasslands. Although livestock may have degraded rangelands adjacent to settlements as in the Rio Grande Valley, their numbers were small and their effect on regional grasslands was probably minimal. In 1804, the two largest settlements, Tucson and Tubac, reported a total of 4,500 cattle and 7,600 sheep. In addition, Tucson reported 1,200 horses (Bahre 1995: 236-237; McCarty 1976).

Although land grants have never played the role in rangeland use and management in Arizona that they have in New Mexico, there were a few Spanish land grants in southern Arizona, and Mexican stock-raising land grants were established in the grasslands of the San Pedro, San Rafael, and Santa Cruz Valleys of southeastern Arizona. Large numbers of cattle were reported on these ranches, but increased Apache raiding during the Mexican period drove off the majority of the ranchers by the mid-1800s (Bahre 1995: 237-240; Mattison 1946, Officer 1987, Ruyle and others 2000, discussed in Wildeman and Brock 2000: 5). Considerable numbers of livestock were also reported in the area of the Hopi villages in northeastern Arizona with one having 30,000 sheep in 1776 (Schickedanz 1980). Overall, both human populations and livestock numbers were low prior to the American period. According to Bahre (1995: 240), both Spanish and Mexican populations were concentrated in the upper Santa Cruz Valley, and the most significant human impact on the grasslands was probably the establishment of the large-scale Mexican ranches.

Bahre (1995: 238-240) provides a good listing of the several American military and scientific expeditions that passed through the region in the mid-1800s, and provides descriptions of the grasslands. He notes that many of them mentioned wild cattle on the abandoned Mexican land grants, but none provided written evidence of overgrazing. He says that the numbers of animals and their effects are not known and further states: "No doubt some grasslands were severely affected, but, given the descriptions of the lush grasslands in southeastern Arizona in the 1850s and 1860s, ranching during the Mexican occupation seems to have caused little long-lasting and extensive disturbance" (Bahre 1995: 240). Many Spanish and Mexican archival records remain to be examined for information on the southeastern Arizona grasslands.

Southern Plains

The Early Historic period on the Plains of eastern New Mexico was also characterized by continuing conflicts between nomadic American Indian groups and Hispanic settlers. Highly mobile Comanche and Apache groups following the bison herds raided and traded with more sedentary Pueblos and Hispanic villages to the west. Actual Spanish settlement in the area began in 1794 with the establishment of San Miguel del Bado, the first Hispanic village east of Pecos Pueblo. Throughout the Spanish and Mexican periods, various attempts were made to settle and use the Plains grasslands, but these were generally not long lived owing to raiding by the nomadic groups (Baxter 1987, Fredine and others1999, Sebastian and Levine 1989a,b).

Farther to the east on the Plains of western Oklahoma, Spanish explorers out of New Mexico encountered nomadic hunters and traders, as well as the ancestors of the historic Wichita with whom they fought a battle on the Red River in 1759 (Hays and others 1989, Hofman 1989, Simpson and others 1998). By 1800 the area was controlled by France and became part of the United States with the Louisiana Purchase in 1803 (except for the Oklahoma Panhandle, which became part of Oklahoma Territory in 1890). The land was designated Indian Territory in 1804 (Burgess and others 1963, Cojeen 2000).

Other Resource Use

Other resource use activities during the Spanish and Mexican periods, such as trapping, mining, and logging, played less of a role in the grasslands than in other areas of the Southwest. Extensive beaver (*Castor canadensis*) trapping, first in New Mexico and then extending into Arizona, reached its peak in the 1820s and concluded by the late 1830s with the decline in popularity of the beaver hat. The large numbers of animals taken by the trappers caused severe reductions and extirpation of some local beaver populations leading to habitat degradation and decreasing water retention in streams and valleys (Scurlock 1998: 119-121, 155-158; Wildeman and Brock 2000: 5-6, 11).

Small-scale mining occurred during the Spanish period in both Arizona and New Mexico with the first large-scale mining venture in New Mexico beginning with the copper mines at Santa Rita in 1804 (Gregg 1954, Scurlock 1998: 118-119; Wildeman and Brock 2000: 6). The impacts of mining did not become significant until the American period, however. Extraction of mineral resources was slow in the area owing to the nature of the ores, the lack of transportation, and raiding by the Navajo, Apache, Comanche, and Ute (Hadley and Sheridan 1995: 46-47; Paul 1963, Wildeman and Brock 2000: 6).

Although large-scale commercial logging did not occur during the Spanish and Mexican periods, there was considerable use of pinyon (*Pinus edulis*), juniper (*Juniperus* spp.), ponderosa pine (*Pinus ponderosa*), and fir (*Abies* spp.) for construction and for fuel wood. Pinyon and juniper were the preferred fuel woods for both Hispanos and Native Americans leading to intensive exploitation of the woodlands around settlements. Scurlock (1998: 118) states that as more land grants were established in the plains and mesas of eastern New Mexico, local residents packed fuel wood back to population centers in the Rio Grande valley for sale (discussed in Scurlock 1998: 117-118). This practice may have had considerable effect on local vegetation. Research in southwestern New Mexico has shown that precontact Native American groups profoundly affected local vegetation, greatly reducing tree cover in riparian and other areas. Areas were cleared for agriculture, and wood was routinely harvested for fuel and construction timbers. As an example, before A.D. 1000 cottonwood was a common source of fuel wood and construction materials. This source of wood became depleted, however, and wood charcoal remains show that groups were being forced to go farther from communities and use upland trees as substitutes because cottonwoods were no longer available (Le Blanc 1985, Periman and Kelly 2000).

Environmental Manipulation

As in earlier times, both the American Indian and Hispano populations of the Southwest manipulated the environment to enhance production of desired resources. Fire was often the management tool of choice in these endeavors. In New Mexico, Hispano ranchers and farmers reportedly burned forests to create grazing lands and drive game. They also burned livestock pastures to stimulate new grass growth and burned sheep rangelands to kill encroaching woody species (Allen 1984, Scurlock 1998: 269).

Wildfires were relatively common in the grasslands of southeastern Arizona during Spanish and Mexican times, based on historical records and fire scar data (Bahre 1985, 1991, Dobyns 1981, Pyne 1982). Bahre (1995: 240) suggests that American Indian groups, mainly the Apache, were responsible for the burning but also states that the incidence of lightning-caused fires is so high in the area that the cause of the burning is difficult to determine. Continued research, both archival and archeological, is needed to understand the incidence and scope of these practices and their full effects on both past and present landscapes.

Anglo-American Land Use

The majority of the lands in the study area became part of the United States at the conclusion of the Mexican-American War (1846 through 1848). Western Oklahoma and the extreme southern portions of Arizona and New Mexico were acquired with the Louisiana (1803) and Gadsden (1853) Purchases, respectively. The Oklahoma Panhandle and the eastern portion of New Mexico were part of disputed lands claimed by both Texas and Mexico. With the end of the war, these lands went to Texas and were turned over to the Federal government in 1850.

Changing Patterns of Land Ownership and Use

Annexation of the region ultimately led to changes in both landownership and patterns of range use especially in New Mexico. Differences in American and Spanish land laws combined with unscrupulous land speculation resulted in the loss of over 80 percent of the Spanish and Mexican land grants by their original owners (Eastman and others 1971, Harper and others 1943, Westphall 1965). Under the Treaty of Guadalupe Hidalgo, the United States was supposed to recognize and respect the property rights of the landowners of the region.

To obtain sound titles according to U.S. law, however, land grantees had to petition for title confirmation, at first through the Surveyor General to the Congress and after 1891 to the Court of Private Land Claims (Eastman and others 1971, Eastman and Gray 1987). To accomplish this, claimants usually had to hire an attorney, file their claim, and obtain necessary supporting documents. Eastman (1991: 103) notes, "...landholders were turned into claimants who had to incur a substantial expense to have their property respected." Money was scarce in the subsistence economy of New Mexico, so many claimants signed over portions of their land as payment for legal fees. Because of this, even successful claimants lost land since the legal fees often amounted to from one-third to one-half of the land involved (Eastman 1991: 103). Additionally, many land claims were rejected; about 24 percent of the acres claimed in New Mexico were confirmed compared to about 73 percent in California (Ebright 1987: 33). Lands from unconfirmed claims became part of the public domain.

The Surveyor General and the Court of Private Land Claims did not confirm grants for a variety of reasons. Boundaries were sometimes vague, original titles may have been lost, and communal ownership of pasture and woodlands did not conform to 19th century American ideas of private ownership (Eastman and others 1971). The court often confirmed house lands and irrigated plots but did not confirm community pastures, surrounding rangelands, and woodlands that had always provided the Hispano villagers with their primary grazing and fuel wood resources. Villagers also lost confirmed land because they could not pay property taxes under the American system of monetary tax payments.

There was also during this time considerable unscrupulous land speculation, which took advantage of many Hispanic farmers who did not speak English and did not understand the American legal system (de Buys 1985 personal communiction). Large parcels of land were bought and sold by outside investors, often depriving families of land they had occupied for generations. These losses were a source of bitterness throughout the region—a bitterness that has surfaced periodically since U.S. annexation and came to a head in the land grant movement of the1960s (Gardner 1970, Knowlton 1967, 1980, 1985, Rosenbaum 1981, Schlesinger 1971).

Large-Scale Commercial Livestock Ranching of the Mid to Late 1800s

Other forces of change were also at work during this time as the economy changed from subsistence to commercial orientation, and the population of the territory grew tremendously with in-migration from the United States. Many of these immigrants brought considerable amounts of capital for investment in large-scale operations and a 19th century, entrepreneurial resource utilization ethic focused on maximum harvest for maximum profit (Scurlock 1995b: 2). To add to the climate of growth and development, Federal and Territorial legislation promoted intensive use of the environment (McCormick 1865, Scurlock 1995b: 2). Detailed discussions of this legislation are beyond the scope of this chapter but can be found in Clark (1987), Scurlock (1998), and Donahue (1999), for example.

These growth and development factors, combined with expanding markets opened by the development of military bases and the entrance of the railroad into Arizona and New Mexico in the 1870s and early 1880s, as well as the final subjugation of the nomadic Indian groups, led to rapid increases in large, commercial ranching operations (Bailey 1994, 1998). These operations often displaced the older, subsistence farms and ranches of the Hispano villagers. Commercial farming, timbering, and mining also flourished (Harper and others 1943: 48; Rothman 1989: 192-204; Wildeman and Brock 2000: 17).

Prior to the Civil War, substantial numbers of sheep and cattle were driven to the gold mining camps of California from the Southwest. Between 1852 and 1860 more than 550,000 sheep were trailed from New Mexico (Bailey and Bailey 1986), with an estimated 15,000 to 20,000 head of cattle driven from Texas through southern Arizona in the peak year of 1854 (Hadley and Sheridan 1995, Wildeman and Brock 2000: 13). The Civil War disrupted ranching operations and put an end to these drives, as ranchers left to fight in the war and Native American raiding increased with soldiers occupied elsewhere. By the end of the war, there were extremely large herds of free-ranging cattle in Texas from ranches that had been abandoned during the war. There were markets for these animals both in the North and in the West with the reestablishment of military posts and American Indian reservations (Wildeman and Brock 2000: 15-16).

Thus began the period of cattle drives out of Texas lasting from 1866 to 1880. During this time an estimated 10 million to 12 million animals were driven out of Texas, often through New Mexico and Arizona. This period also saw the decimation of the Plains bison herds between 1868 and 1881 (Schickedanz 1980). The herds were slaughtered for both economic and political reasons. Their decimation caused intense retaliation by the Plains Indian Tribes, who were ultimately subjugated by the U.S. Army. What had been bison range became cattle range, leading to the westward spread of the livestock industry and its full development in the Southwest (Wildeman and Brock 2000: 16-17).

Although there are conflicting livestock numbers, the cattle industry in Arizona grew from an estimated 5,000

Corral on the road to Gallina, NM, 1993. (Photo by Alice M. McSweeney)

animals in 1870 to around 200,000 in 1872. By 1891 various authors report cattle numbers ranging from over 980,000 to 1.5 million (Antle 1992, Ferguson and Ferguson 1983, Peplow 1958, discussed in Wildeman and Brock 2000: 18), in addition to an estimated 700,000 sheep brought into the State from New Mexico and California. In 1870, some 17,000 sheep were reported for the State. This growth was fueled by an extremely large flow of capital into the area, primarily from nonlocal investors (Bailey and Bailey 1986, Wildeman and Brock 2000: 18).

The livestock industry in New Mexico grew in the same way, with the entry of large numbers of animals into the area fueled by financial investment from outside the region. Extremely large numbers of both sheep and cattle were put on the rangelands with more than 1.5 million sheep in 1870 rising to 4 million or 5 million in 1883. In addition to the sheep, there were an estimated 250,000 cattle (Bailey and Bailey 1986, Schickedanz 1980). Rangeland use in both Arizona and New Mexico reached its peak in the late 1880s to early 1890s with almost 9 million animal units (AU) in New Mexico and 4.5 million in Arizona (Bailey and Bailey 1986, Wildeman and Brock 2000: 19).

Land Degradation and Reform Legislation

The land could not sustain the large number of animals being grazed in the attempt to achieve maximum economic gain. In the Southwest, the cattle population crashed after severe drought in the summers of 1891 and 1892. Severe ecosystem degradation resulted from several interacting factors including overstocking of rangelands, decrease of herbaceous plant cover, drought, suppression of natural fires, and removal of beaver along streams by trapping earlier in the century (Tellman and others 1997, Wildeman and Brock 2000: 19-20). Heavy stocking of the rangelands stressed native grasses, leading to decreased cover and root depth. In addition, some of the Southwestern grasses may have had less resistance to grazing than plains grasses that evolved with grazing by bison (Hyder 1972, Loftin and others 2000). Fire suppression, which became a major factor early in the 20th century, in combination with reduced plant cover from grazing, allowed woody shrubs and plants with low grazing preference to increase across the landscape. The combination of drought and overgrazing led to soil cover loss from wind and water erosion (Wildeman and Brock 2000: 20).

An example from the Pajarito Plateau west of Santa Fe discussed in Raish (2000b: 287) can help to illustrate the environmental and social consequences of commercial development that resulted in overstocking and subsequent environmental damage. Throughout the 1800s, local Hispanic and Pueblo residents of the area used the plateau as common property. They brought small herds to the plateau for summer grazing, harvested timber for personal use and small-scale ventures, and planted occasional summer crops. They were not attempting to maximize profit by developing large herds because for much of the period there was limited access to commercial markets. The small size and noncommercial nature of these operations ensured that sufficient grass and fuel wood were available for those who needed them (Rothman 1989: 192-194).

During the livestock boom period in 1885, a commercial Texas cattle operation leased a large amount of land on the plateau and brought in 3,000 head of cattle. The area could probably support around 300, according to modern calculations. This large operation drove off the small-scale Hispanic and Puebloan operations that had used the area for many years, as the resources were simply not sufficient to provide for both commercial and subsistence economies simultaneously. The large Texas enterprise failed after the severe winter of 1886 to 1887, and the small, local operators returned with their herds. However, the considerable overstocking had caused long-lasting damage to the resources of the plateau (Rothman 1989: 200-204). Rothman (1989: 202) notes:

> In an arid marginal region, the impact of comprehensive overgrazing persists for generations. Ecological climax communities in arid areas like the Pajarito Plateau take hundreds of years to mature. Because its soils were fragile, thin, and highly erodible, the removal of first-growth cover by overstocking precluded the slow process of natural recovery.

This damage was increased by commercial timber operations that began after the failed cattle business. Small-scale timber and personal-use fuel wood harvest was replaced by clear-cutting on large areas of the plateau. The combined effects of the commercial cattle and timber enterprises damaged the native grasses and removed much of the old-growth timber. Destruction of the grass cover and the spread of less palatable plants affected the subsistence of the local people, forcing them to pasture their animals farther away and compete among themselves for increasingly poor range (Rothman 1989: 203-204). This small vignette was repeated throughout the Southwest with effects lasting to the present day.

To deal with such problems of land degradation and resource overexploitation throughout the West, the direction of Federal legislation changed from promotion of intensive resource use to promotion of resource conservation. Scurlock (1998: 331-384) provides a detailed review of the development and implementation of conservation legislation and the resulting Federal land management agencies and programs. In brief, first attempts to deal with resource conditions occurred with creation of Forest Reserves from public domain lands in 1891 during the presidency of Benjamin Harrison. He set aside more than 13 million acres, and President Grover Cleveland reserved another 21 million acres during his time in office. President Theodore Roosevelt added over 16 million acres (Ferguson and Ferguson 1983, Steen 1976: 22-46). In 1905 President Roosevelt signed the bill transferring the Forest Reserves to the Department of Agriculture, and in July of that year the Bureau of Forestry was renamed the United States Forest Service (Steen 1976: 74-75). In 1906 fees were charged for grazing on Forest Service land. By 1908, there were 21 Forests in Forest Service District 3, which would become the 11 National Forests of contemporary Region 3 (Tucker 1989: 1-2, 68). Tucker (1989, 1992) provides an excellent discussion, personal interviews with early day Forest Service employees, and general sources of information on the development and early days of the Forests of Region 3.

The National Grasslands administered by Region 3 in eastern New Mexico, the Panhandles of Texas and Oklahoma, and western Oklahoma had a somewhat different history. Between 1900 and 1930 increasing numbers of farmers settled in the area as higher than normal rainfall levels and high wheat prices encouraged local farmers to move from stock raising and planting a variety of crops to a single cropping system. As grasslands were plowed under, the ground surface became destabilized and susceptible to erosion. In the 1930s, drought and dust storms hit the area prompting the Federal government to attempt to restore Dust Bowl areas to grassland (Bonnifield 1979, Cojeen 2000, Fredine and others 1999, Lewis 1989, Simpson and others 1998).

Much land reverted to the government during this period owing to foreclosures and sale incentives. In 1934, Congress passed the Taylor Grazing Act, which established the U.S. Grazing Service within the Department of the Interior to rehabilitate overgrazed and eroded lands and manage grazing on public lands. Under this act the bulk of unappropriated grassland was closed to further settlement. In 1946 the Grazing Service merged with the General Land Office to form the Bureau of Land Management. From 1938 until the early 1950s the Soil Conservation Service assumed responsibility for restoring grasslands in the areas under discussion. In 1953, the Forest Service took over responsibility for managing what became the Kiowa, Rita Blanca, and Black Kettle National Grasslands (Bonnifield 1979, Cojeen 2000, Fredine and others 1999, Lewis 1989, Simpson and others 1998).

Forest Service Response to Rangeland Degradation

The 20th century Forest Service response to issues of grassland condition and overutilization has been discussed in various publications and reports going back to and preceding Aldo Leopold in 1924 and *The Story of the Range* in 1926 (see for example Alexander 1997, Barnes 1926, Leopold 1924, Roberts 1963, Robinson 1975, Rowley 1985, Scurlock 1998, Steen 1976). Concerning the condition of lands that became part of the National Forest system, Robinnson (1975: 199) states:

> So long as the forests were part of the public domain, their access and use [were] virtually unrestricted. As a consequence, many of the lands were heavily overgrazed…When management responsibility was transferred to the Forest Service, regulation of grazing, through a system of permits and grazing fees, was established as one of the first and foremost management tasks.

In the West, where much rangeland is Federally owned and ranchers are heavily dependent upon public land for grazing, most grazing permits are issued for 10 years. This term provides greater security for the rancher than do annual permits, which are more common in the East and South where much more land is privately owned (Robinson 1975: 200). Since agency establishment, regulation of livestock grazing on National Forest land has been a compromise, although an often contentious one, between the interests of the livestock industry, the range management goals of the Forest Service, and, more recently, the preservation/protection concerns of the environmental movement.

Research on rangeland condition has been ongoing in the Forest Service since the early years of the agency and its predecessors with the development of the section of Special Investigations in the first year of Gifford Pinchot's tenure at the Bureau of Forestry. By 1902, the section had become a division, and by 1908 the Fort Valley Experiment Station, with Gus Pearson as Director, was established in northern Arizona (Steen 1998: 5-8). In 1915, the Branch of Research was officially established, and by the end of the 1920s, 12 regional Research Stations were operating (Steen 1976: 131-137). In 1915, the Santa Rita and Jornada Experimental Ranges in Arizona and New Mexico were transferred to the Forest Service from the Bureau of Plant Industry, with the research goals of restoring, improving, and maintaining basic range resources and obtaining the greatest returns on livestock (Steen 1998: 17). Medina (1996) has provided a comprehensive annotated bibliography of the work of the Santa Rita Experimental Range.

Early and continuing issues of major concern between the Forest Service and the livestock industry included grazing fees and regulations surrounding issuance of grazing permits, including their duration, use limits, and transferability (Robinson 1975: 209-214). These issues are discussed in detail by Robinson (1975), Rowley (1985), and Steen (1976), among others, and will not be reviewed here. Of great concern to the agency (throughout the early years of the 20th century and continuing to the present) have been improving the condition of and/or restoring Western grasslands and rangelands. As Robinson (1975: 202) notes: "The task of surveying the range and adjudicating individual permits to conform with range capacity began in 1910. Sixty years later it is still unfinished, and range improvement continues to be a major concern of management."

Suffice it to say that the condition and sustainability of the region's rangelands remains an issue of concern and considerable philosophical debate today among not only land managers but also among many concerned groups.

Over the years, the Forest Service and other land management agencies have undertaken range improvement practices using a variety of techniques including reducing stocking levels, shortening grazing seasons (postponing entry into allotments), implementing specialized grazing systems (discussed in greater detail in following sections), and emphasizing range improvements by permittees such as development of waters and fencing (Alexander 1997: 179-194; Robinson 1975: 202). Additional programs, some of which are considered controversial by today's standards, were also implemented in the region. According to Bahre (1995: 247), the contemporary grassland landscape of southeastern Arizona resulted from many Forest Service and BLM practices designed to protect watersheds and improve the livestock industry. Some of these included contour plowing, fencing, using specialized grazing systems, prescribed burning, suppressing fire, controlling woody plants with herbicides, introducing nonnative forage plants, constructing check and spreader dams, and controlling weeds, as well as chaining and bulldozing pinyon (*Pinus edulis*), juniper (*Juniperus* spp.), mesquite (*Prosopsis* spp.), and oak (*Quercus* spp.).

Often, agency-mandated improvement programs met with varying degrees of resistance from user groups and the livestock industry. An example from the forests of northern New Mexico is instructive in this regard, highlighting the tension between traditional use practices and Federally mandated management programs. In this part of the region, loss of land grant lands limits the grazing areas open to small, local communities, many of which are now surrounded by National Forest (Van Ness 1987: 201). In 1938 a Forest Service report estimated that demand for grazing on portions of the Carson and Santa Fe National Forests exceeded potential by 111 percent (Hassell 1968: 12). In the late 1960s, estimates showed grazing obligations on the two Forests for 21,637 cattle and 32,203 sheep, compared to an estimated capacity of 14,370 cattle and 25,237 sheep. Over the years the agency attempted to deal with these discrepancies using a variety of techniques.

Beginning in the 1920s and continuing into the 1960s, livestock ranching on the two Forests changed as the economy changed and the Forest Service implemented range improvement programs (de Buys 1985: 247-249). There was a steady decline in both the number of permittees and the number of animals, from 2,200 permits in 1940 to fewer than 1,000 in 1970. For example, the permittees of the village of Canjilon who grazed animals on the Carson lost permits for 1,000 cattle over a period of a few years (de Buys 1985: 247-259). Free-use permits, issued for animals used in household operation such as milk cows and draft horses, were completely phased out by 1980.

Also during this period, there was a major change in the kinds of animals being grazed, with large declines in sheep and goats under permit. By 1980, there were no goats on either Forest and no sheep on the Santa Fe (de Buys 1985: 247-248; Van Ness 1987: 202). These changes occurred both as a result of Forest Service direction and as a result of changes brought about by the switch from a subsistence-based to a cash-based economy. Land losses and cutbacks in herd size undoubtedly pushed many people into the cash-based economy of wage work. Although there were important rangeland health issues that required treatment, these changes seriously impacted the livelihoods of many villagers who had ranched and farmed in northern New Mexico for generations prior to U.S. conquest. These losses have contributed to resentment of and protest against the Forest Service that continues to the present day.

Today, Federal agency grassland/rangeland management is critiqued by both user groups and the organized environmental community (for example, see Donahue 1999 for an argument against livestock grazing on arid and semiarid Western public lands). In recent years, Region 3 has been increasingly involved in appeals and lawsuits related to efforts to align livestock grazing activities with Federal environmental statutes such as the National Environmental Policy Act (NEPA), National Forest Management Act (NFMA), and the Endangered Species Act (ESA). Under NEPA, environmental assessments are required for issuing and reissuing grazing permits to determine if livestock grazing is an appropriate and suitable use of the land (Recision Act of 1995, PL 104, Section 504). In an example of this type of analysis, the Black Kettle National Grassland has produced an environmental assessment of livestock grazing and associated vegetation management, which includes all 114 rangeland units of the grassland (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000). The role and impact of environmental legislation and efforts at mediation and conflict resolution on the management and sustainability of grassland areas will be discussed in greater detail in following sections of this chapter.

The following portions of this chapter also briefly discuss the current condition of rangeland vegetation resulting from human uses, which are primarily ranching/grazing by domesticated livestock followed by recreation and mineral extraction. Certain effects of recreation and mineral extraction on wildlife are also presented. Rangeland/grassland condition and the effects of grazing on wildlife in general and on threatened, endangered, and sensitive species in particular are examined in detail in volume 2 of this report.

The effects of livestock grazing and recreational use on archeological sites of varying types are taken into consideration in environmental assessments and must be considered in issuing and reissuing grazing permits. Several studies have examined the damaging effects of trampling and rubbing (against the walls of standing structures) by both domesticated livestock and wild ungulates such as elk; other ongoing problems are human visitors' trampling, leaning, or sitting on walls, or removal of artifacts (Gifford-Gonzalez and others 1985, Knudson 1979, Osborn and others1987). Rubbing or leaning against fragile walls can lead to their collapse. Fencing off standing structures in grazing allotments has been used as a means of excluding domesticated animals from the structures. Trampling can damage sites and interfere with their correct interpretation by reducing artifact size through breakage and by displacing artifacts from their original positions. Adding additional fractures to chipped stone tools can obscure interpretations of functionality and production technique. This type of damage primarily affects surface artifact scatters of ceramics and chipped stone but can also affect buried deposits near the surface (Osborn and others1987).

Contemporary Condition of Federal and Non-Federal Rangelands

Despite improvements in the condition of many grasslands since the early years of the 20th century, significant issues remain to be addressed. Land use and management activities such as domesticated livestock grazing, fire suppression, agriculture, and urban/suburban development impact the grasslands of the region as do the expansion of woodlands and the introduction of nonnative plant species (Bahre 1995: 243-255; Mitchell 2000). Mitchell (2000) reviews these issues for the entire United States, as well as for the region. The following discussion is primarily drawn from his work, with other sources cited where appropriate.

Information from the National Resource Inventory (NRI) conducted on non-Federal lands by the Natural Resources Conservation Service (NRCS) in 1982 and 1992, including an unpublished supplemental study conducted in 1992, is used to characterize rangeland condition on non-Federal lands during the late 20th century (USDA Natural Resources Conservation Service 1995, USDA Soil Conservation Service 1987, 1990). The assessments use a species composition model (estimated on a biomass basis and compared to a typical "climax" plant community for the site) to evaluate range condition (Dyksterhuis 1949, Mitchell 2000: 28).

Mitchell (2000: 27-28) discusses problems with this classificatory scheme and reviews the scientific advances in understanding rangeland condition of the past 25 years as follows. Earlier concepts of range condition were based on the Clementsian equilibrium theory of retrogression produced by overgrazing, and by secondary succession to a stable climax after removing the disturbance caused by grazing (Dyksterhuis 1949 discussed in Mitchell 2000). Rangeland condition was divided into four classes: excellent (rangeland in a near climax condition with 76 to 100 percent remaining climax), good (rangeland in a late successional stage with 51 to 75 percent remaining climax), fair (rangeland in a midsuccessional stage with 26 to 50 percent remaining climax), and poor (rangeland in an early seral stage with 0 to 25 percent remaining climax) (discussed in Holechek and others 1998). Many range managers now use the terms climax (or Potential Natural Community, PNC), late seral, mid-seral, and early seral to replace the terms excellent, good, fair, and poor, distinguishing range ecological condition from how well existing vegetation may be suited for specific uses such as grazing by domesticated animals (Holechek and others 1998). Tables 5-1 and 5-2 show use of these classifications.

U.S. land management agencies have maintained these classification schemes in inventory and assessment programs although scientific evidence has challenged the Clementsian theory of succession (discussed in Mitchell 2000). Work by Westoby and others (1989) set forth an alternate hypothesis for nonequilibrium vegetation dynamics on disturbed rangelands. In addition, the view that ecosystem behavior under stress should be described only in terms of vegetation responses has been questioned by ecologists from various disciplines (Rapport and others 1985). Joyce (1993) has summarized the changes in the conception of range condition from a prior simplistic model to the current changing, complex situation (discussed in Mitchell 2000).

In 1994, the Committee on Rangeland Classification proposed a new paradigm for assessing rangeland conditions based on several years of review and study. This assessment paradigm is based on nonequilibrium, state-and-transition models of succession, focusing on ecosystem function rather than state or plant community composition. Three major criteria of ecosystem function are soil stability and watershed function, distribution of nutrient cycling and energy flow, and recovery mechanisms (Committee on Rangeland Classification 1994). The Society for Range Management also assembled a Task Group on Unity in Concepts and Terminology (1995) that called for making sustainability—defined in terms of maintaining soil productivity—the fundamental goal of range management. Unfortunately, these advances have not yet been incorporated into national data sets. Thus, the older classificatory system is used in Mitchell's review and discussion (2000).

Table 5-1 shows the condition of non-Federal rangelands in Arizona, New Mexico, Oklahoma, and Texas in both 1982 and 1992. Arizona and New Mexico have slight improvements (defined as increases in excellent and good condition classes) over the 10-year period, while Oklahoma shows greater improvement. Texas, on the other hand, has virtually no improvement, and according to Mitchell (2000: 30), the non-Federal rangelands of Texas are more degraded than those of any other Great Plains State. Since less than 2 percent of Texas ranges are Federal, these condition estimates essentially cover the entire State (USDI Bureau of Land Management 1997).

Federal ranges of the area include lands managed by the BLM and the Forest Service. Discussing their condition is complicated because the two agencies report condition in different terms. The BLM currently uses the Ecological Site Inventory (ESI) procedure (USDA Soil Conservation Service 1976, USDI Bureau of Land Management 1984), and the Forest Service assesses the condition of rangeland vegetation in terms of established Forest Plan Management Objectives (FPMO). Uplands and riparian range areas are described separately by both agencies. Mitchell (2000: 33) is critical of the Forest Service evaluation system, observing that many National Forests are operating under plans approved in the 1980s, "…and the elements relating to rangeland health are not well correlated to trends in indicators judged to be relevant by contemporary standards" (Committee on Rangeland Classification 1994).

Table 5-2 gives information on the condition of upland BLM rangelands in Arizona and New Mexico in 1986 and 1996. There are but a few hectares in Oklahoma and none in Texas. Both Arizona and New Mexico have increases in areas with late seral

Table 5-1. Condition of non-Federal rangelands in Arizona, New Mexico, Oklahoma, and Texas, 1982 and 1992, from the National Resource Inventory (adapted from Mitchell 2000: 29, 31, table 3.2 and 3.6).

		Year			
State	**Condition class**	**1982**		**1992**	
Arizona	Excellent	518[1]	2%	650	2%
	Good	4,923	16%	8,446	27%
	Fair	16,574	54%	15,886	50%
	Poor	8,832	28%	6,661	21%
New Mexico	Excellent	659	2%	591	2%
	Good	12,262	30%	14,314	36%
	Fair	22,617	55%	21,227	53%
	Poor	5,422	13%	3,645	9%
Oklahoma	Excellent	907	6%	1,749	12%
	Good	3,601	24%	4,492	32%
	Fair	7,639	51%	5,835	42%
	Poor	2,904	19%	1,951	14%
Texas	Excellent	480	1%	174	<1%
	Good	13,546	15%	16,324	18%
	Fair	53,543	57%	49,899	55%
	Poor	25,681	27%	24,922	27%

[1]Acres x 10^3 and percent of total area for which condition ratings were applied.

Table 5-2. Condition of rangelands managed by the Bureau of Land Management (BLM) in Arizona and New Mexico, 1986 and 1996 (adapted from Mitchell 2000: 32, table 3.7).

		Year			
State	**Condition class**	**1986**		**1996[1]**	
Arizona	PNC[2]	467[3]	4%	521	9%
	Late seral	2,801	24%	2,217	40%
	Mid seral	6,068	52%	2,217	40%
	Early seral	2,334	20%	652	12%
	Unclassified	0		913	
	Not inventoried			5,123	
New Mexico	PNC	125	1%	102	1%
	Late seral	3,002	25%	3,555	36%
	Mid seral	6,003	50%	4,673	48%
	Early seral	2,877	24%	1,422	15%
	Unclassified	500		305	
	Not inventoried			2,540	

[1]The acres and percentages by condition class for 1996 are based only on those acres inventoried using the Soil-Vegetation Inventory Method (used for 5 years prior to the ESI) or the Ecological Site Inventory (ESI) and classified by condition.
[2]Potential Natural Community (Kuchler 1964)
[3]Acres x 10^3 and percent of total.

communities (developmental stages of an ecological succession) and decreases in the areal extent of early seral communities, distinct from other BLM rangelands in Western States, which show little change during the period (Mitchell 2000:31). Table 5-3 presents upland range condition information for Forest Service grazing allotments in Arizona and New Mexico, showing virtually no change over the 3 years of assessment used in the study (1995, 1996, 1997). Slightly over 25 percent of the ranges under discussion are verified or estimated as neither meeting nor moving toward FPMO, as compared to 15 percent in the Rocky Mountain Assessment Region as a whole. This assessment region encompasses Forest Service grazing allotments in Montana, Idaho, Wyoming, North Dakota, South Dakota, Nevada, Utah, Colorado, Kansas, Nebraska, Arizona, and New Mexico (Mitchell 2000: 35-37). Mitchell (2000: 36) notes:

> Vegetation in the mostly dry Southwestern Region has been subjected to a history of fire suppression since the late 19th century and improper grazing, primarily between the 1880's and World War I (Secretary of Agriculture 1936, Rasmussen 1941, Cooper 1960, Buffington and Herbel 1965, Mortensen 1978). These factors and others have caused upland vegetation and soil changes that are slow to improve, some of which probably will not ever recover to pre-existing conditions (Schlesinger et al. 1990, Wang and Hacker 1997). As a result, the higher percentage of rangeland in R-3 (USFS Southwestern Region 3) that is not meeting or progressing towards FPMO's, roughly 25 percent, should not be surprising…

Conditions of riparian rangeland areas in Arizona and New Mexico managed by the BLM are reported in terms of Proper Functioning Condition (PFC) (Barrett and others 1995, USDI Bureau of Land Management 1991), while those managed by the Forest Service are reported in terms of FPMOs, as are upland ranges (Robertson 1992). In Arizona, 39 percent of BLM-managed riparian areas are reported in Proper Functioning Condition, while 33 percent of those in New Mexico are considered to be in PFC. Arizona reports 58 percent Functioning at Risk and 3 percent Nonfunctional. New Mexico has 45 percent Functioning at Risk and 22 percent Nonfunctional (Mitchell 2000: 41, table 3.20). These BLM data are for 1997.

The Forest Service assessment of riparian area condition within grazing allotments is reported for the Rocky Mountain Assessment Region as a whole for the years 1995, 1996, and 1997. Table 5-4 gives this information and shows that about 1 out of 6 acres of riparian land does not meet and is not moving towards FPMO; this is roughly the same as reported for the entire BLM Rocky Mountain Assessment Region (Mitchell 2000: 41). As discussed for the upland areas and reviewed in following paragraphs, various historical and contemporary factors and activities, as well as site-specific attributes of different locales, must be taken into consideration when assessing the condition of riparian areas throughout the region.

Table 5-3. Condition of rangelands in grazing allotments managed by the Forest Service in Arizona and New Mexico (combined) as related to Forest Plan Management Objectives (FPMO), 1995, 1996, 1997 (adapted from Mitchell 2000: 37, table 3.14).

	Year		
Land category	**1995**	**1996**	**1997**
Verified meeting FPMO	1,676[1]	1,859	1,945
Estimated meeting FPMO	2,541	2,595	2,376
Total	4,217	4,454	4,321
	(26%)	(27%)	(27%)
Verified moving toward FPMO	1,486	1,456	1,455
Estimated moving toward FPMO	6,227	6,208	6,105
Total	7,713	7,664	7,560
(47%)		(46%)	(47%)
Verified not meeting or moving toward FPMO	788	673	741
Estimated not meeting or moving toward FPMO	3,663	3,702	3,548
Total	4,451	4,375	4,289
	(27%)	(27%)	(26%)
Undetermined status	1,938	1,839	1,849
Total (Lands with range vegetation management objectives)	18,319	18,322	18,019

[1] Acres x 10^3 and percent of total lands with range vegetation management objectives.

Table 5-4. Area of riparian range vegetation within grazing allotments on National Forest System lands in relation to Forest Plan Management Objectives (FPMO), Rocky Mountain Assessment Region (adapted from Mitchell 2000: 42, table 3.22).

	Year		
Land category	**1995**	**1996**	**1997**
Verified meeting FPMO	1,391[1]	1,450	1,777
Estimated meeting FPMO	4,549	4,421	4,662
Total	5,940	5,871	6,439
	(42%)	(45%)	(47%)
Verified moving toward FPMO	922	858	875
Estimated moving toward FPMO	4,519	3,973	4,092
Total	5,441	4,831	4,967
	(39%)	(37%)	(37%)
Verified not meeting or moving toward FPMO	402	266	208
Estimated not meeting or moving toward FPMO	2,226	2,059	1,905
Total	2,628	2,325	2,113
	(19%)	(18%)	(16%)
Undetermined status	2,331	2,290	2,241
Total (Lands with range vegetation management objectives)	16,340	15,317	15,760

[1] Acres x 10^2 and percent of total.

Specific information on the rangeland condition of the National Grasslands in New Mexico, Texas, and Oklahoma is presented in a somewhat different manner in the Geographic Area Assessments prepared by grasslands management during 1999 and 2000. The Black Kettle and McClellan Creek National Grasslands are located in Oklahoma and Texas. Two of their four geographic areas (High Plains and Redbed Plains) are used for regular livestock grazing under permit. The Lake Marvin Area is not currently grazed, and the Lake McClellan Area has occasional grazing as a management tool. All rangelands within the four geographic areas are described as in satisfactory range condition, which is defined as follows (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000: appendix 38): "Satisfactory Rangelands—Those that have sufficient vegetative cover to protect the soil resource and provide a sustainable base for desired plant/animal communities." Satisfactory rangelands are those where current soil loss (the amount of soil that would be lost under the current or existing vegetative ground cover) is less than tolerance soil loss (the amount of soil that would be lost under conditions of the potential natural vegetative ground cover and where soil development equals soil loss) (TES Handbook 1986). Further information on rangeland condition can be found in the Geographic Area Assessments and in the Environmental Assessment for Livestock Grazing and Associated Vegetation Management, Black Kettle National Grassland—Roger Mills County, Oklahoma (1999).

The Kiowa National Grassland is in New Mexico and contains the Mills Canyon, Mills Upland, and the western third of the Southern Prairie Geographic Areas. The Rita Blanca, in Oklahoma and Texas, comprises the remainder of the Southern Prairie Geographic Area. The three areas are described as in satisfactory rangeland condition with the exception of small areas occupied by playa lakes, which contain substantial portions of bare ground. The floodplain of the Mills Canyon Area is also described as in unsatisfactory condition resulting from severe flood events (Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999). Again, more detailed information can be found in the Geographic Area Assessment.

As the previous discussion indicates, regional rangeland condition and the ways in which it is measured and assessed are topics of both concern and debate. The Southwestern rangelands/grasslands are used for and impacted by many activities, which are also topics of debate. Discussions of these activities, such as livestock grazing, mining/energy extraction, recreation/tourism, and other special uses, follow.

Certainly the most widespread and increasingly scrutinized use, especially on public lands, is livestock grazing. Mitchell (2000: 31) summarizes:

> The two predominant opposing viewpoints (on grazing) are epitomized by Fleischner (1994) and Box (1990). Fleischner believes grazing has caused a loss of biodiversity, disruption of ecosystem function, and

> irreversible changes in ecosystem structure, while Box concludes that the trend of U.S. public rangelands, on the average, has been upwards over a number of decades and the land is in the best ecological condition of this century.

For a detailed argument supporting the view expressed by Fleischner, see Donahue (1999). For a rebuttal to the Fleischner paper, see Brown and McDonald (1995).

Mitchell (2000: 42) states further:

> Some reviewers conclude that grazing by domestic livestock is not compatible with restoring watersheds and water quality, at least in the short term (Belsky et al.1999). However, comparative [sic] research and case studies show that improved rangeland and livestock management practices are compatible with watershed and water quality improvement when designed to address the attributes of each individual site (Elmore and Kauffman 1994). Ultimately, social and political values, along with scientific knowledge, will drive future laws and regulations affecting the grazing use of riparian areas, just as they will for uplands (Lee 1993).

Livestock Ranching on Federal and Non-Federal Lands

The section describes the form and occurrence of ranching in the Southwest, as well as its socioeconomic and cultural contributions to local communities and the region. The focus is on livestock ranching on public lands. Ranching and range condition on Tribal lands are beyond the scope of this review, but information on this topic can be found in Brugge and Gerow (2000). Ranches in Arizona and New Mexico are generally composed of a base of private land augmented by Federal and/or State grazing allotments. Ranchers do not hold fee-simple property rights to grazing allotments, but these public lands are traditionally seen as part of the ranch and are considered in determining ranch sale prices and appraisals for Federal estate tax purposes (Ruyle and others 2000: 380).

The number of cattle ranches in Arizona varies depending on the definition of a ranch. Ruyle and others (2000: 387) note that during 1995 there were about 2,500 farms and ranches reporting at least one beef cow. Most livestock operations in the State have fewer than 50 cows, which is consistent with the national average. The majority of Arizona ranches are cow-calf operations, consisting of a base cow herd and the animals needed to support them.

Most of the ranches in New Mexico are also small, cow-calf operations with from one to 99 head. This size class comprised 70 percent of the State's 8,313 ranches in 1996. In the north-central mountain area of the State, the small operations made up 82 percent of the listed 1,804 ranches. This area also has fewer large (greater than 500 head) ranches—3 percent of the total; Statewide, they account for 7 percent of the total (Torell and others 1998: table 1). In both New Mexico and Arizona, 30 percent of ranches range from medium to large with at least 100 head (Ruyle and others 2000: 387-388). In northern New Mexico, however, only 17.5 percent fall within this size range (Torell and others 1998: table 1). Thus, northern New Mexico has considerably fewer medium-to-large ranches than either Arizona or New Mexico as a whole.

New Mexico has over twice as many grazing permittees on National Forests as Arizona, but New Mexico has 35 percent fewer animals (Raish and others 1997: 28-35). These figures result in large part from the small ranches and small herd sizes of northern New Mexico, as well as from the tendency to have multiple-permittee grazing allotments on National Forests in the area. These figures indicate the continuity of the long-standing tradition of small-sized operations and communal herding prevalent in the area since Spanish colonial times (Raish and others 1997: 28-35).

Because many ranching operations in the region rely to some degree on public land, regulations and management decisions affecting these lands can impact the operation and future of ranching throughout the area. Ruyle and others (2000: 380-383) found that for Arizona, but also applicable to New Mexico, many operators rely on a combination of privately leased land as well as State and Federal (USFS and BLM) grazing allotments. In Arizona, public and State grazing permits and leases account for roughly 85 percent of the State's grazing land, excluding American Indian lands. In New Mexico, 53 percent of the land is nonprivate (Fowler 2000: 423).

The degree to which a ranch relies on leased and permitted land affects the complexity of ranch management, with regulations, fees, and enforcement often varying between agencies and within the same agency from location to location. Because ranchers rely on government grazing permits, they are affected by the permitting agency's regulations. The managing agency defines grazing seasons and stocking rates, which are often limited by competing uses and values such as recreation or riparian restoration. Restrictions stemming from the Endangered Species Act (ESA), the National Environmental Policy Act (NEPA), and the National Historic Preservation Act (NHPA), for example, can affect the timing and construction of range improvements such as water developments and fencing (Ruyle and others 2000: 382-383).

The grassland areas of northeastern New Mexico, Texas, and Oklahoma comprise significantly greater amounts of private land than do other portions of the Southwest (Fowler 2000: 423-427; USDI Bureau of Land Management 1997). Fowler (2000: 423-427: table 2) notes that northeastern New Mexico has the most productive rangeland, supports the greatest number of ranches (27 percent), and the greatest number of large ranches (35 percent) in the State. These figures come from the total of 8,313 ranches identified in 1996 (Torell and others 1998: 4, table 1). Both cow-calf and the majority of the State's yearling operations are located in the area. Yearling operations—not including cow-calf ranches that purchase weaned calves when forage is available—purchase calves to put on leased pasture. They are generally grazed on winter wheat until they reach sufficient size to be sent to the feedlot (Fowler 2000: 426).

The grasslands of the Great Plains States (North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, and Texas) contained almost 40 percent of the nation's beef cow herd according to 1996 statistics (USDA National Agricultural Statistics Service 1997). In these six States income from livestock generally exceeds income from other agricultural commodities, and cattle grazing is the predominant land use. There are twice the numbers of stocker cattle in these States, especially in Kansas, Oklahoma, and Texas, than in the Western States (including Arizona and New Mexico). Engle and Bidwell (2000: 105) found:

> For a variety of reasons, including closer proximity to feedlots and lower winter feed costs for cow-calf enterprises, overall costs of cattle production in the Great Plains are lower than in the regions to the west. The sum of these factors provides a competitive edge for Great Plains grassland cattle over cattle production elsewhere in the western United States (Cheeke and Davis 1997).

Livestock Numbers on Forest Service Land

Nationally, Region 3 ranks second to Region 4 (southern Idaho, Nevada, Utah, and western Wyoming) in permitted grazing use, according to the Forest Service Grazing Statistical Summary available at the time of this report, which reports information for the 1998 grazing season (USDA Forest Service 1999). Table 5-5 gives figures on the numbers of permittees, animals, animal unit months (AUMs), and head months (HMs) both permitted and authorized to graze on National Forest system lands (including the grasslands) in Region 3 in 1998. An AUM is the amount of forage required to support a mature, 1,000-lb cow and calf or its equivalent for 1 month, while an HM is the time in months that livestock spend on National Forest system land (used for billing purposes) (USDA Forest Service 1999). "Permitted to graze" indicates "Livestock permitted by a grazing permit, grazing agreement, livestock use permit, or other permitting document." "Authorized to graze" indicates "The number of livestock that are authorized and billed for grazing on National Forest System land" (USDA Forest Service 1999: 96). Authorized livestock are those that are actually grazing on the Forest and being paid for. Tables 5-6, 5-7, and 5-8 present the same information broken down for the National Forests in Arizona and New Mexico (excluding the National Grasslands), and for the National Grasslands of the region located in New Mexico, Texas, and Oklahoma.

The figures from tables 5-6, 5-7, and 5-8 show that the average permitted number of animals per permittee is around 318 for the National Forests in Arizona, 102 for the Forests in New Mexico, and 65 for the Grasslands. There are 2.5 times as many permittees in New Mexico (1,082) as in Arizona (425) with the Arizona permittees averaging roughly three times as many permitted animals. The larger numbers of permittees and smaller numbers of animals in New Mexico in many cases reflect the persistence of traditionally small operations with multiple permittees per allotment, especially in the

Table 5-5. Livestock grazing on National Forest System lands in Region 3 in 1998.

	Number of animals	AUMs	HMs
Permitted	258,279	2,452,253	2,115,615
Authorized	217,970	1,900,545	1,650,336
Total of 1703 permittees.			

Table 5-6. Livestock grazing on National Forests in Arizona in 1998.

	Number of animals	AUMs	HMs
Permitted	135,188	1,327,115	1,113,027
Authorized	107,081	916,392	779,235
Total of 425 permittees.			

Table 5-7. Livestock grazing on National Forests in New Mexico in 1998.

	Number of animals	AUMs	HMs
Permitted	110,393	1,044,416	928,407
Authorized	98,466	904,940	798,381
Total of 1082 permittees.			

Table 5-8. Livestock grazing on National Grasslands in Region 3 in 1998.

	Number of animals	AUMs	HMs
Permitted	12,698	80,722	74,181
Authorized	12,423	79,213	72,720
Total of 196 permittees.			

north-central portion of the State. Figures from the Grasslands tend to be somewhat misleading because livestock operations in the Plains areas generally have access to considerably greater amounts of private land on which to graze other portions of their herds. There are also a fair number of part-time farmers and ranchers with small herds using the Grasslands (Dickerson personal communication 2001) as there are throughout the Southwestern Region.

Region 3 rangeland use statistics show a fluctuating but primarily downward trend from 1982 to 1995 in numbers of permittees, animals permitted and authorized to graze, and AUMs. During this period, permittee numbers in both Arizona and New Mexico dropped by about 25 percent, and animals authorized to graze dropped by approximately 25 percent in Arizona and 20 percent in New Mexico (Raish and others 1997: 36). Declines are apparently related to climatic and market fluctuations, permit consolidation, and growing urbanization in the area (Raish and others 1997: 36). From 1996 through 1998, regional permittee numbers rose from 1663 to 1703, an increase of 2.4 percent. Permitted animals declined from 270,068 to 258,279, or a 4.4 percent decrease. AUMs increased by 1.9 percent from 2,406,708 in 1996 to 2,452,253 in 1998 (USDA Forest Service 1997, 1998, 1999). These are relatively slight fluctuations when compared to the declines of the period from 1982 to 1995.

Grazing Management Systems on Forest Service Land

A considerable body of information exists concerning grazing systems in use on the arid and semiarid rangelands of the Southwestern and Western United States. This information is ably defined, reviewed, and discussed elsewhere, for example in Holechek and others (1998) and Kruse and Jemison (2000), to name only two. From a relatively rough assessment and review of grazing systems used in the Southwestern Region (USFS), it is apparent that systems incorporating deferment, rest, and rotation are the most common. (All regional rangeland information used in this discussion and in table 9 was provided by Gene Onken, Rangeland Management Staff, USFS Southwestern Region). According to Holechek and others (1998: 228-229):

> Deferment involves delay of grazing in a pasture until the seed maturity of the key forage species. This permits the better forage plants to gain vigor and reproduce. Rest is distinguished from deferment in that the range receives nonuse for a full year rather than just during the growth period. This gives plants a longer period to recover from past grazing influences and provides wildlife with a pasture free from livestock use during the critical dormant period…Rotation involves the movement of livestock from one pasture to another on a scheduled basis. It is the critical feature of all specialized grazing systems.

Table 5-9. Grazing systems in use on the range management units of the National Forests in Arizona and New Mexico.

	Number and percentage	
Grazing system	**Arizona**	**New Mexico**
Continuous	33 (6%)	109 (23%)
Deferred	95 (18%)	51 (11%)
Deferred-Rotation	200 (37%)	130 (28%)
Rest Rotation	183 (34%)	132 (28%)
Rotation or Alternate	13 (2%)	42 (9%)
Other/Unlisted	18 (3%)	3 (.6%)
Total	542 (100%)	467 (100%)

Out of approximately 1,009 allotments that I was able to determine for the Forests of the Region (excluding the National Grasslands), 330 (33 percent) report using deferred-rotation, 315 (31 percent) rest-rotation, 146 (14 percent) deferred, and 142 (14 percent) continuous systems (table 5.9). Another 55 (5 percent) use rotation or alternate forms (table 5-9). Thus, around 83 percent are using some form of deferment, rest, or rotation.

Deferred-rotation "discontinues grazing on various parts of a range, allowing each part to rest successively during the growing season…two, but usually three or more, separate units, or pastures, are required" (Kruse and Jemison 2000: 41). In rest-rotation, grazing is deferred for a complete year on various portions of the range during succeeding years. Two or more units are required but most rest-rotation systems involve three or four pastures (Holechek and others 1998: 244; Kruse and Jemison 2000: 42). Deferred systems discontinue grazing on an area for a specified period (during the growing season, for example) to promote plant reproduction, establishment of new plants, or restoration of vigor to older plants (Kruse and Jemison 2000: 40). Continuous grazing, on the other hand, refers to grazing the same area throughout a year or that portion of the year when grazing is feasible. It may be yearlong or shorter depending upon environmental or other restrictions (Kruse and Jemison 2000: 37). Table 9 presents information on the primary systems in use on the National Forests in Arizona and New Mexico broken down by State. It should be noted that many of the continuous systems reported for New Mexico are seasonal in that grazing is not feasible in high altitude units during portions of the year.

As with the Forests of the Region, a rough count from the available information was used to examine grazing systems in use on the National Grasslands. The Geographic Areas Assessments for the Kiowa and Rita Blanca (1999) show deferred-rotation as the system in use for the range management units with available information. These comprise approximately 80 units in the Mills Canyon and Mills Upland Geographic Areas. The Black Kettle and McClellan Creek National Grasslands systems are described as variable season grazing and variable season rotational grazing. Grazing units occur in the High Plains and Redbed Plains Geographic Areas. The vast majority of the 114 rangeland units are described as being grazed in a rest-rotation type of system with variable livestock numbers and/or seasons. According to the description for the High Plains Geographic Area (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000: 27-30), a maximum number of AUMs is set forth according to season and vegetative conditions with a minimum 30-day recovery period during the growing season.

Sociocultural and Economic Role of Livestock Ranching

Ranching in New Mexico and Arizona, as well as on the Federally managed grasslands of eastern New Mexico, Oklahoma, and Texas, provides both economic and sociocultural contributions to families, communities, and the region. As discussed by Ruyle and others (2000: 379):

> Despite major shifts toward urbanization in the state, livestock grazing remains the most widespread use of Arizona rangelands. Through the sale of calves, yearlings, stocker cattle, and culled cows and bulls, cattle ranching accounts for nearly 25% of the agricultural economy in the state. However, based on profit alone, the economic viability of ranching in Arizona is questionable. Because of economic factors related to income production, decisions to remain in the ranching business are not entirely financial. Lifeway considerations have long played an important role in the process.

Economic status of ranches is determined by the larger factors of productivity, market prices, and production costs. More specific factors include range conditions, weather, the number of AUMs assigned to a grazing permit, management decisions, and ranch size (Fowler and others 1994, Ruyle and others 2000). Ranch size can be extremely important. Fixed costs, which remain the same whether animal numbers rise or fall, drive the move toward larger ranches. Seperich and others (1995) show that large-sized ranches do cover the fixed cost of operator salary better than do medium or small-sized ranches. A ranch should exceed 150 AUs to meet minimal family living expenses, a greater number than the one to 99 AUs of the small-sized ranches. Ruyle and others (2000: 393) estimate that small-sized operations would need to bring in about 50 percent of their income from nonlivestock sources to meet living expenses (for Yavapai County, Arizona).

Fowler and others (1994: 1, appendix 4) discuss off-ranch income in New Mexico as follows:

> Seventy-five percent of the small ranches (1-99 A.U.) had people employed off the ranch contributing 44 percent of the family's income. Fifty-five percent of the medium size (100-350 A.U.) ranches had family members working off the ranch also earning 20 percent of the income. Forty-three percent and 36 percent of the large and extra large ranches had people working off the ranch, respectively, earning 13 percent and 6 percent of the family's income.

In addition to economic considerations, culture, tradition, and quality of life seem to become increasingly important in the decision to continue ranching among owners of the smaller ranches. Of course, these qualities are also valued by the owners of larger ranches. In addition, other factors influence the decisionmaking process among the small ranch operators. Eastman and others (2000: 543) found that small ranches in northern New Mexico, but equally applicable to other areas of the Southwest, are often viewed as an investment and a form of savings:

> While the ranch may produce little or even a negative operating income, the assets have a high value, which is expected to increase. Most northern ranchers own their homes, land, and cattle, and these constitute a significant investment and form of savings, which often has very high value. Managed properly, operating losses often provide income tax write-offs against other income. Thus, small operators stand to benefit from a reduced tax burden while their assets increase in value.
>
> The ranchers often view their animals as *banks-on-the-hoof,* which can be used…for emergencies, for periods of unemployment, or for special needs such as college tuition for the children…(Eastman and Gray 1987; de Buys personal communication 1995).

In addition to providing family income at various levels, ranching operations pay a variety of taxes that contribute to community, State, and Federal resources. At the State level, average ranches in Arizona paid $10,468 in combined property, livestock, and other taxes in 1991 (Fowler and others 1994). In New Mexico average ranches paid $4,479 in 1991 (Fowler and others 1994). Of course, property taxes vary according to millage rates and the amount of private land owned by the ranch. Property and other taxes (workmen's compensation, unemployment, and sales tax) are considerably higher in Arizona than in New Mexico (Fowler and others 1994).

Local communities also benefit from ranch-related expenditures from all size classes of livestock operation. Fowler and others (1994) estimate that ranch expenditures in local communities averaged $20,680 in Arizona and $16,529 in New Mexico in 1991. Although smaller ranches spend less in their local communities than larger ranches, basic family expenses, such as food, clothing, medical expenses, gasoline, and vehicle repairs, must be met to maintain a viable operation.

Feeding on the mesa, northern New Mexico, 1993. (Photo by Alice M. McSweeney)

Ranch business also contributes to veterinary and livestock supply businesses as well as to restaurants and movie theaters. "If these ranches were no longer viable the local economies could expect a sharp decrease in their business, as many of these (businesses) are very dependent upon the ranches of the rural regions of the state (New Mexico)" (Fowler and others 1994: 2, appendix 4).

As discussed for the Kiowa and Rita Blanca National Grasslands but applicable to the Black Kettle and McClellan Creek as well, agriculture and the rural lifestyle are strongly predominant, almost to the exclusion of other forms, in these areas. Grazing is the primary agricultural use, with few if any other industries present or predicted to develop. The local communities are almost entirely supported by agricultural income, and taxes on rangeland and livestock contribute the majority of the county's tax revenue. Many of the ranchers in the immediate area are dependent upon the National Grasslands because of their extent and the pattern of intermingled private and Federal lands. Most could not sustain their operations without grazing permits on the Grasslands (Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999).

In addition to economic studies, various authors have examined the sociocultural contributions of the rural ranching lifeway to the continuity of culture and tradition in the Southwest. For example, recent studies by Atencio (2001), Eastman and others (2000), McSweeney (1995), and Raish and McSweeney (2001), among others, examine the role of ranching in maintaining the culture, heritage, and ties to ancestral lands of the Hispanic villages of northern New Mexico. In many cases in this area public land serves as a replacement for grazing lands and resource collection areas lost by local communities in the period following U.S. conquest. Thus, their continued use is considered to be critically important in maintaining community solidarity and traditional lifeways (Eastman and others 2000). These cultural values often outweigh the purely economic contributions of agricultural operations (Hess 1990; also discussed in Raish and McSweeney 2001).

This is the case in many other areas of the Southwest as well, especially with respect to family-owned farms and ranches. Research conducted on rancher motivations in Arizona showed that approximately 72 percent of the 89 ranchers interviewed indicated that profits were not a primary motivating goal for them (Smith and Martin 1972). Other research between 1970 and 1991 showed that ranchers had noneconomic values, obtaining social and psychological benefits from ranch ownership that offset economic problems (discussed in Ruyle and others 2000). According to Fowler (2000: 440):

> Ranchers in New Mexico and Arizona were similar…Arizona ranches had been in the business for 36 years and New Mexico 35 years; the average (rancher's) age was 56 years. Arizona ranch families had been in their respective states for over 64 years. This information strongly suggests longevity in the industry (and) is usually associated with long-term commitment and stewardship. Ranching as a "way of life" was also supported by such demographic information.

Writings by ranch owners express strong attachment to one particular place with enduring ties to the local community and its economy and considerable reluctance to relocate (discussed in Ruyle and others 2000). Although more recent writings express concern over low income returns from ranch operations, ranchers place tremendous value on their operations and want their children to carry on the family tradition even if it provides lower income than other professions. These writers express pride in agricultural occupations, feeling that they are superior to urban occupations. They respect the knowledge of the older generation, which is often based in experience rather than formal education. They believe that ranching allows them to feel closer to the earth and provides a sound place to rear children. There is high value placed on rural life with a strong attachment to the land and land stewardship and a deeply felt sense of place (Cofer n.d., Duncklee 1994, Flieger 1991, Hughes 1980).

Mineral Resource Extraction on Forest Service Lands

Without doubt, livestock grazing is an important use of Forest Service grasslands in the Southwest; however, other uses of these lands merit attention. This section focuses on mineral/energy extraction, and the next section focuses recreation—two other major land uses.

Both surface and hard rock mining and oil and gas extraction occur on National Forest System lands, although the proportion of grasslands that support these activities is not known. The mining laws of 1866, 1870, and 1872 set the stage for mineral extraction activities; under the Mining Act of 1872 and the Mineral Leasing Act of 1920, mining and energy concerns are allowed to explore, stake claims, and apply for leases on the public domain (Robinson 1975: 4; Roth 1997: 242). Rules and regulations for mineral extraction on Forest Service land are presented in 36 Code of Federal Regulations (CFR) Chapter 11 (7-1-00 Edition), Part 228—Minerals. Standards are also contained in Surface Operating Standards for Oil and Gas Exploration and Development (USDI Bureau of Land Management and USDA Forest Service Rocky Mountain Regional Coordinating Committee 1989) and in the Uniform Format for Oil and Gas Lease Stipulations (USDI Bureau of Land Management and USDA Forest Service Rocky Mountain Regional Coordinating Committee 1989).

Although mining has a long history in the region going back to American Indian and Spanish Colonial times, large-scale mining efforts primarily developed after U.S. takeover. In addition to collecting and quarrying many types of rocks, minerals, and clays, American Indian groups mined turquoise for ornamentation and religious purposes with major quarries and mines located near present-day Cerrillos, NM. Galena, or lead ore, was also mined in the Cerrillos area, as well as in the San Pedro, Sandia, and Sangre de Cristo Mountains in New Mexico. Much of the lead was used in producing lead glaze paint for Puebloan pottery (Scurlock 1998: 100-103).

Mining became progressively more developed under the Spanish, Mexican, and American governments. Although exploration for mineral wealth was a major impetus for Spanish exploration, these riches were never realized in the region. In the Middle Rio Grande, the Spanish colonists did mine turquoise and lead, as the American Indian groups had done, as well as copper, silver, and gold. They also extracted nonprecious minerals and rocks, such as mica, which was used to cover window openings (discussed in Scurlock 1998: 118-119). Silver and gold were sought during the Spanish and Mexican periods in southern Arizona, although mining in the area was limited. There is some evidence of mining for gold, silver, and lead in the region, but since later operations often used traditional Spanish and Mexican techniques, it becomes difficult to determine whether the mining occurred before or after the United States assumed control of the area (Hadley and Sheridan 1995: 46-47). According to Whittlesey and others (1997: 288), there were no mining centers north of the Gila River until 1860.

Anglo-American takeover led to major changes in the mining industry in technology, extent, and intensity. Transportation was limited until entry of the railroad from 1879 into the 1890s, with construction of both major lines and spurs into specific mining locations. After railroad entry, serious commercial markets developed. In the Rio Grande, Mexican period gold mines in the Ortiz and San Pedro Mountains continued to be productive after U.S. conquest. Other productive locations for gold, silver, lead, and other minerals included Elizabethtown, Cerrillos, Bland-Albemarle (which remain as ghost towns within the Santa Fe National Forest), and Socorro-Magdalena (Scurlock 1998: 129-134). Scurlock (1998: 130-132) gives a detailed listing of mining locations with types of minerals and dates of production for the Rio Grande area. Most of these locations are found in the forested, montane regions, however; Scurlock does not detail the extent of grasslands or meadows that might also be present.

American period mining in southern Arizona is described by Hadley and Sheridan (1995: 47-65) as producing several mining booms after the Gadsden Purchase (1854) and into the 1880s despite threats from Apache raiding. Lead and silver were produced in the Mowry area from the Mowry and adjacent mines near Patagonia into the 1950s. Other mines produced silver, lead, zinc, copper, and gold ores. Mining occurred at Washington/Duquesne, Sunnyside, Harshaw, and Meadow Valley. Mines in central and western Arizona increased during the 1860s as a means of financing the

Civil War (Whittlesey and others 1997: 288). Central Arizona mines produced gold, silver, and copper and were dependent to a considerable degree on the state's network of railroads. Copper mines along the Verde River led to the boomtown of Jerome, and copper and other minerals drew settlement to the Tonto Basin and surrounding areas (Whittlesey and others 1997: 290).

As in New Mexico, many of the historic hardrock mining locations are situated in or adjacent to contemporary National Forests and are generally located in the more mountainous, forested zones as opposed to the grasslands. Nonetheless, growth of the mining industry affected livestock production and associated grasslands as demands for meat, milk, hides, and tallow from the mining camps increased, encouraging greater livestock production and ranch development, producing what has been discussed by West (1993) as a mine-ranch complex (Hadley and Sheridan 1995). Contemporary mining activities include those for both locatable (for example, gold and silver) and common variety minerals (for example, common variety pumice). Some examples include gold, silver, and copper mines on the Kaibab, Tonto, Lincoln, and Gila National Forests and common variety mineral and rock extraction on virtually all Forests in Region 3. Common variety pumice, caliche, sand, gravel, landscape/building stone, and cinders are all extracted and, collectively, have considerable value to forest users (Linden personal communication 2001).

The most prevalent mineral extraction activities on the grasslands of Region 3 consist of oil and gas leases, which occur on several Ranger Districts of the Santa Fe and Carson National Forests and on the National Grasslands, as well (Linden personal communication 2001). In some cases, these leases have been in effect for 50 to some 100 years. The government receives a 12.5 percent royalty from producing oil and gas wells, with approximately 50 percent of revenue from public domain lands returned to the county and 25 percent of revenue from acquired lands (such as the majority of the National Grasslands) going back to the county. The Jicarilla Ranger District of the Carson National Forest, primarily grazed woodland/canyon country, is located within a major oil and gas field in the San Juan Basin with an estimated 630 producing wells (Linden personal communication 2001). Coal bed methane is also extracted from some 150 to 200 of the producing wells. In fiscal year (FY) 2000 oil and gas production on the Carson generated $9,168,935.69 in rents and royalties. The Cuba and Coyote Ranger Districts of the Santa Fe have about 60 oil and gas wells located mainly in sagebrush mesa country. The forest as a whole reported $287,905.71 in rents, bonuses, and royalties (USDA Forest Service 2000, Linden personal communication 2001).

Concerning the National Grasslands themselves, the Kiowa and Rita Blanca report no drilling on the Kiowa portion in New Mexico with minimal drilling (about two wells) on the Rita Blanca in Texas (Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999, Linden personal communication 2001). The Black Kettle and McClellan Creek, on the other hand, report that most of the land tracts within the High Plains and Redbed Plains Geographic Areas (approximately 21,500 acres) are under oil and gas leases effective for 10 years unless a producing well is drilled. If a producing well is drilled, the lease remains effective as long as the well produces. The lands in the Lake Marvin (577 acres) and Lake McClellan (955 acres) Geographic Areas are also under oil and gas leases with producing wells. Both Marvin and McClellan are under private development; the Forest Service does not own the mineral rights, although the facilities are located on Forest Service land. Overall, the Black Kettle and McClellan Creek listed $1,425,083.82 in rents, bonuses, and royalties from oil and gas operations in FY 2000 (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000, USDA Forest Service 2000, Linden personal communication 2001).

Issues and controversies related to mining on public land vary according to the type, extent, location, and management of the extraction activity. Extraction and processing can cause environmental impacts through earth removal and the distribution of metals and chemicals. Removed earth can obstruct watercourses resulting in redirection and/or downstream erosion. Both groundwater and soil may be impacted by acid tailings and processing chemicals percolating downward into the water table (Dean 1982:1-10, discussed in Hadley and Sheridan 1995: 62-65). Indirect impacts come from increased settlement and mine working activity including building roads, hauling ore, and clearing and leveling home sites. Historically, woodcutting for fuel, construction, and ore processing had significant impacts on forested lands surrounding mine sites in the region. In many cases, impacts from historic mine sites are still apparent today. Hadley and Sheridan (1995: 64-65) describe the effects of mining at some of the historic southern Arizona locations in terms of seriously altered topography from earth removal, denuded lands on which trees have not regenerated, and eroded, downcut watercourses.

Although recent years have seen growing opposition to mining on public land, the Forest Service has little authority to deny mining on its lands. Congressionally designated Wilderness areas, however, have been closed to mining exploration since 1984 under provisions of the Wilderness Act of 1964 (Robinson 1975: 158-160; Roth 1997: 238-242). States hold the permitting regulations for mining, and the New Mexico regulations are stringent in terms of bonding, close-out plans, and engineering plans (Linden personal communication 2001). Many of the small operators cannot meet the standards and are forced out of business. Those that remain can cause difficulties for Forest Service administrators in terms of obtaining sound operating and reclamation plans and proper environmental clearances.

Although the larger mining concerns generally contract for their own environmental work, Forest Service administration and review of these efforts can still be costly in terms of time and personnel. As an example, the proposed Carlota Copper Mine on the Tonto National Forest, with an environmental impact statement prepared by a private company, still required the time and effort of considerable numbers of Forest Service employees for review and comment. There are also ongoing monitoring and clean-up programs for abandoned mines that are on Forest land or on lands bordering National Forests. These can be both costly and time consuming (Linden personal communication 2001).

Oil and gas leasing follows the standards set out in the previously mentioned Surface Operating Standards and Uniform Format for Oil and Gas Lease stipulations developed by the BLM/Forest Service Rocky Mountain Regional Coordinating Committee (USDI Bureau of Land Management and USDA Forest Service Rocky Mountain Regional Coordinating Committee 1989). Stipulations cover surface uses such as road construction, drilling and producing operations, reclamation, and abandonment. The density of wells, referred to as well spacing, is regulated by the State based on efficient draining of underground reservoirs. Spacing varies by formation, depth, and the kind of resource being exploited and is generally one well per 80, 160, or 320 acres. Many of the older leases, especially those on the Jicarilla Ranger District, were approved over 50 years ago and do not fall under the contemporary standards. In these cases, operators are "encouraged" to conform to the current guidelines, with greater success reported for the larger, richer operators than for the smaller ones (Linden personal communication 2001).

The timing and location of drilling must be consistent with cultural resources, wildlife, and other resource values in the vicinity, as is the case with other projects on Federal lands. Drilling takes 2 weeks to a month with the resulting well in production for many years. Impacts from drilling and production can include habitat fragmentation from road density, as well as disturbance from truck traffic hauling out water and oil and servicing the well. Noise from both road traffic and compressors used to pump is an additional disturbance (Linden personal communication 2001). Research is still needed to assess the standards for density of surface mining structures (well spacing) and the impacts of general operations on the varying grassland ecosystems and human cultural values of the area (Seescholtz personal communication 2000).

Recreation on Forest Service Lands

Recreation in the National Forests of the Southwest has a long history and includes many varied activities such as camping, hiking, picnicking, trail biking, hunting, fishing, water sports, and viewing wildlife and historic sites. In recent years, participating in volunteer archeology projects such as Passport in Time and heritage tourism activities, such as Heritage Expeditions, have increased in popularity. Since World War II, recreation has played an extremely significant role in National Forest System management by virtue of the large numbers of people engaged in recreational activities on public lands. Indeed, the economic impacts of recreation outweigh those from traditional resource extraction activities (discussed in Menning and Raish 2000, among others).

The national trend toward increased recreation (Boyle and Samson 1985, Flather and Cordell 1995) is evident in the Southwest in the rising numbers of recreationists in the Forests in Arizona and New Mexico (Raish and others 1997: 38). Although a great amount of recreational activity occurs in the ponderosa pine forests of the region, grassland areas also contribute to the trend in rising recreational use. Holechek and others (1998: 15) note:

> The large human population increases in the United States since the 1940s have made rangelands increasingly important as places for people to engage in outdoor recreational pursuits. Hiking, camping, trail biking, picnicking, hunting, fishing, and rock hounding are some of the important recreational uses of rangelands. The importance of open space, scenery, and aesthetic values from rangelands in the United States is difficult to quantify.

Recreation use increased in Region 3 (including the National Grasslands) from 1992 to 1995 with over 40 million visitor days recorded in 1995 (Raish and others 1997: 38, fig. 8). Most visitors were viewing scenery, camping, picnicking, or swimming. Hiking, horseback riding, and river rafting have increased rapidly (Flather and Cordell 1995), while hunting, fishing, winter sports, and resort camping have remained fairly stable. Nonconsumptive wildlife recreation, such as bird watching, has also increased (Raish and others 1997: 38, fig. 8), as has visiting historic sites and heritage tourism in general.

In 1996, wildlife-related recreation expenditures tied to National Forests totaled $6.8 billion, with anglers spending $2.7 billion, participants in nature-related activities spending $2 billion, and hunters spending $2 billion (NatureWatch 1996). In the same year, wildlife watching expenditures/sales totaled $426.9 million in Arizona and $223.2 million in New Mexico (NatureWatch 1996). With respect to fishing in the Southwest, the following information was drawn from the Region 3 Forest Service Web site (www.fs.fed.us/r3, select Recreation, FishUS, September 2001):

> The uniqueness of the Southwestern Region's fisheries is a valued resource. Fishing…is big business, critical to the rural economies of the arid Southwest. The streams and lakes of the Southwest provide 10 percent of all angling (3.9 million angler visits per year) in national forests and grasslands across the nation. Fishing is so popular in the Southwest that no other Forest Service region has the fishing pressure per surface acre of water (more than 300 hours of fishing per acre annually on 48,735 surface acres). According to the latest United States Fish and Wildlife Service survey (1996), fishing in the Southwest brings in more than $550,000,000 annually. More than 4,000 jobs are directly related to fishing enterprises, with another 10 to 15 thousand jobs indirectly associated with fishing in Arizona, New Mexico, and Texas…In Arizona 54 percent of the fish habitat is in the national forests; in New Mexico, 25 percent.

This section covers fishing, hunting, and wildlife viewing—all major recreational activities along the rivers and lakes in the grassland areas of the region.

Examination of recreation trends on all the forested lands of the region is well beyond the scope of this discussion, so the focus here is on recreation in the grasslands. Regionally, grassland recreation occurs on the Apache-Sitgreaves National Forest in the montane grasslands around Big Lake and within the Escudilla Wilderness, as well as at artificially created lakes in the Tonto Basin (Tonto National Forest), although much of this latter area is Sonoran Desert. The Valle Vidal (Carson National Forest) and the Pecos and San Pedro Parks Wilderness areas (Santa Fe National Forest) have considerable acreages of montane grassland that provide recreation, elk hunting, and livestock grazing (Moir personal communication 2003). According to Brent Botts, Recreation Assistant Director for the Southwestern Region, a request was sent to the Forests of the Region for information on recreational activities and issues on their grassland area, but little additional information was received except from the officially designated National Grasslands—Kiowa/Rita Blanca and Black Kettle/McClellan Creek (Botts personal communication 2001). Hence, these areas form the basis of the present review.

Visitor use figures are reported by Forest and are not separated out by grassland portions of specific National Forests. National Visitor Use Monitoring reports for 2000, 2001, and 2002 were sought for the Kiowa/Rita Blanca and Black Kettle/McClellan Creek National Grasslands to determine visitor days for these units, but figures for these areas are included with figures for the Cibola National Forest (which manages the Grasslands) as a whole and are currently not broken out for the National Grasslands themselves.

Recreation use on the three geographic areas of the Kiowa/Rita Blanca includes both developed and dispersed sites. There is one developed campground in Mills Canyon located adjacent to the Canadian River, and three developed day-use areas on the Southern Prairie, including two picnic grounds and an interpretive hiking trail associated with the Santa Fe Trail. There are no developed facilities in the Mills Upland Geographic Area (Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999).

The majority of recreation use on the Kiowa/Rita Blanca is dispersed. In the Mills Canyon Geographic Area hunting and fishing are the two primary uses, which include hunting bear (*Ursus americanus*), Barbary sheep (*Ammotragus lervia*), mountain lion (*Felis concolor*), deer (*Odocoileus hemionus*), (scaled) quail (*Callipepla squamata*), dove (*Zenaidura macroura*), and pronghorn antelope (*Antilocapra americana*), as well as warm-water fishing in the Canadian River. There is also dispersed camping, hiking, picnicking, water play, and scenery and wildlife viewing. Mills Canyon has been designated a New Mexico Wildlife Viewing site in cooperation with the New Mexico Department of Game and Fish. Mills Upland has only dispersed recreation with hunting deer, quail, dove, pronghorn antelope, and coyote (*Canis latrans*) the main use. Other recreational uses of the area include camping, hiking, picnicking, viewing scenery, and wildlife watching. Southern Prairie also has a predominance of dispersed recreation featuring the same types of uses as the other areas (Kiowa and Rita Blanca National Grasslands Geographic Area Assessments 1999).

Although three developed camping sites exist within the High Plains and Redbed Plains Geographic Areas of the Black Kettle/McClellan Creek National Grasslands, the majority of recreation use is dispersed, with an emphasis on hunting and fishing as described for the Kiowa/Rita Blanca. Other uses include developed and dispersed camping, hiking, bird watching, and horseback riding. The Black Kettle is considered one of the best public hunting locations in the country for Rio Grande turkey (*Meleagris gallopavo intermedia*) and bobwhite quail (*Colinus virginianus*) and is the largest public hunting area in the western half of Oklahoma. It is also the only public hunting unit in this region where an over-the-counter permit can be purchased. These qualities have led to problems with hunting pressure during some seasons. Surveys found one hunter per 54 acres on the Black Kettle during the opening day of the 1999 deer rifle season, leading to a potentially unsafe situation for the public (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000).

Another problem related to dispersed recreation use has been indiscriminate driving on interior (unpaved, undeveloped) roads and indiscriminate parking in undeveloped areas. These activities have impacted soil, water, and wildlife resources. Holechek and others (1998: 446-447) and Payne and others (1983) discuss the damage that unregulated, off-road vehicle travel can cause to soils and vegetation under different moisture regimes and during different cycles of the growing season. They conclude that unregulated off-road vehicle travel can be as destructive as uncontrolled grazing. They recommend regulation of recreation use on rangelands, just as other uses are regulated (Holechek and others 1998: 447). The High Plains and Redbed Plains Geographic Areas have moved in this direction by creating designated parking locations for dispersed recreation, which are designed to reduce vehicle access and encourage walk-in use. The areas have also instituted road closures in conjunction with the parking spaces for the same purpose (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000).

Recreation on the Lake McClellan and Lake Marvin Geographic Areas (in Texas) tends to be considerably more developed than on the High Plains and Redbed Plains, although dispersed recreation is also present. Lake Marvin offers warm-water fishing, developed camping (two campgrounds and a recreation building), picnicking, hiking, and bird watching. The primary uses are fishing and camping, but hiking the trails and viewing birds and other wildlife are also popular. Hunting is prohibited owing to the area's small size (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000).

Developed recreation facilities in the Lake McClellan Area are managed under a concessionaire permit and consist of several campgrounds and picnic grounds, as well as a store offering rentals such as paddleboats and lifejackets. Activities at the lake include camping, picnicking, fishing, hiking, and wildlife viewing, as well as boating, jet skiing, bicycle riding, and motorcycle riding. The north end of the area is open all year, while the south end is closed from November 1 to March 31 to protect eagle roost habitat. Dispersed recreation throughout the area consists of fishing, boating, water play, camping, and motorcycle riding; fishing is the primary dispersed use. Hunting is generally not allowed. Motorcycle riding is permitted by the concessionaire and is popular on the motorized trail system. It is, in fact, the primary use of the trail (Black Kettle and McClellan Creek National Grasslands Geographic Area Assessments 2000). No problems were mentioned concerning the impact of motorcycle use, which is apparently being controlled and regulated by designation of the motorized trail.

As discussed previously, however, off-road vehicle use by recreationists is growing in extent and numbers of occurrences and has considerable potential for negative environmental impacts. All-terrain vehicles (ATVs), in particular, are growing in popularity, and their use is on the rise in many areas, with the potential for serious environmental damage. The growing use of off-road vehicles has prompted a study by the San Dimas Technology and Development Center (USFS) on the resource impacts of off-road vehicles on soil, water, air, vegetation, wildlife, and aquatic species of National Forests throughout the country. The ongoing project is a survey examining types of uses, equipment, resource impacts, and use regulations.

A Nevada study examined by Holechek and others (1998: 446-447) compared the effect of motorcycle, four-wheel-drive truck, and no traffic on infiltration rate and sediment production. In the study, infiltration rates were decreased and sediment production was increased by both forms of traffic, with the four-wheel-drive truck having considerably greater impact than the motorcycle (Eckert and others 1979). As populations and urbanization increase in the Southwest, all forms of recreational use will increase on the region's rangelands. Assessing, regulating, and controlling the effects of these uses will become increasingly important. The population growth, with its attendant urbanization and suburbanization, are the topics of the final section of this chapter.

Continuing/Future Trends in Southwestern Rangelands and Rangeland Management

Conversion of rangeland to nonagricultural uses during the past 20 years is becoming an important concern for public land managers and private ranchers and farmers. In the United States, this trend primarily results from human population increases and a demographic shift from the eastern to the western portion of the country (Holechek 2001: 39). World population projections show the North American population rising from 172 million in 1950 to 297 million in 1995 with a projected increase to 384 million in 2050 (Holechek 2001: 39, table 1; McQueen 2000, United Nations 1998). In a listing of States with the greatest population growth rate from 1990 through 1994, Arizona ranks third and New Mexico ranks eighth (Mitchell 2000: 25, table 2.9; Riebsame and others 1997). Increasing urbanization and suburbanization result from both population growth and inmigration from other areas. For example, Bahre (1995: 243-244) describes southern Arizona:

> The rapidly growing population of Arizona, especially since the 1940s, has led to expanded urban and rural development on privately owned lands and is threatening federal and state trust lands, which are continually being sold to the public for development. Since 1950 the number of rural sub-divisions being built in southeastern Arizona has exploded,

> especially in Cochise County, which has both the largest area of grassland and the largest amount of private land (41 percent)—almost all of it being used for agriculture, ranching, or subdivision development (Hecht and Reeves 1981). Cattle numbers in the county have declined, in part because of tract development in former rangelands, and much private grazing and agricultural land has been purchased by people seeking rural retirement or investment opportunities.

It often seems that the urban residents, as well as those newly arrived in the region, have different land attitudes and values from many of the long-time rural residents. These differing views can lead to conflict over land use and land management goals and strategies. Conflicts over environmental protection legislation and the role of organized environmental groups are examples. Many present-day environmental issues and concerns in the Southwest are aligned with a national shift in public attitudes and values concerning the appropriate use of public land, primarily in the West (Macon 1998). Many urban Americans now hold environmental protection oriented public land values, in contrast to the commodity and community economic development orientation of the earlier conservation era (from approximately 1900 to 1969) (Kennedy and others 1995, Macon 1998). As Kennedy and others (1995) point out, rural agricultural communities use and view land differently from urban groups, thus producing contrasting perceptions and values. A large amount of contemporary conflict over timber, wildlife, and rangeland issues stems from disjunct rural and urban values concerning human relationships with nature and its uses.

Nature values are not intrinsic but are human creations originating in the minds of individuals and groups as their changing perceptions and needs interact with environmental, political, and economic systems (Kennedy and others 1995: 128). For example, the conservation movement of the early to mid-20th century fit well with the needs of an industrialized nation for sustained-yield timber and forage production to provide commodities for growing factories and cities. Even though recreational, biocentric, and esthetic values were a part of some early conservation visions, these views did not become a dominant force in public attitudes until the 1960s, with the advent of an urban, postindustrial society and the environmental movement (Kennedy and others 1995).

The later 1960s and 1970s saw passage of the main body of environmental protection legislation. Mitchell (2000: 7) in a discussion of legislation affecting rangelands stated:

> From a legislative context, the 1970's could be considered the decade of the environmental movement. The Wilderness Act was enacted in 1964. Then, starting in January 1970 with the Environmental Policy Act of 1969, no fewer than 12 major environmental laws affecting the conservation and management of U.S. rangelands were signed into law during the following 10 years. Among such laws were the Resources Planning Act of 1974 (RPA) and the National Forest Management Act of 1976 (NFMA), which called for a recurring assessment of America's forest and rangeland situation.

Other pertinent legislation includes the landmark Endangered Species Act of 1973, the Federal Land Policy and Management Act of 1976 (FLPMA), the Forest and Rangeland Renewable Resources Research Act of 1978, the Public Rangelands Improvement Act of 1978, and the Archeological Resources Protection Act of 1979.

The orientation of range research has also changed over the years from a more traditional range/livestock grazing system approach to a more integrated ecosystem approach. Contemporary research now includes more ecologically based subjects such as watershed and riparian area management, biodiversity, disturbed site reclamation, and wildlife/livestock interactions, to name a few (Evans 1990, Everett 1992). Additionally, studies concerning grazing on public land now include information on visitor attitudes about grazing (Mitchell and Fletcher 1996), which indicates the increasing role of public concern and input into land management agency decisionmaking.

One result of the growing role of both public and interest group involvement in Federal agency planning and decisionmaking is increased litigation over agency projects, programs, and plans. A rising number of appeals and lawsuits from both environmental and industry groups have resulted from the agency's efforts to bring livestock grazing into compliance with Federal environmental statutes. For example, during 2001, there were 11 grazing-related lawsuits against the Southwestern Region of the Forest Service (Gonzales personal communication 2002).

The increasingly litigious environment surrounding rangeland management has spurred the rise of regional conflict resolution groups such as the Malpai Borderlands Group in southeastern Arizona and southwestern New Mexico, the Diablo Trust in northern Arizona, the Arizona Common Ground Roundtable, and the Quivira Coalition in New Mexico, to name a few (Raish 2000c, Ruyle and others 2000, Sheridan 2001). These groups attempt to protect both the environment and traditional lifeways and to diminish conflict, controversy, and litigation. They often include environmentalists, ranchers, and public land managers and espouse collaborative stewardship, mutual education, and citizen participation in agency decisionmaking. They emphasize that sound land management and healthy ecosystems can coexist with traditional rural economic practices such as ranching (Guess 1999).

Ruyle and others (2000) describe the work of the Malpai Borderlands Group and the Diablo Trust to coordinate management of multiagency lands as a promising means of maintaining the viability of ranching on public land, protecting open space, and reducing habitat fragmentation. The Malpai Borderlands Group has worked with the Nature Conservancy to protect and restore wildlife habitat and with public land managers in projects to restore fire to the ecosystems of the area. They have also developed a program of conservation easements to protect open space from future development and subdivision (Clifford 1998, Guess 1999).

These groups focus on the threat of growing development and suburbanization to Western lands and landscapes, as escalating rates of development are impacting both private lands and adjacent public lands. U.S. Department of Agriculture statistics show that the nation lost almost 1.4 million acres a year to development from 1982 to 1992. In the mid-1990s the rate more than doubled to almost 3.2 million acres per year (Rome 2001: 264). Land-use change from population growth is greater in rural areas than urban ones because of the dispersed nature of exurban development. Exurban development refers to rural residential development that occurs beyond incorporated city limits and often results from the subdivision of ranches into smaller parcels for home sites ("ranchettes") generally ranging from 1 to 20 acres or so (Sullins and others 2002, Theobold 2000). Almost 80 percent of the land used for houses built between 1994 and 1997 was in nonmetropolitan areas (Heimlich and Anderson 2001). In addition, locations in proximity to public land are particularly desirable for development (Riebsame and others 1996, Swanson 2001).

Sheridan (2001: 146-147) describes development, which has escalated since the late 1960s, in the Sonoita-Elgin area of southern Arizona plains grassland near the Coronado National Forest. In 1989 Sonoita had approximately 400 homes, which had risen to 707 by 1995, a 76 percent increase. Numerous ranches have been subdivided in the region in the past 20 years, leading to segmentation of the open grassland into ever smaller parcels (the minimum lot size is currently 1 to 3 acres). Sheridan (2001: 146-147) discusses the ways in which the resulting land use and human occupation can affect wildlife, native vegetation, and soil cover. Problems can include house and outbuilding construction, fencing (which inhibits the movement of large wildlife), removal of native vegetation, introduction of exotics, and introduction of domestic pets that prey on or harass wildlife. This type of land fragmentation acts against maintaining the biodiversity that stems from large, unfragmented ecosystems, as wildlife habitat and corridors that link grasslands with montane use areas are fragmented or destroyed. There is also a growing constraint on large-scale ecosystem management, such as the reintroduction of fire as a natural process or management tool, which is generally resisted as a threat to homes and property.

According to James H. Brown, a University of New Mexico biologist and past president of the Ecological Society of America, "Far more habitat has been destroyed to provide water for cities, subdivisions, and irrigated (commercial) agriculture than by even the heaviest grazing pressure" (quoted in Clifford 1998). In an effort to explore this notion of the negative effects of development, Maestas and others (2002) conducted a study designed to examine wildlife species and plant diversity on ranches, protected areas (Colorado Division of Wildlife's State Wildlife Areas), and exurban developments in the foothills of the Colorado Front Range of the Rocky Mountains northwest of Fort Collins. The wildlife species studied were mammalian carnivores and songbirds. The study found that biodiversity differed among the different land use categories, with wildlife species occurrence and densities more similar between ranches and protected areas than on exurban developments. In terms of plant communities, the most nonnative species were found on exurban developments, with more species of native plants found on ranches than on the other two land uses. The most common nonnative was cheatgrass. However, the cover of native species did not differ statistically across the land uses (Maestas and others 2002). The information derived from this study has considerable implications for conservation efforts. Maestas and others (2002: 27) stated:

> A generalization from our study is that there is an increase in human-adapted wildlife and non-native plant species with exurban development. Interactions among native, non-native, and human-adapted species could result in the simplification of the Mountain West's natural heritage favoring species whose evolutionary life histories allow them to exist with humans. This change has negative implications for the maintenance of biodiversity at both the site and landscape scales and its consequences are increased with increasing development (Knight 2002).

This chapter has shown how Southwestern grasslands have been impacted by human activity for thousands of years. Native American groups used and modified the area for hunting, gathering, and agriculture before and after the arrival of Spanish colonists in the 1500s. European settlement brought a wide array of new domesticated plants and animals, as well as new technologies and land-use strategies. However, changes and impacts to the area's land base accelerated more rapidly with commercialization of both farming and ranching after U.S. takeover, with development of railroad transport and wider national markets. In the years since the beginning of the 20^{th} century and especially after World War II, conservation and preservation efforts have come to the fore as concerns for ecosystem health and sustainability have increased as national priorities. New challenges now face the region in terms of population growth and inmigration resulting in a proliferation of urban, suburban, and exurban developments, affecting rural land uses, water supplies, and wildlife habitat. Sheridan (2001: 147) notes: "Some human impacts can be reversed but subdivisions are forever." Managing this new challenge of human population growth and development will undoubtedly be a major effort of public land managers, environmental groups, and farmers and ranchers into the 21st century.

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Chapter 6:

Historic and Current Conditions of Southwestern Grasslands

Introduction

Southwestern grasslands today share general differences from their pre-Euro-American settlement conditions. With few exceptions, grasslands—whether in the desert, prairie, or mountains—were, prior to non-Indian settlement, more diverse in plant and animal species composition, more productive, more resilient, and better able to absorb the impact of disturbances. Southwestern grasslands today are missing the elements of disturbance regimes that kept them functioning in the prelivestock period. Such interruptions include human activities leading to loss of palatable plant species and keystone grazers such as the buffalo and prairie dog. The loss of most fire events resulted in the accelerated loss of soil from wind and water erosion, including loss of the highest productivity sites from gully formation. Soil changes also include loss of soil pore space through compaction, decrease of soil organic matter, retention of surface water, and nutrient cycling processes whether permanent or temporary. The result is a loss of large-scale connectivity by isolating grasslands through urbanization and other anthropogenic influences.

The frequency, magnitude, and extent of these changes vary among major kinds of grasslands and among landscapes and plant communities occurring within them. But almost everywhere in the Southwest, the amount of change is vast. Perhaps if early alarms voiced by Wooton in 1908 in New Mexico, Griffiths in 1901, and Thornber (1910) and Jardine and Hurt (1917) a few years later had been heeded, much of the subsequent disruptive change could have been averted—at least in those places not already degraded too far to recover and resemble their former state.

For example, Humphrey (1987) and Turner and others (2003) photographed landscapes along the Southwestern United States-Mexico boundary to document changes in vegetation as seen in prior historical photos. From the severely degraded conditions in the 1890s and the limited amount of recovery since, many irreversible changes had already taken place in the unforgiving climate along the border. Farther north, Aldo Leopold wrote in 1924 (p. 8), "[In] northern Arizona there are great areas where removal of grass by grazing has caused spectacular encroachment of juniper on park areas. But here again both grass competition and fire evidently created the original park, and both were removed before reproduction came in." Many thousands of pages have been written since that time, with sound recommendations for moving Southwestern grasslands back toward ecosystem sustainability. Much progress has been made toward recovery, but still the gap widens across tens of millions of acres.

With the exception of older Euro-American occupations such as in the Rio Grande Valley, most change has taken place since the end of the Civil War and the beginning of the cattle era. The cattle boom began in 1865, reached its height about 1885, and gradually adjusted to an environment that no longer included a frontier. Between 1875 and 1883, much of the grassland was fenced with barbed-wire (Hollon 1961). The advent of the railroads increased the extent, area, and rate of change, shrinking grasslands, as they once were, into smaller, relictual fragments, and relegating them to remote locations, or at least locations remote from water. The 1936 Department of Agriculture report to the Senate titled "The Western Range" documented the widespread degradation of the grasslands. Some 553 million acres of Western rangeland, 76 percent of the total, were continuing to degrade at the time of the report (USDA 1936). Much of the upland deterioration has been halted, with restoration of adequate vegetation cover to stem upland erosion over at least a majority of the grasslands. However, the creation of gullies had effectively and almost universally dewatered the grasslands with the exception of the flatter portions of the shortgrass prairie. The gullies reduced the effectiveness of the precipitation across the grasslands with some loss of cienega and sacaton communities. Recovery of the grassland cover in the uplands has occurred and continues, but without parallel restoration of the incised valley floors and basins.

Pre-Euro-American settlement grasslands are difficult to describe accurately. Qualitative and descriptive information must be pieced together and inferences made from inventories of isolated and protected landscape fragments, pastures managed for good and excellent rangeland health, properly functioning watersheds, historical documents, life-history studies of plants and animals, professional experiences of those working and doing research in grassland ecosystems, and documented accounts of early explorers and land survey records.

Reference conditions that describe what any sustainable historic grassland community should ecologically represent are difficult to achieve. Even the most homogeneous grassland has extremely dynamic ecological processes. Plant species readily vary with time on any given site according to rainfall patterns and the latest disturbance process. Vegetation dynamics in grasslands can be characterized by the shifting patterns of abundance and sequential species replacement over time, or succession. The complex patterning of small-scale partitioning by grassland species in less degraded conditions makes consistent classification challenging at a finer scale.

For all except the montane grasslands, increased vegetative cover and productivity of pre-Euro-American settlement grasslands are hypothesized to have increased convective warm season precipitation over today's rainfall amounts (Grissino-Mayer 1996). However, drought in earlier times was just as important a disturbance factor as it is today. Short-term deviations in rainfall amount and timing influence species frequency and occurrence, but long-term changes have an effect on grassland boundaries and extent.

The lower elevation grasslands have numerous plant species in common. Long-term climate changes enable some species to spread long distances while others retract in ebb and flow across time and space. The result leaves relictual pockets of species far from their normal associates, with big bluestem (*Andropogon gerardii*) migrating from the tallgrass prairie to the Zuni mountains of New Mexico, and black grama (*Bouteloua eriopoda*) from the desert grassland to Kansas and Oklahoma. While today's conditions favor movement of more invasive species, the more diverse and resilient systems of yesterday would have favored periodic expansion of a wider variety of species.

Because they were more diverse than the grasslands of today, all grasslands in the Southwest were historically able to withstand drought. Tilman and Downing (1994) found grassland species richness led to greater drought resistance because some grasses within the complex of species were more drought resistant than others and partially compensated for the decreased growth of the less drought resistant species. This appears to have been most prominent in the prairie grasslands, although their topographic homogeneity may have made such relationships more noticeable. The higher diversity in all grasslands would have had the similar benefits of fire, hail, insect outbreaks, and other disturbances. Even with the benefit of increased diversity, there were periodic events of sufficient magnitude to overcome the resiliency of any grassland. The common situation in grassland ecosystems, both then and now, is to be in a state of transition from some type of disturbance.

Wooton (1908) noted the New Mexico Territory included approximately 300 grasses and sedges, of which only 25 or 30 furnished the "great bulk" of livestock forage. Livestock homogenized the grasslands, masking or eliminating both local and regional differences. The ecotones of the nonmontane grasslands were naturally broad. Tolerance limits of species such as blue grama (*Bouteloua gracilis*) and sideoats grama (*Bouteloua curtipendula*) were broader than the limits of the major grasslands.

It is useful to look at the distribution of accompanying shrubs to set a practical boundary for the various grasslands. The limits of big sagebrush (*Artemisia tridentata var. tridentata*) help set a climatic and physiographic boundary for Colorado Plateau and Great Basin grasslands. Creosote bush (*Larrea tridentata*) likewise is an aid in approximating the boundary for desert grassland and shrubland. The zone of integration between *Yucca glauca* of the shortgrass prairie and the taller *Yucca elata* of the desert grassland provides a good boundary. The ecotone between the two is the home of *Yucca intermedia*, an intermediate between the two yuccas.

Montane Grasslands

Montane grasslands ranged from the alpine and subalpine regions at high elevations through the ponderosa pine (*Pinus ponderosa*) ecosystem at lower elevations. These grasslands were the most naturally fragmented and ranged in size from thousands of acres, such as those in the White Mountains of Arizona where grasslands covered about 80,000 acres above 9,000 feet (Baker 1983), to only a few acres, limited by topography or the surrounding forest. The origin and maintenance of these grasslands, except for those few acres in the alpine, are a subject of debate, perhaps because the causative factors not only vary from place to place but also usually combine to create a variety of effects. Below the alpine zone, fire and climate appear to have been the major factors in both the creation and maintenance of these grasslands. For wet meadows, fire may have been the ultimate creator of many openings; with soil moisture levels beyond what trees could tolerate maintaining them. Wind desiccation, and snow and ice abrasion readily maintained at least the larger meadows and may have interacted with drought and/or insect and disease outbreaks in the creation of some meadows. The greater level of soil organic material and higher inherent productivity of meadow communities helped grasses and grasslike plants play a major, competitve role in maintaining these grasslands as tree free.

In pre-Euro-American settlement times, most montane grasslands would have been less fragmented with a greater degree of connectivity, and their total acreage would have been greater. Almost all of these grasslands, either large or small, have either yielded acreage to forest, or disappeared altogether. Inhibition of water movement through the soil—due to compaction by livestock and ungulate wildlife—decreased soil moisture availability because of increased tree density in surrounding forests. Water diversions and road development combined to reduce the condition and extent of wet meadows.

The alpine ecosystem in the Southwest has few gentle slopes conducive to development of grasslands. Most alpine is dominated by the forb-rich fellfield community, but where the topography permits, Kobresia (*Kobresia myosuroides*) dominated communities of sedges and grasses are considered cushion plants and develop what can be termed a grassland (Andrews 1983). Baker (1983), in his study on Wheeler Peak, identified 10 alpine communities in a complex mosaic defined mainly by slight variations in topography, exposure to wind and sun, and snow accumulation. In the harsh elements above timberline, "grassland" turf is compact and complete in cover except where broken by surface rock or pockets of gopher activity. Where the sod has been broken, as in the case of heavy sheep use, or misplaced recreation trails, wind erosion unravels the turf to the rocky substrate below. The degraded area expands until a change in aspect or a surface rock boundary is reached. The loss is permanent. Old soil level marks on rocks near Santa Fe Baldy reveal near complete loss of the Kobresia community in that area. Unlike other grassland communities, fire is not likely to have played a role in either the creation or maintenance of the alpine grasslands. Pre-Euro-American settlement alpine grasslands would have covered a few thousand acres.

As with subalpine Thurber fescue (*Festuca thuberi*) grassland communities, there is considerable intermixing with other grasses such as oatgrass (*Danthonia* spp.), which tends to favor slightly less productive sites than Thurber fescue. Thurber fescue is often well in excess of a meter tall and is much sought after by large herbivores. Thurber fescue, was more of a community dominant and was more widespread in pre-Euro-American settlement times than in today's subalpine grasslands. The result has been an expansion in dominance of sheep fescue (*Festuca ovina*) from above, and Arizona fescue (*Festuca arizonica*) from below. These high elevation (more than 8,500 feet) grasslands were, and continue to be, diverse in both grasses and forbs.

Allen (1984) documented extensive reductions of the Thurber fescue grasslands in the Jemez Mountains within the previous hundred years as both the conifer and aspen forests expanded. An interrupted fire regime and a decreased grassland competitive ability due to overutilization by large herbivores have contributed to a widespread reduction in Thurber fescue grasslands. These high elevation grasslands are well adapted to periodic fire.

The Arizona fescue (*Festuca arizonica*)/mountain muhly (*Muhlenbergia montanus*) grassland community in the mixed-conifer and ponderosa pine forests were naturally less productive and less diverse than the Thurber fescue community, but far more widespread. These grasslands and their variations, such as the screwleaf muhly (*Muhlenbergia virescens*) community, were the most extensive montane grasslands of the Southwest. A few relictual areas still have mountain muhly as a significant component in lower elevation woodland openings, but how common this once was is unknown. Because these grasses were the primary grasses in the understory of the open pine forests, even the smaller forest openings had a greater connectivity than in the understory-deficient, dense pine forests of today. In 1911, Woolsey (quoted in Fletcher 1998,

p. 86) described the typical yellow pine forest of the Southwest as a "pure park-like stand made up of scattered groups of from 2 to 20 trees, usually connected by scattering individuals. Openings are frequent, and vary greatly in size. Within the type are open parks of large extent…"

Arizona fescue and screwleaf muhly are cool season species, and mountain muhly is a warm season grass. Between the two variations is a natural ebb and flow in composition in step with periodic variations in seasonal precipitation. Arizona fescue is more tolerant of fire than mountain muhly. Large Arizona fescue grasslands in what would otherwise be the upper extent of ponderosa pine forest, such as Hart Prairie near Flagstaff, appear to have burned with a higher frequency. This favored Arizona fescue over mountain muhly and limited the density of surrounding ponderosa pine, resulting in an extremely open forest. The 1904 inventory of forest conditions on what is now the Coconino National Forest (Leiberg and others 1904, p. 35) noted "large parks occur in townships 21 and 22 north, ranges 4 and 5 east, one of them containing 16,000 acres."

The cool season growth of Arizona fescue played a large role in maintenance of the larger parks and smaller openings by directly competing with ponderosa pine seedlings. Maximum growth of ponderosa pine coincides with maximum growth of Arizona fescue. Pearson (1949) reported that numerous ponderosa pine seedlings had started in an Arizona fescue opening in the prolific seedling year of 1919, but all had died within a year. Where Arizona fescue was suppressed or reduced by cattle grazing, the survival of ponderosa pine seedlings increased, and a point was quickly reached where ponderosa pine dominated. Today, Arizona fescue still continues to decrease in the dense forest understory across much of the Southwest.

Before the replacement of most wet meadow plant diversity with Kentucky bluegrass (*Poa pratensis*), wet meadows were typical for all but the narrowest and steepest drainages in Southwestern forested communities. The expansion of Kentucky bluegrass was made possible by heavy livestock use in the wet meadows. Compaction of wet meadow soils, particularly during dry years, inhibited water movement below ground, effectively shrinking the wet meadow and resulting in the elimination of riparian vegetation and favoring expansion of Kentucky bluegrass. Kentucky bluegrass has also replaced native grasses in some of the upland meadows (Dick-Peddie 1993).

Desert Grasslands

The climatic potential for desert grasslands has been more or less similar to that of today for the past 4,000 years (Van Devender 1995). Desert grassland in the pre-Euro-American settlement era occupied extensive acreage across the Southwest between 3,000 and 5,000 feet elevation. (Martin 1975). The grasslands were largely free of "brush," and streams and rivers dissecting the grasslands were lined with galeria forests and marshes (cienegas) (Bahre 1995). Just how much some portions of the desert grassland have changed can be visualized from a description of southeastern Arizona in the mid-19th century:

> The valley bottoms were covered by a dense growth of perennial sacaton grass, oftentimes as high as the head of horseman and so thick and tall that cattle, horses, and men were easily concealed by it. The uplands were well covered with a variety of nutritious grasses, such as the perennial black grama, and the many annuals that spring into growth during the summer rainy season. The abundant vegetation, both on highlands and in valley bottoms, restrained the torrential storms of the region so that there was no erosion in valley bottoms. Instead the rainfall soaked into the soil and grew grass.
>
> Sloughs and marshy places were common along the San Simon, the San Pedro, the Santa Cruz and other streams, and even beaver were abundant in places where it would now (1919) be impossible for them to live. (USDA 1937, p. 23)

Another reference from the same era noted beaver on the Rillito and on the Santa Cruz where they had "many dams that backed up water and made marshy ground" as far down as Tucson (USDA 1937, p. 11).

Conditions began to change rapidly with the buildup of the livestock industry. Harvest of native grasses as hay contributed to the decline. Bush muhly (*Muhlenbergia porteri*) was sufficiently abundant that in 1879 and 1880 hundreds of tons were delivered as hay to military posts in Arizona (USDA 1936). The summer of 1885 was unusually dry, and 1886 had half the normal rainfall (Hastings 1959). The major portion of the desert grassland received between 8 and 14 inches of precipitation annually with a small segment in southeastern Arizona and a portion of southwestern New Mexico's bootheel, receiving approximately 14 to 20 inches.

Typically, the uplands within the grassland yielded runoff in summer thunderstorms, irrigating the alkali sacaton (*Sporobolus airoides*) and tobosa (*Pleuraphis mutica*) in floodplains below. Except during major thunderstorm events, the desert grassland likely offered only a slight contribution of water to perennial streams and rivers. Vegetation along most drainages and in the floodplains would have been sufficient to reduce peak flows originating in the uplands.

Much of the historic literature addressing the desert grassland expresses some concern about the expansion of desert shrubs. Pre-Euro-American settlement desert grasslands occupied large areas today covered at least in part by creosote, burroweed (*Isocoma tenuisecta*), mesquite, and other shrubs undesired by the livestock manager.

> Many plains and uplands of the Chihuahuan Desert in southern New Mexico are in transition from a perennial desert grassland to a mesquite dominated shrubland. Shrubs encroach into grasslands by individual plants becoming established into islands of mesquite with a greater biodiversity than the surrounding grasslands. The mesquite islands eventually coalesce into mesquite "front" and biodiversity decreases. (Beck and others 1999, p. 84).

Similar increases in shrubs were seen 3,900, 2,500, and 990 years ago. Shrub increases late in the 19th century were a natural response to drought. This response differed from earlier episodes due to the additive factor of massive numbers of livestock (Van Devender 1995).

York and Dick-Peddie (1969) investigated survey records beginning in 1858 for 31 townships in southern New Mexico and found they had changed from approximately 75 percent covered with grass to less than 5 percent grass cover at the time of the study in the late 1960s. One of their findings yielded a clue on the origins of the spread for mesquite in the uplands. They found a detailed 1882 map of the vegetation west of Las Cruces. Scattered across the mesa were scattered pockets of mesquite. Each of those "small locations" was found to have a Native American campsite near its center. The Native Americans used mesquite extensively for food and later as food for their horses. Other sites such as breaks on the edges of watercourses also were found to have a mesquite component.

The drier portions of desert grassland in New Mexico were apparently less adapted to fire than desert grasslands of southeastern Arizona where the higher precipitation provided greater fuels and connectivity for fire to spread (Bahre 1985). Tobosa and the sacatons are well adapted to fire, but these grasses favor topography where moisture accumulates. The typically patchy vegetation of drier, upland black grama sites would have restricted fire spread. In periods of greater than average rainfall, the connectivity of vegetation would have expanded, increasing the likelihood of fire in black grama dominated communities. Black grama does not fare well after fire with rainfall amounts typical of most of the desert grassland. A prescribed burn in black grama was conducted on the Bernalillo Watershed (White and Loftin 2000) near the northern limits of the desert grassland in November 1995 and January 1998. In May of 2001, marked differences were still evident between burned areas previously dominated by black grama and adjacent unburned black grama stands. Neither area had been more than minimally grazed by trespass livestock in several decades. At a landscape scale, infrequent and patchy burns in drier portions of the desert grassland would have enhanced diversity. Desert grassland developed under conditions of lower fire frequencies than other Southwestern grasslands. Fire events followed by wet years would have little lasting impact on community structure, but when followed by drought, the result could be a long-term change in community structure (McPherson 1995). The greatest threats to existing Southwestern desert grasslands include uncontrolled grazing, desertification, introduction of exotic and invasive species, and urban development (Havsted 1996).

Unique and isolated sites such as Dutchwoman Butte, and Research Natural Areas such as Otero Mesa, Bernalillo Watershed on the Cibola National Forest, and Rabbit Trap on the Gila National Forest, would be useful in establishing reference conditions for some of the communities within the desert grassland. Of greater importance are larger areas such as the Appleton-Whittell Research Ranch in southeastern Arizona where fire has been maintained. The most severely degraded elements of desert grasslands, such as the giant sacaton community, are hardest to find in suitable condition for use in characterizing reference conditions. The extreme variability of desert grasslands across its range makes application of any standardized reference condition problematic at best.

Great Basin Grasslands

Brown (1982) recognized grasslands above the Mogollon Rim in Arizona as having an affinity with those in the Great Basin. However, in our (this current) assessment they are being referred to as Colorado Plateau grasslands. Therefore, in the context of this current assessment, the Great Basin grasslands ranged southeast to the Rio Grande Valley from just south of Albuquerque, north to the Colorado border. They also extended at least as far south as the San Augustin Plains in New Mexico. Brown (1994) called them Great Basin Grasslands to emphasize that influence, but believed they were transitional between his Plains grassland to the east and the true Basin and Range communities to the west and northwest.

Weddell (1996) reported that summer drought was the primary factor excluding large herds of bison from the steppe of the Intermountain West. As a result, there was a lack of significant selective pressure from large herbivores, limiting the co evolution between ungulates and Intermountain grasses. This left native grasslands vulnerable to invasion by exotic grasses (particularly cheatgrass, *Bromus tectorum*) when livestock grazing degraded these grasslands. "The Western Range" (USDA 1936) cited a Forest Service study in western Utah where during the 1931 to 1934 drought a 20 percent decrease in forage plants occurred on ungrazed plots, but the reduction was 60 percent on nearby overgrazed areas.

One situation that makes grassland particularly vulnerable to degradation is a higher complement of cool season grasses, which are more sought after by livestock and are less resistant to grazing than warm season grasses, coupled with the soils prone to erosion across the majority of the Great Basin grassland. The complement of factors degrading other grasslands—mainly interruption of the fire regime and heavy use by livestock during dry years—may have caused greater change here than even in the desert grassland. Acre for acre, the Great Basin in pre-Euro-American settlement condition had greater hydrologic function (infiltration) and higher primary productivity than the desert grassland.

The spread and increased density of junipers by expansion onto deeper, formerly grassland soils has long been a concern (Johnsen 1962, Miller and Wigand 1994, Wright and others 1979), and the condition continues today. Even if it became politically acceptable to reintroduce fire to manage juniper (*Juniperus* spp.), sagebrush (*Artemisia* spp.), and other shrubs, the break in fire connectivity due to the myriad of incised watersheds (gullies) together with the lack of grasses productive enough to provide needed fire intensities, make restoration efforts challenging.

With the exception of some more mesic portions of the Great Basin grassland, defining reference conditions is challenging because of a reduced productivity and species diversity. The Rio Grande watershed's Rio Puerco is one of the most infamous and degraded watersheds in the Southwest (Dortignac 1960). Portions of the Rio Puerco have been grazed at least 100 years longer than most of the other Great Basin grasslands. Current trends could lead to similar conditions of other watersheds within the Great Basin grassland.

Plains Grasslands

The pre-Euro-American settlement Plains Grasslands are better understood when placed in context of the tallgrass and shortgrass components of the midcontinental expanse of grasslands. Dr. John Weaver, student of the American Prairie, shared more than 40 years of study in his 1954 book *North American Prairie*. He explained:

> When the white man came to North America magnificent grassland occupied the central part of the continent. From Texas it extended north to Manitoba, where it gave way to boreal forest. From the forest margins of Indiana and Wisconsin it extended westward far into the Dakotas and half way across Kansas. Early settlers designated this great area of waving grasses bedecked with wonderful flowers as Prairie. Beyond, even more extensive but drier and sparser grassland stretched away to the Rocky Mountains. This was early designated as the Great Plains. (Weaver 1954, p. 2).

The Prairie and Great Plains were separated according to the height of dominant grasses—in essence, their productivity.

Prairie grasses were classified into three groups according to the height they attained. Any grass that normally attained a height of 5 to 8 feet, or more, belonged to the tallgrass group. Big bluestem (*Andropogon gerardii*), switchgrass (*Panicum virgatum*), and sloughgrass (*Beckmannia syzigachne*) fell into this category. Midgrasses were 2 to 4 feet tall and included species such as little bluestem. On dry ridges and crests of hills, especially in Western prairie, were the shortgrasses, 0.5 to 1.5 feet tall. These grasses included blue grama, hairy grama, and buffalo grass. Species with "great" drought resistance, such as sideoats grama, blue grama, and buffalo grass (*Buchloe dactyloides*), or those adapted to evade drought, such as western wheatgrass (*Pascopyrum smithii*), usually occurred in most of the prairie only in small amounts because they were not normally able to compete with tallgrasses. These grasses increased rapidly and occupied large areas in intense drought (Weaver 1954).

Dr. F.W. Albertson described the typical situation in the mixed-grass prairie just before the turn of the 20th century.

> The vast majority of the land was native prairie. It was neither broken for cultivation nor overgrazed by livestock. Short grasses and low growing broadleafed herbaceous plants occupied the hilltops. Many of the hills were dotted with bunches of little bluestem and in the favored areas such as buffalo wallows, side oats grama, and big bluestem were common. The hillsides were occupied primarily by big and little bluestem, side oats grama, Indian grass (*Dichanthelium acuminatum*) and panic grass (*Sorghastrum nutans*). All but the little bluestem and side oats grama were dominant on the lowlands" (Albertson 1949, p. 10).

Dr. Albertson also noted that the dust storms of severe droughts in the 20th century, magnified by cultivating lands that could not support sustained agriculture, also occurred as natural events in severe droughts before 1900.

The higher productivity of the mixed grass prairie brought with it more frequent and higher intensity fires than fires in the shortgrass prairie. In 1924, Shantz noted tallgrass prairies were often burned in late summer or winter, and that early settlers and travelers could find safety only by starting backfires. The flames were impossible to pass through to the safety of the burned areas behind (Weaver 1954). In 1825, Joseph C. Brown was surveyor in a party marking what was to become the Santa Fe Trail. Brown noted that Cow or Cold Water Creek, near the Great Bend on the Arkansas River in Kansas, marked the beginning of the shortgrass prairie, and the shortgrass was the boundary of the annual burning of the prairie (Hollon 1961).

Risser and others (1981) included as mixed grass prairie, the strip of sandy soils with Havard shin oak (*Quercus havardii*) and midgrasses running across the Texas Panhandle into eastern New Mexico about Tucumcari. This community is also common along the southeastern border of New Mexico. In 1844, Josiah Gregg described a portion of the community as a region covered with sandy hillocks entirely barren of vegetation in places while others were covered with shin oak and plum. In 1845, Lt. J. W. Albert found the community to be a continuous succession of sandhills with sand sagebrush and Havard shin oak. Big bluestem, sand bluestem, Indiangrass, switchgrass, giant dropseed (*Sporobolus* spp.), plains bristlegrass (*Setaria leucopila*), cane bluestem (*Bothrichloa barbinodis*), and sideoats grama are among the grasses. The range of this community has not increased since the mid-19th century (Peterson and Boyd 1988). However, the composition appears to have less of the component of taller grasses and more sand sagebrush and mesquite than before the livestock era. The oak component grows slowly and is long lived. It is estimated to have made up only 5 to 25 percent of the prelivestock era community. While in some areas the oak component has not expanded onto former grasslands (Peterson and Boyd 1988), in other areas at least the individual oak motts now occupy what was once a mixed grass prairie community. Both midgrass and tallgrass components and oak are well adapted to most fire regime variations (Peterson and Boyd 1988) while mesquite is not. Simpson (1977) contended the range of mesquite had not spread, just the densities within the range. Mesquite densities, within the strip identified by Risser and others (1981) as mixed-grass, continue to increase, but others found mesquite present in the community in 1845 (Abert and Carroll 1999).

Even though more acreage remains for the mixed grass than the tallgrass prairie, the patchiness of the grasslands means the ecosystem operates on a much smaller scale than in the prelivestock era. Fire is implemented as a management activity by individual landowners rather than occurring at landscape scales (Weaver and others 1996). Herbivory by livestock has different impacts on prairie forbs and grasses than by buffalo. Even where a high diversity of forbs remains, fragmentation favors a continued reduction in associated animal species. Homogenization of large or small pastures by livestock reduces the inherent ability of the ecosystem to withstand drought. Intense development of stock tanks across native prairie pastures interrupts the functioning of watersheds, limiting free flow of water with a coincident loss of biological diversity. Interruption of the natural fire regime likely had as much to do with expansion of oak and sand sagebrush in Oklahoma and mesquite in Texas and New Mexico as did overgrazing by livestock.

Establishing reference conditions for the mixed-grass prairie is even more difficult than for the other grasslands because of the diversity needed to provide the ecosystem's response to climatic variability. Fragmentation limits the value of potential reference sites, excluding smaller areas, even those in excellent condition. Allowances must be made for both fire, and the suite of short-, mid-, and tallgrasses necessary to withstand drought events. The buffalo, hugely reduced in number, is a missing keystone link across most of the mixed-grass prairie.

Colorado Plateau Grasslands

Castetter (1956) lumped the shortgrass prairie, or steppe, and the Great Basin grassland into a mixed prairie "association" because the grasses were also those of the true steppe and occurred as a "climax composition" in more or less equal amounts of mid- and shortgrasses. Throughout the association, blue grama was felt to be the climax dominant. Sideoats grama was an important component as was hairy grama. Mostly in lower spots topographically, western wheatgrass was "rather common" in association with blue grama. On sandy soils, little bluestem was a common associate often found with sand bluestem and Indian grass. In low saline areas, alkali sacaton stands were common. On the elevated plains, the cane cholla (*Opuntia* spp.) and soapweed (*Yucca elata*) were uncommon due to grass competition.

Blue grama is the common species among Southwestern grasslands, being even more abundant today in degraded conditions of some grasslands than in the prelivestock era. The resistance of blue grama to drought and heavy grazing played a major role in the sustainability of the shortgrass prairie. Buffalo benefit from this survival mechanism because the grazed lawn has a denser concentration of younger, higher quality forage. Having many small shoots also facilitates rapid response to ephemeral water availability limiting loss to drought and speeding recovery after defoliation (Coughenour 1985). Blue grama is one of the grasses best adapted to periodic water stress. Blue grama has a high root/shoot ratio because a high proportion of carbohydrates are translocated below ground instead of being used to grow shoots. This increases survivability and keeps carbohydrate resources higher than in less drought tolerant grasses (Detling 1979). Blue grama was the ideal grass on the flat expanses of shortgrass steppe where winds limited snow deposition and increased desiccation in the spring, and where growing season precipitation was scattered in time and space, requiring frequent dormancy and regreening in a single growing season. Blue grama was the ideal grass in higher use areas where herbivores large and small would have eaten and trampled a less-adaptive species to oblivion.

Fire in the Colorado Plateau grasslands was of low intensity but adequate to keep woody shrubs from expanding. At the edge of sandier soils, wildfire would have restricted sand sagebrush. Fires could reduce pricklypear (*Opuntia* spp.) by either killing the plant in a hotter fire or by burning the spines, making the

Arizona vistas: Contrasts of exurban development, near Flagstaff (above), with undeveloped land, farther south near Naco and the U.S./Mexican border (below). (Photos by John Yazzie)

cactus more available to a wide range of herbivores. Abert and Carroll (1999) noted the short flame lengths of the typical wildfire, saying it was not difficult to pass through the flame front unharmed.

The prickly pear was widespread and likely increased during drought periods just as it does today. Pricklypear can withstand drought better than most steppe plants, and its shallow root system utilizes moisture from even light rainfalls (Branson 1985). Pricklypear increases in overgrazed areas, and it can increase during drought even in grasslands protected from livestock.

The best sites for developing reference conditions in the Colorado Plateau grasslands are on large private ranches because they are less patchy than areas administered by the Federal government. A variety of grazing intensities would best depict the variation found in the pre-Euro-American settlement era.

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Chapter 7: Grassland Sustainability

Introduction

In this chapter we discuss grassland sustainability in the Southwest, grassland management for sustainability, national and local criteria and indicators of sustainable grassland ecosystems, and monitoring for sustainability at various scales. Ecological sustainability is defined as:

> [T]he maintenance or restoration of the composition, structure, and processes of ecosystems over time and space. This includes the diversity of plant and animal communities, and the productive capacity of ecological systems and species diversity, ecosystem diversity, disturbance processes, soil productivity, water quality and quantity, and air quality. (USDA Forest Service 2000: Glossary)

Sustainability is measured over various spatial scales, often including a nested hierarchy of smaller and larger spatial scales. For example, if the habitat of a rare species was studied at only a fine spatial scale, the impact of disturbances and larger landscape patterns that affect species distribution and viability would not be considered.

Ecosystem integrity has been variously defined and incorporates the concepts of ecosystem functioning and resilience. Grumbine (1994) described five goals of ecosystem integrity:

- Maintaining viable populations (biodiversity)
- Ecosystem representation
- Maintaining ecological processes
- Protecting evolutionary potential
- Accommodating human use

Definitions and measures of integrity are discussed by De Leo and Levin (1997).

Nearly 75 percent of all threatened ecosystems in the United States are either grasslands or shrublands (Mitchell and Joyce 2000). Grasslands are home to more than 7,500 plant and animal species in the United States. Many grassland species are now either threatened or endangered, and more than 700 species are candidates for listing under the Endangered Species Act.

The majority of Southwestern rangelands are grasslands. Rangelands are defined as those areas where the potential natural vegetation predominately comprises grasses, grasslike plants, forbs, and shrubs, and where herbivory is an important ecological process (Anderson and others 1976, Frank and others 1998, Mitchell and Joyce 2000). Rangelands affect the quality of life of every person in the United States. This land accounts for approximately 706 million acres, or 40 percent of the lands in the United States, including grasslands, shrublands, tundra, alpine meadows, Southwestern deserts, and wetlands across the country (Colorado State University 2001). The U.S. Federal government manages over 21 million acres of prairie grasslands, including the shortgrass prairie of eastern New Mexico (National Wildlife Federation 2001).

We discuss grassland and rangeland management in terms of maintaining ecosystem processes, ecological integrity, and ecological sustainability. Sustainable rangeland management is defined as management of rangeland ecosystems to provide a desired mix of benefits to the present generation without compromising their ability to provide benefits for future generations (Colorado State University 2001). Managing grassland resources, including rangelands, for sustainability will help ensure that USDA Forest Service meets its stewardship responsibility of passing the nation's resources on to future generations in improved condition (Kaufmann and others 1994).

Without an effective way to accurately monitor social, ecological, and economic aspects of rangeland, and therefore grassland sustainability, it is difficult to measure progress toward sustainability. In recent years Federal land management agencies have been criticized for a lack of consistent, standardized indicators for reporting the status of rangelands. In response to this need, local, national, and international criteria for grassland sustainability are being developed. Efforts include the Sustainable Rangelands Roundtable (SRR), the Local Unit Criteria and Indicator Development (LUCID) project, and the Montreal Process. The purpose of grassland monitoring can vary, but an important aspect is to determine whether management activities have affected ecosystem sustainability and integrity.

Grassland Management for Sustainability

Southwestern grasslands are managed for a variety of uses, including livestock grazing, wildlife habitat, protecting water quality, and for recreation (National Research Council 1994, USDA Forest Service Kiowa and Rita Blanca National Grasslands 2001). Grasslands in the West and Southwest have been degraded by overgrazing and intensive use of riparian areas (USDA Forest Service Research and Development 2001). Forest Service research and management priorities have addressed stream bank erosion, sedimentation and erosion rates, restoring riparian vegetation, providing improved habitat for native species, and the cumulative impact of wildlife and livestock grazing.

Management actions must be consistent with Federal and State regulations, including the Clean Water Act, the Multiple Use and Sustained Yield Act of 1960, the Resources Planning Act of 1974, the National Forest Management Act of 1976, the Federal Land Policy and Management Act of 1976, and the Soil and Water Resources Conservation Act of 1977.

An important principle of all grassland management is to plan for drought conditions. Hydrological extremes are associated with climatological events, especially the El Niño Southern Oscillation (ENSO). The Southwestern United States has a strong teleconnection to the ENSO cycle. In general, El Niño brings periods of heavier winter precipitation in areas that are affected, while La Niña events are associated with drought. Depending on location, December through March precipitation in New Mexico is 138 to 214 percent that of average conditions in response to El Niño events (see National Weather Service map, NOAA 2001). February through April temperatures are lower in response to an El Niño event.

For management purposes, the frequency and severity of drought are more important than long-term average climate conditions. Too often land managers plan for average climate conditions, rather than the climatic extremes that can be expected. Drought conditions and low spring runoff occur in response to La Niña events at New Mexico sites (Dahm and Molles 1992, Molles and Dahm 1990, Molles and others 1992), while in response to an El Niño, winter/spring precipitation nearly equals summer precipitation. Drought conditions, including response to La Niña events, also correspond to an increase in fire frequency that is detectable in histories reconstructed from fire-scar data.

Grazing intensity can be reduced to prevent excessive vegetation and ecosystem damage during drought. In addition, grassbanks are being used in northern New Mexico and other areas as one part of a wider solution to address drought conditions. Grassbanks are rangeland management systems that provide alternative forage for livestock displaced from their regularly permitted allotments, which are undergoing restoration (USDA Forest Service Manual, Interim Directive 2001). Grassbanks can be useful during drought conditions to alleviate the stress on rangelands that do not have the capacity for higher grazing intensity due to climatic conditions. See chapter 8, "Tools for Grassland Management," for further discussion of grazing management.

Adaptive Ecosystem Management

Forests in the Southwestern Region utilize adaptive management, including management of grassland ecosystems (Kaufmann and others 1994). Adaptive management acknowledges the complexity of ecosystems and the uncertainty involved in predicting ecosystem responses to management actions. It ensures that land managers incrementally assess whether goals are met and whether interim results are acceptable. This means that decisions affecting ecosystem sustainability are incrementally reevaluated as monitoring data are collected, summarized, and evaluated. Therefore, adaptive management establishes a pathway to alter the course of management action as new knowledge is acquired.

A grassbank has been established in Chihuahuan Desert grassland on the Diamond A Ranch (Eastern Division) in southwestern New Mexico. (Photo by Charles Curtin)

For example, improvements in grazing management can result in reduced erosion and soil compaction, promote increased infiltration of precipitation into the soil (Roberson 1996), and increase biodiversity. When monitoring data show that resource conditions are below thresholds for ecological factors, then grazing management can be altered immediately or prior to the next grazing season. Steps can be taken to prevent overutilization of forage, provide rest to disturbed areas, alter the season of use, and improve overall management of livestock. Restoration programs to increase soil and streambank stability or to revegetate denuded areas are also part of adaptive management. To be effective, plans for adaptive management could include specific actions in response to observed or measured conditions, a timeline, and specific thresholds that trigger the management actions (Roberson 1996).

Criteria and Indicators for Sustainable Grassland Ecosystems

Developing indicators and monitoring them over time can help to determine whether problems are emerging, whether any action is desirable or necessary, what action might yield the best results, and how successful past actions have been. To develop and implement sound environmental policies, data are needed that capture the essence of the dynamics of environmental systems and changes in their functioning. These kinds of data then need to be incorporated into indicators (National Research Council 2000).

A criterion is a category of conditions or processes that is an explicit goal of sustainable development or by which sustainable development can be assessed. A criterion is too general in scope to monitor directly but can be characterized by a set of indicators that can be monitored over time. An indicator is a variable that can be assessed in relation to a criterion. It should describe attributes of the criterion in as objective, verifiable, and unambiguous manner as practicable, and it should be capable of being estimated periodically in order to detect trends (Colorado State University 2001). Indicators are designed to inform us quickly and easily about something of interest. They communicate information about conditions and, over time, about changes and trends. Like economic indicators, environmental indicators are needed because it is not possible to measure everything (National Research Council 2000). For example, a criterion on the conservation of biological diversity includes indicators related to ecosystem diversity, species diversity, and genetic diversity (Colorado State University 2001).

National Indicators of Sustainability

One effort for identifying criteria and indicators (C&I) for the sustainable management of temperate and boreal forests at a national scale, the Montreal Process, has become widely recognized. Moreover, the concept of using C&I as factors for evaluating all facets of sustainability, including resource supplies, is receiving increasing acceptance (Corson 1996, Mitchell 2000). In 1995, the United States agreed to use the Montreal Process Criteria and Indicators to measure national progress in achieving the goals of sustainable forest management. This in turn generated the need for sustainability C&I for grass and shrubland ecosystems, as well as for energy and minerals (USDA Forest Service Inventory and Monitoring Institute 2001). There is a need for consistent, national baseline information to provide a common language and standards for assessment and planning that will lead to proper and effective decisionmaking. A comprehensive set of criteria and indicators should provide this tool (Colorado State University 2001).

The Montreal Process lists seven criteria and 67 indicators for the conservation and sustainable management of temperate and boreal forests. The C&I encompass a set of interrelated ecological, economic, social, and institutional factors. The seven criteria are:

- Conservation of biological diversity.
- Maintenance of productive capacity of forest ecosystems.
- Maintenance of forest ecosystem health and vitality.
- Conservation and maintenance of soil and water resources.
- Maintenance of forest contribution to global carbon cycles.
- Maintenance and enhancement of long-term multiple socio-economic benefits to meet the needs of societies.
- Legal, institutional, and economic framework for forest conservation and sustainable management.

The first five of the seven criteria, along with 28 indicators, have been used to assess the applicability of Montreal Process biological and abiotic indicators to rangeland sustainability at a national scale (Mitchell and Joyce 2000). See *The International Journal of Sustainable Development and World Ecology*, volume 7 (2), June 2000, for a detailed analysis of the challenge and promise of developing C&I for rangelands, and for implementation at the national scale. So far 16 indicators have tentatively been identified for rangelands, including indicators for landscape diversity, community diversity, and population diversity. Genetic diversity has been particularly difficult to define due to a lack of baseline data. Some of the indicators lack standardized protocols. More work needs to be done on refining definitions, designing monitoring systems, and testing critical assumptions.

Other efforts to develop national sustainability indicators for rangelands include the formation of the Sustainable Rangelands Roundtable, which is sponsored by Colorado State University, the USDA Forest Service, USDI Bureau of Land Management, and USDA Agricultural Research Service. The roundtable includes representatives from nongovernmental organizations, public and private land management professionals, rangeland scientists, and university professionals (Colorado State University 2001). The roundtable will identify indicators of sustainability based on social, economic, and ecological factors, in the effort to provide a framework for national assessments of rangelands and rangeland use. For more information, access http://sustainablerangelands.cnr.colostate.edu/Roundtable_description.htm.

Local Indicators of Sustainability

Criteria and indicators for sustainability are being developed for specific scales. The intended scale for the Montreal Process was for all forests of a country, regardless of land ownership. The Local Unit Criteria and Indicator Development Project (LUCID) was initiated by the Inventory and Monitoring Institute to test criteria and indicators of sustainability at local levels. LUCID targets a local scale, such as the Tongass National Forest in Alaska or the Blue Mountain Province National Forests in Oregon. The local criteria and indicators should be revised periodically to incorporate new research results, technological advances, and new methods of measurement.

LUCID sites include forest, rangeland, and shrubland ecosystems. This process was developed to identify conditions that are needed to sustain ecological, economic, and social systems, and to determine the criteria and indicators for assessing how resource management influences sustainability. The Blue Mountains LUCID pilot test addressed C&I with verifiers (measurement protocols) and standards for shrublands and grasslands. The three Blue Mountains forests located in Oregon and Washington use the same ecological definitions, and the C&I are intended to apply to the entire province. Although it is too soon to draw conclusions from C&I testing at Blue Mountains, future reports will address water quality, forest health, and community growth.

Here is an example of a local grassland criteria and indicator that is based on the LUCID criteria and indicators for maintenance of ecosystem integrity. The degree of fragmentation is an indicator of landscape structure (USDA Forest Service Inventory and Monitoring Institute 2001). Accelerated habitat fragmentation can be a detriment to species viability and adversely affect ecological patterns and processes such as disturbance (Forman 1995). One measure of fragmentation is the ratio of edge to interior habitat, with higher ratios indicating increasing fragmentation. Corresponding landscape metrics based on the LUCID criteria and indicators are shown in figure 7-1.

Maintenance of Ecosystem Integrity

Criteria:	landscape structure
Indicator:	fragmentation and connectedness
Verifier:	ratio of edge to interior habitat area

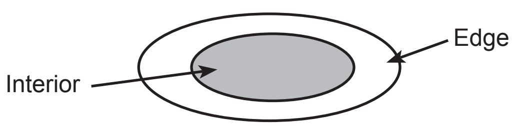

Figure 7-1. Criteria, indicator, and verifier for fragmentation: a sample metric for monitoring sustainability at landscape scale that is based on the LUCID criteria and indicators for maintenance of ecosystem integrity (USDA Forest Service Inventory and Monitoring Institute 2001). An edge is the boundary between habitats that protects the core. Higher ratios of edge to interior habitat area indicate increasing landscape fragmentation (Forman 1995, Prendergast 2000). Note that verifiers should have specific target values.

Monitoring for Sustainability in Grasslands

Monitoring is a step-wise process that involves: framing a question(s) and developing a study plan to address the question(s) using a standard protocol; collecting data according to the monitoring plan; storing the data for retrieval, and evaluating the results. Monitoring should include goals, thresholds for change, and remedial actions that occur when thresholds are met or exceeded. An ecological systems approach to monitoring ensures a strong foundation in ecological theory, adequate consideration and understanding of cause-and-effect relationships, and a systematic approach to select and evaluate parameters that are monitored (Sieg 1999). Furthermore, the questions should focus on key ecological processes and interactions, rather than individual parts of the system.

Conversely, *inventory* involves gathering data that are needed to analyze and evaluate the status or condition of resources (Powell 2000). Examples include the Forest Service programs entitled Terrestrial Ecological Unit Inventory (TEUI), Biological Forest Resource Inventory, Rangeland Resource Inventory, Wildlife Habitat Inventory, Threatened and Endangered Species Habitat Inventory, and the Human Dimension Heritage Inventory. Corporate strategies, goals and approaches for integrated natural resource inventories were developed for national (strategic), forest (tactical), and project (operational) scales (USDA Forest Service 1999).

The Forest Service recognizes three types of monitoring related to Land Management Planning: implementation, effectiveness, and validation monitoring. *Implementation monitoring* tracks compliance with Forest Plan standards and guidelines and helps determine whether planned management activities were completed. *Effectiveness monitoring* helps determine whether desired outcomes were achieved by management actions. *Validation monitoring* tests the assumptions and models of Forest Plan implementation. In addition, monitoring associated with applied research can supply critical information for decisionmaking. The scope of *ecological monitoring* needs to include programmatic monitoring that tracks and evaluates trends of ecological, social, or economic outcomes (Powell 2000).

Despite the development of protocols and indicators for sustainability, the criteria of sustainable grasslands are not easily measured. Some helpful references for grassland monitoring include Elzinga and others (1998), Gibbs and others (1998), and Munn (1988). Examples of existing grassland monitoring programs can be found in Wondzell and Ludwig (1995). A successful monitoring program for grasslands could include the following considerations.

- A description of each step of the monitoring system and the quality assurance/quality control protocol (Bormann and others1994, Everett and others 1994, Moir and Block 2001).
- The statistical parameters to be measured.
- Trend detection.
- A funding commitment from management, including maintenance of field equipment or repeated remote-sensing applications.
- Early determination of who monitors and a commitment of adequate human resources to accomplish the tasks over time.
- Public stakeholders' involvement in the monitoring program (Cortner and Moore 1999).

Monitoring for grassland sustainability should address current practices for managing grazing animals and grazing herds, especially conservative stocking rates. A summary of grazing systems is beyond the scope of this chapter. However, an analysis of the importance of grazing intensity versus grazing system is provided by Holechek and others (1999).

Forest Monitoring Reports

According to Land and Resource Management Plan monitoring reports for U.S. Federal government fiscal years 1998 and 1999, most National Forest monitoring in the Southwestern Region was implementation monitoring—that is, 65 percent. (Data are only available through the USDA Forest Service Intranet site.) About 32 percent of the reported monitoring was evaluation monitoring, and the remaining 3 percent was validation monitoring. The information is not detailed enough to know whether grassland monitoring followed this general trend.

Study Design

Successful monitoring programs should have the following characteristics (Powell 2000, 2001):

- Be purposeful and conducted to answer specific questions.
- Be done at the appropriate spatial and temporal scale to answer the question.
- Be done in collaboration with others (for example, agencies, interested publics, researchers, and nongovernmental organizations). Collaboration results in sharing the workload (including obtaining data from other sources), gaining expertise, and building credibility and trust.
- Use the best available science and established protocols to collect and evaluate the data.
- Use modern information management techniques and tools.
- Apply stringent selection criteria so that a monitoring activity is only conducted if it is feasible, realistic, and affordable.
- Emphasize evaluation as much as the collection of the data.

The established protocol identified in a monitoring plan should include standard sampling and analytical methods that determine the precision and accuracy of measurements. These are the procedures for quality assurance and quality control. Accuracy refers to the degree of agreement between a calculated or measured quantity and the true value of the parameter. Precision refers to the degree of agreement between replicate measurements of the same parameter, and includes the concepts of duplicability, repeatability, and reproducibility. For example, in the USDA Forest Service Global Change Research Program, quality management establishes programwide policies and procedures that ensure adequate documentation and data quality for all field, analytical, and modeling activities. Quality assurance implements these policies by establishing and monitoring quality control (QC) procedures including the identification of variability and followup control recommendations to improve the accuracy and precision of measurements. QC procedures are implemented by scientists within each project and are designed to produce a sustained reduction of error and document systematic error within statistically defined limits (USDA Forest Service Global Change Research Program 2001).

Proper training and supervision of field and laboratory staff is necessary to ensure adherence to the protocol and the success of the monitoring program.

Scale

The temporal and spatial scales chosen depend on the question or problem that monitoring will address. Spatial scales for assessing sustainability can be viewed, for example, as ecological units (Bailey 1995); a nested watershed hierarchy; or other appropriate units such as community, ecosystem, and landscape scales. Temporal scales can range from the time it takes bacteria to reproduce or molecular processes to occur through evolutionary and geological time scales.

It is necessary to conduct ecological studies at the appropriate temporal and spatial scales because changes occur at many scales at the same time, and different processes are likely to be important at different scales. Depending on the scale chosen, two species can appear to be highly interrelated or completely independent. For example, Levin (1992), when considering the reasons for shrimp distribution in the ocean, found that krill distribution at small spatial scale was a function of the behavior of individuals. However, at large scale, krill distribution was a function of oceanographic processes. Thus, the conclusions about the factors that affect the distribution of these organisms varied according to the scale where measurements were made.

Processes and functions that are appropriate to monitor at watershed scale include succession, biogeochemical cycles, energy transfer through the food web, disturbance, and competition.

Because studies at fine spatial resolution have greater detail, the study results can detect heterogeneity and other ecological patterns. However, those same study results also have a low potential for

Table 7-1. Criteria and indicator linkage between Montreal Process and Local Unit Criteria and Indicator Development (LUCID). Reproduced from LUCID homepage, http://www.fs.fed.us/institute/lucid/.

MONTREAL PROCESS CRITERIA (1-7)
LUCID Indicators

1. CONSERVATION OF BIOLOGICAL DIVERSITY
I.2.2.1 Vegetation types and structural classes
I.2.4.5 Ecologically sensitive areas, e.g., riparian areas are retained
I.2.5.1 Populations of indigenous species
I.2.6.1 Exotic species
I.2.6.2 Community guild structure
I.2.6.3 Species at risk
I.2.7.1 Gene frequencies change

2. MAINTENANCE OF PRODUCTIVE CAPACITY OF FOREST ECOSYSTEMS
I.2.2.3 Fragmentation and connectedness
I.2.2.2 Linear features
I.2.3.2 Primary productivity
I.3.1.2 Land base available for production

3. MAINTENANCE OF FOREST ECOSYSTEM HEALTH AND VITALITY
I.2.1.2 Disturbance processes

4.CONSERVATION AND MAINTENANCE OF SOIL AND WATER RESOURCES
I.2.1.1 Hydrologic condition
I.2.4.1 Pollutants
I.2.4.2 Soil quality e.g., soil compaction, displacement, erosion, puddling, loss of organic material
I.2.4.3 Soil nutrients
I.2.2.5 Water quality e.g., dissolved oxygen, suspended sediments and water nutrients
I.2.4.6 Morphology and function of stream channels

5. MAINTENANCE OF FOREST CONTRIBUTION TO GLOBAL CARBON CYCLES
I.2.3.1 Nutrient cycling
I.2.4.4 Ecological legacies and structural elements

6. MAINTENANCE AND ENHANCEMENT OF LONG-TERM MULTIPLE SOCIO-ECONOMIC BENEFITS TO MEET THE NEEDS OF SOCIETIES
1.1.1 Wilderness
1.1.2 Aboriginal and non-Aboriginal cultural, spiritual, social sites/values
1.2.1 Scenery
1.3.1 Recreational, tourism and education opportunities (by activity)
1.4.1 Access to forest resources
1.6.1 Worker health and safety
1.6.2 Public health and safety
1.7.1 Subsistence and non-subsistence gathering
I.3.1.1 Community economic trade balance (imports and exports)
I.3.2.1 Annual and periodic removals of products (timber and non-timber)
I.3.2.3 Money spent by visitors in local communities (by activity)
I.3.2.4 Value to products including value-added through downstream processing
I.3.2.5 Resource production component of economy
I.3.2.6 Income from National Forest activities
I.3.2.7 Employment of local population in resource management
I.3.3.1 Rent capture
I.3.3.3 Community economic diversity

7. LEGAL, INSTITUTIONAL AND ECONOMIC FRAMEWORK FOR FOREST CONSERVATION AND SUSTAINABLE MANAGEMENT
1.4.2 Ownership and use rights
1.5.1 Participation/involvement in decision-making
I.3.3.2 Mechanisms for economic benefits sharing

making generalizations. Studies at broad scale show the dynamics of a system, detect slower rates of process or system change, and have a high potential to derive generalizations. Studies at multiple or nested scales can be used to aggregate and extrapolate fine-scale results to larger scales. Examples of how to identify questions at the appropriate scale are provided by Powell (2001).

Criteria and Indicators for Monitoring

For each question that is raised, criteria and indicators to answer that question should be identified and monitored. Because an essential goal of ecosystem management is ecosystem sustainability (Funston 1995, Rauscher 1999), questions associated with ecological monitoring might address whether ecosystem conditions, or management activities affecting ecosystem conditions, are sustainable. The international Montreal Process was developed to identify criteria and indicators for forest sustainability at a national scale. Subsequently, the applicability of the criteria and indicators from the Montreal Process to grasslands has been assessed (Flather and Sieg 2000, Joyce 2000, Joyce and others 2000, McArthur and others 2000, Neary and others 2000). The Local Unit Criteria and Indicator Development (LUCID) Project addressed sustainability at a forest management scale. Monitoring for these criteria and indicators will help determine whether ecological conditions are sustainable (table 7-1).

Tools

There is no standard "tool kit" for ecological monitoring in grasslands. The scientific methods and processes used to answer ecological questions are the monitoring tools. The methods and processes chosen will depend on the question(s), the ecological indicators that are selected, temporal and spatial scales of concern, degree of detail, precision and accuracy that is required, and financial budget. Therefore, the concept of monitoring tools is a broad one that can include:

- Methods for framing your questions and multiple working hypotheses.
- Instrumentation for quantitative or qualitative measurements.
- Analytical tools to detect patterns and change across time and space (such as GIS, remote sensing, fractal analysis).
- Other methods for data analysis such as statistics, conceptual and mathematical models, fuzzy logic.
- Decision-support systems or other expert systems.

Vegetation sampling methods for rangelands are discussed by Stohlgren and others (1998). Other grassland monitoring methods are discussed in Jacobsen and others (1998).

Evaluating and Interpreting Results

Much of the Forest Service monitoring data are being stored in a corporate database, the Natural Resource Information System. These data require scientific evaluation considering current scientific knowledge and research results. One step in evaluating monitoring data includes a comparison to critical values for "keystone" indicators of change. When critical values are approached or exceeded, then further actions are likely warranted to ensure sustainability. For example, the Air Resource Management Program has identified wilderness values that can be affected by air pollution, and their associated sensitive receptors and concern thresholds. These indicators and thresholds reflect pollution transport and interactions between the atmosphere, geosphere, and biosphere, and these are contained in the Natural Resource Information System, NRIS-Air module. They provide management guidelines for protecting Air Quality Related Values in Class I Wilderness Areas.

Evaluation also includes summarizing data in statistical and graphical formats and identifying changes or trends. A variety of methods exist for detecting ecosystem changes and trends including statistical methods (analysis of variance, cluster analysis, and so forth), comparisons of current conditions with past reconstructions (dendrochronology, paleolimnology, pack-rat middens, and so forth), modeling, and spatial analysis using GIS, satellite imagery, and remote sensing. A common approach for trend detection is analyzing data from repeat sampling over time, including data from long-term monitoring networks. In some cases, a qualitative assessment and professional judgment may be adequate evaluation tools.

Condition Classes

One way to assess and categorize monitoring data is according to condition classes. A good example is the Soil Condition Rating Guide to determine whether soil quality objectives are met (table 7-2). The guide incorporates multiple indicators for three soil ecosystem functions—hydrologic function, stability, and nutrient cycling—into ratings of either satisfactory, impaired, or unsatisfactory.

Other monitoring systems that use multiple indicators and classification include Thalweg-Watershed

Table 7-2. Soil Condition Rating Criteria. Reproduced from FSH 2509.19, Soil Management Handbook, R3 supplement 2509.18-99-1, effective 10/20/1999. Also see Doran and others (1994).

SOIL CONDITION RATING GUIDE

Function	Indicator	CONDITION CATEGORY		
		Satisfactory	**Impaired**	**Unsatisfactory**
HYDROLOGIC	Surface Structure [1]	Moderate/strong granular or single grained.	Sub-angular blocky or weak granular.	Massive or platy.
	Surface Pore Space [1]	Many/common tubular pores, high vertical continuity.	Common/few tubular pores.	Few tubular pores, low vertical continuity.
	Rupture Resistance [1]	Loose to slightly hard (dry) Loose to friable (moist).	----	Very hard to very rigid (dry), Extr. firm to very rigid (moist).
	Near Surface Subzones [1]	No surface crusting or subsurface compaction.	Water compacted or non-biotic surface crust present.	Mechanically compacted.
	Bulk Density	Bulk density not increased.	Moderate bulk density increases (5-15%). (>15%).	Significant increase in bulk density
	Infiltration	No decrease in infiltration.	Moderate decrease in infiltration. (10-50%).(>50%).	Significant decrease in infiltration
	Penetration Resistance	No increase in resistance.	Moderate increase in resistance (10-50%).(>50%).	Significant increase in resistance
STABILITY	Modeled Soil Loss	Current soil loss ≤ tolerance.	Current soil loss > tolerance.	
	Visible Sheet Rill & Gully Erosion	Sheets/rills/gullies not evident.	Rills/gullies are small, discontinuous, poorly defined & not connected into any pattern.	Rills/gullies actively expanding, well-defined, continuous & connected into a definite pattern.
	Pedestaling	No/slight pedestaling of plant, litter and rocks. No evidence of exposed roots.	Grasses, forbs and rock fragments are pedestaled. Small, fibrous root strands of forbs & grasses are exposed on the soil surface.	Trees and shrubs are pedestaled and may be hummocked. Shallow, lateral roots of trees and shrubs are exposed.
	Erosion Pavement [2]	None to slight. If erosion pavement exists it is discontinuous or localized.	Erosion pavement is continuous or exists in interspaces between canopy cover of trees & shrubs.	
	Soil Deposition	Not unusual or excessive.	Soil and/or litter deposition is present. Fine litter may be patterned as small debris accumulations.	Soil and/or litter is deposited on the uphill side of logs, brushpiles, etc. Soil may be moving offsite.
	Surface ("A") Horizon	"A" horizon is present, well distributed, not fragmented.	"A" horizon is present, but not evenly distributed. Changes in physical properties exist.	"A" horizon is absent or present in association with prominent plants. Properties are similar to those of the underlying subsoil.
NUTRIENT CYCLING	Vegetative Community Composition	Distribution of desirable, perennial plants reflects species by vegetative layer (i.e. trees, shrubs, forbs and graminoids) as identified in the potential plant community.	Changes in vegetation composition indicate a shift towards a drier, less productive plant community. There may also be an increase in annual plants, shallow-rooted grasses, taprooted woody perennials or invasive plants.	The perennial forb and/or graminoid vegetative layers are absent or sparse.
	Litter	Litter is distributed evenly across the soil surface and is associated with all vegetative layers	-----	Litter is either absent or is associated only with prominent plants and not evenly distributed across the soil surface.
	Coarse Woody Material	Pipos/Quga-----5-10 t/ac. Pipos/Fear2-----7-14 t/ac. Abco/Fear2-----8-16 t/ac.	-----	Pipos/Quga-----<5 t/ac. Pipos/Fear2-----<7 t/ac. Abco/Fear2-----<8 t/ac.
	Root Distribution[1]	Many/common roots in surface horizons.	Moderately few roots in surface horizons.	Few/very few roots in surface horizons.

1/ Categories and/or descriptions defined in USDA Handbook No. 18, Soil Survey Manual, October, 1993.
2/ Certain soils within desert ecosystems inherently contain erosion pavement (desert pavement) surfaces. Desert pavements are not used to indicate soil condition.

Link (T-Walk) and Proper Functioning Condition (PFC) for riparian areas. T-Walk classifies stream conditions as either robust, adequate, diminished, impaired, precarious, or catastrophic (Ohlander 1998). PFC protocol uses the categories Proper Functioning Condition, Functional - At Risk, and Nonfunctional to describe riparian conditions (USDI Bureau of Land Management 1993). Further examples of rangeland classification are provided by the National Academy of Sciences (National Research Council 1994).

BLM has also recently developed procedures for rangeland health assessment (Pellant and others 2000). These are based on three attributes of rangeland health: soil/site stability, hydrologic function, and integrity of the biotic community. The procedures use 17 indicators that focus on vegetation or soil stability to assess the function of the three attributes of rangeland health. The indicators are:

- Rills
- Water flow patterns
- Pedestals and/or terracettes
- Bare ground
- Gullies
- Wind-scoured, blowouts, and/or deposition areas
- Litter movement
- Surface soil resistance to erosion
- Soil surface loss or degradation
- Plant community composition and distribution relative to infiltration and runoff
- Compaction layer
- Functional/structural groups
- Plant mortality/decadence
- Litter amount
- Annual production
- Invasive plants
- Reproductive capability of perennial plants

There are also additional optional indicators.

Attributes are rated qualitatively according to condition classes that compare current conditions to departure from reference areas or ecological site description. Condition classes include extreme departure, moderate to extreme, slight to moderate, and none to slight.

Modeling

State-of-the-art modeling tools can be used to examine changes in grassland processes. For example, the Century Model simulates the dynamics of carbon, nitrogen, phosphorus, and sulfur over time. It can be used as a tool for ecosystem analysis, to evaluate the effects of management changes, and to analyze changes in the biogeochemistry of grasslands (Parton and others 1996). The model is based on net primary productivity as a function of water availability, nitrogen availability, and temperature; and soil carbon and nitrogen pools that are active, slow, or passive in their turnover times. Century includes a grassland/crop submodel that is linked to a soil organic submodel simulating the flow of nutrients through plant litter and soil. The model has recently been used to assess the response of temperate and tropical grasslands to climate change (Pace and Groffman 1998).

HilleRisLambers and others (2001) used mathematical modeling to interpret vegetation patterns (patches of vegetation and bare soil) in semiarid grazing systems. They noted that there is a prevalent, positive relationship between plant density and water infiltration that results in formation of vegetation patterns in semiarid areas throughout the world. Modeling results showed that other factors such as herbivory (grazing) are not as important in generating the patterns. Where plant dispersal is low, increased herbivory is predicted to lead to a transition from closed vegetation cover to spatial patterns in vegetation to bare soil. This same transition is not likely to occur, however, where plant dispersal is high. The model also predicts that vegetation changes are reversible if grazing is reduced.

Summary

Grasslands in the Southwest have a semiarid to arid climate and a patchy distribution of precipitation. However, land managers often plan for average climate conditions and distribution, rather than the extremes that can be expected. For management purposes, the frequency, severity and distribution of drought are more important than long-term average climate conditions.

The USDA Forest Service continues to address various management concerns in grasslands including streambank erosion, sedimentation and erosion rates, restoring riparian vegetation, providing improved habitat for native species, and the cumulative impact of wildlife and livestock grazing. It is important to determine whether management activities impact ecosystem integrity and sustainability. Ecological sustainability can be defined as "the maintenance or restoration of the composition, structure, and processes of ecosystems over time and space." One way to measure sustainability is using criteria and indicators of sustainability at local levels; an example is the Forest Service's LUCID program, which was designed to test appropriate monitoring parameters at the local level. While there is no standard "tool kit" for assessing ecological sustainability, a variety of useful tools are available, including the Forest Service's Soil Condition Rating Guide and modeling tools such as the Century model.

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Chapter 8:

Tools for Management for Grassland Ecosystem Sustainability: Thinking "Outside the Box"

Introduction

Grassland ecosystem management is dynamic and has adapted to the development of new tools and ideas. Our ancestors were indirectly managing grasslands when they learned to move livestock to take advantage of better water and greener forage. One could argue that even their hunting of grassland wildlife, especially the use of fire to drive animals to waiting hunters, had an influence on local grassland ecology.

The science of range management is relatively new and is linked to the realization that land is limited and that managers must do their best to make the available resources last forever (Stoddart and Smith 1955). Poor land stewardship degrades the land's multiple resources and results in poor livestock production, while good management will provide healthy grassland ecosystems so that managers and landowners will have high livestock production and the potential for a viable economic livelihood. Owners, especially private owners, probably have different goals for their lands; traditional livestock production is not always the primary or even a secondary objective, but proper land stewardship should be the ultimate goal regardless of other objectives.

New challenges are impacting the management and health of all natural ecosystems, including Southwestern grasslands, and these forces are affecting land stewardship efforts. One of the greatest challenges is the result of the phenomenal growth of the region's human population, primarily urban and suburban—people with few direct connections to the land. Large expanses of privately owned grassland, such as some near Sonoita, AZ, have been subdivided into small units, fragmenting a relatively homogeneous landscape with potentially adverse impacts on many wildlife species, on regional hydrology, and on the traditional social fabric of the area. The new residents often introduce nonnative plants through their landscaping activities, miles of fences and new roads, and concentrations of predatory pets and domestic livestock. Even without fragmentation, the pursuit of recreational opportunities by this growing population is putting pressure on all open landscapes, impacting the vegetation, wildlife populations, and watershed condition. Other challenges are the growing concerns about the loss of native plant and animal species because of habitat loss, human disturbances, and the introduction of numerous nonnative species that successfully occupy areas of bare soil or out-compete native plants. Many grasslands have been adversely affected by past land management and need to be restored to a more functional ecological condition. But how can this be accomplished ecologically and economically?

Ecological and range management sciences continue to provide managers with new information and tools. Descriptions of many of the common techniques are found in standard range management textbooks dating from the historical coverage by Stoddart and Smith (1955), and from books such as Savory (1988), Holechek and others (1998), and Jemison and Raish (2000) that describe more recent developments. Important information has been gathered and disseminated by Federal, university, and State research, management, and extension agencies.

This chapter covers some new and current tools, issues, and approaches to grassland management. It does not attempt to be all inclusive but does reflect the opinions and expertise of the contributors. The first section includes some programs and tools and principles related to grassland management and restoration, and the second section covers new remote sensing technologies to evaluate grasslands and assist in management planning at a landscape level. The adaptation and use of remote sensing technologies and geographical information system databases to grassland management are just developing. The third section deals with education, a vital aspect of range management if current and future grassland managers are to do their best and if the general public is to be a true shareholder in these efforts.

The chapter has benefited from the written contributions of several USDA Forest Service employees from the Southwestern Regional Office in Albuquerque, NM, or from the Rocky Mountain Research Station. The authors and their sections are indicated throughout the text. Their contributions are greatly appreciated. Special thanks are due to Cathy Dahms, formerly of the Southwestern Region, who recruited the contributors and assisted with the editorial process.

Grassland Resources Management

New Programs to Sustain Southwestern Grasslands

Attitudes and philosophies toward grassland management have shifted over time since the first herds of livestock were introduced into the Southwest. However, many changes have occurred more recently in response to the growing pressure on grasslands and their resources from traditional producers, conservation and environmental organizations, government entities, and especially from the growing regional population. Ranchers, conservation groups, and many members of the general public have a common interest in maintaining open landscapes because of the ecological importance of these systems, recreational and aesthetic concerns, and the desire to preserve the rural livelihood and life style (Raish, this volume). These groups, often working together, have developed innovative approaches to achieve their goals.

Grassbanks—Grassbanks are becoming popular methods for allowing land and grass resources to rest and recover during dry periods while continuing to provide the rancher with an economic return. The Malpai Borderlands Group (MBG), located in southeastern Arizona and southwestern New Mexico, was the first organization to initiate the Grassbank system to restore native grasslands and protect the open spaces of the region. "Grassbank" is a legally registered trademark. Basically, a Grassbank is grass made available from one ranch to another rancher's cattle in return for conservation value equal to the value of the grass (McDonald 1995). The MBG is a nonprofit organization, composed mainly of ranchers, that works to encourage profitable ranching and traditional livelihoods that sustain the region's open spaces (McDonald 1995). A specific goal is: "…to restore and maintain the natural processes, including fire, that create and protect a healthy, unfragmented landscape to support a diverse, flourishing community of human, plant and animal life in our Borderlands Region" (McDonald 1995: 483). Drum Hadley, of the Animas Foundation is credited with the idea of a Grassbank as a method of creating more grass, grass seed, and protection of open spaces. The Animas Foundation's Gray Ranch (also known as the Diamond A Ranch) provided the first Grassbank.

Grassbankers receive access for their cattle to the Grassbank in return for conveying a conservation easement to the MBG. The MBG pays the owner of the Grassbank for the use of the grass. The value of the easement is determined by an appraisal of the development rights on the Grassbanker's ranch. The Grassbanker gets access to the grass for a term that monetarily equals the value of the easement. A conservation easement prevents the sale of land for subdivisions and, thus, prevents landscape fragmentation. Easements can be released back to the landowner under two conditions. The first condition is the dissolution of the MBG, when no acceptable alternatives are available. This condition is grandfathered into original Grassbank agreements but is no longer part of easements now being offered. The second condition is the loss of access to grazing on nonprivate lands that are part of the landowner's ranch when this occurs through no fault of the rancher. Resting grasslands is a recognized method of improving range conditions and ultimately to improve the economics of ranching. The MBG and some government agencies, notably, the USDA Natural Resources Conservation Service, also may cooperate with Grassbankers to conduct improvement projects and monitoring to help the land return to a more productive condition before cattle are returned to the rested ranges. Fencing projects are one example of range improvements. Ranchers on the rested ranges agree to maintain their water systems for wildlife use during periods of rest. Stocking rates have to be maintained at lower levels than normal within the Grassbank so that the land will be able to provide sufficient forage for the additional cattle. Some lands within the Grassbank may be rested or deferred in anticipation of receiving a Grassbanker's livestock. Grassbank strategies need to be flexible to allow for variable climatic conditions.

Grassbanking also is being practiced in the Valle Grande Grass Bank Demonstration Program in northern New Mexico (Valle Grande Grass Bank Briefing Information, Anonymous, undated). The area is east of Santa Fe at the east end of Glorieta Mesa. The program is a cooperative effort involving the USDA Forest Service, the grazing permittees, the Conservation Fund, and the New Mexico State University Cooperative Extension Service. The basic goals of creating sustainable ranching are similar to those of the Malpai Borderlands Group, but this association consists of Forest Service permittees rather than private landowners. The Conservation Fund, a nonprofit organization, which holds the Valle Grande allotment and owns associated private lands, also hopes to reduce conflicts between grazing and other land uses and to demonstrate positive livestock and range management. Education for permittees and the public are important aspects of the program. Grassbankers under the program join the Valle Grande Grazing Association and are assessed a fee to cover operations; however, the Conservation Fund could provide supplemental funding. While their ranges are being rested, members are expected to conduct range improvements, such as riparian fencing and brush removal operations, which the Forest Service will fund. Grassbank arrangements can extend from 3 months to 3 years.

Other ranching organizations that foster proper land stewardship, such as the Quivira Coalition of Santa Fe, NM, also indicate that grassbanks are one tool for fostering ecological, economic, and social health of Western landscapes (Quivira Coalition undated). The USDA Forest Service is using a similar system in a coordinated and cooperative approach with forest partners when opportunities arise following a range assessment. The Tonto National Forest in central Arizona has policies and procedures in place that consider allowing one rancher to use another's unused allotment if all parties are in agreement (B. McKinney 2004, personal communication).

Conservation Easements—Conservation easements are voluntary legal instruments by which development rights or the rights to conduct other activities are conveyed to a qualified conservation entity (Society of American Foresters 2001, Stein and others 2001). The Malpai Borderlands Group requires conservation easements for ranchers to participate in its Grassbank program. Ranchers, conservation organizations, and many government agencies are concerned about the subdivision of large areas of private agricultural, forest, or range lands to create parcels of land for "40-acre ranchettes" or other small home site ownerships. Subdivision can create a personal loss to the original owner because many generations of effort are lost and the land will not be passed to heirs, even though the land sale may be financially profitable. Some landowners may consider selling their ranches for subdivision to provide for retirement or inheritance for their children. Society then loses valuable, scenic open spaces and the viability of local rural economies that depend on traditional agricultural activities (Rosan and others 1997). Subdivided landscapes can threaten the viability of dynamic ecosystems by obstructing traditional migration routes, destroying wildlife habitat, introducing nonnative plant and animal species, and increasing the demands on limited water supplies. Many wildlife species are sensitive to disturbances attributed to increased human activities (Mitchell and others 2002). One study in Colorado compared subdivided and intact ranches and found that subdivision resulted in an eightfold increase in road densities and, on one ranch, in a fourfold increase in number of landscape patches (Mitchell and others 2002). Another Colorado study found native plant communities were maintained better with less bare soil on ranches than in subdivided lands or in nature reserves (Maestas and others 2002). Maestas and his associates (2002) also cite evidence that there is change in animal biological diversity as lands are subdivided, with an increase of human-adapted species and a decrease in sensitive species. The alternative of conservation easements can be an effective tool for maintaining working landscapes, preserving environmental values, and protecting communities from excessive development (Society of American Foresters 2001).

Among the several approaches to conservation easements is the purchase of development rights (PDR). This allows the landowner to conserve working landscapes using market and incentive based, nonregulatory techniques (Stein and others 2001). PDR programs began in the Eastern United States in the 1970s to protect open spaces from suburban sprawl. Currently, more than 1,200 nonprofit land trusts in the United States (Hocker 2002) have protected about 2.6 million acres from future development through conservation easements (Maestas and others 2002). A local government or private organization, such as The Nature Conservancy and various nonprofit land trusts, purchases the real estate development rights from the landowner; however, the owner still owns the land and can use it for range, forestry, or agriculture and still obtains a financial benefit from the management of these resources. However, the activities may not affect the land's conservation value (Rosan and others 1997). The owner may sell the rights to all or part of the land and still retain part for limited development, for example for future home sites for children or to meet cash needs. These lots should not interfere with the ranching operation or detract from the scenery or natural resources (Rosan and others 1997). The land remains in the family and can be passed on to the owner's heirs, and a PDR may reduce inheritance taxes because the land's value has been reduced by 40 to 75 percent (Stein and others 2001). The purchaser of an easement is responsible for monitoring to ensure that the terms of the agreement are followed. The easement usually stays with the land for perpetuity and cannot be cancelled if the property is sold. However, some easements specify shorter durations (Society of American Foresters 2001). The PDR is a particularly important tool in the West because few governments or private organizations can afford to directly purchase large tracts of land. State governments have raised money for PDR programs, for example, by using parts of sales tax levies or by State lottery profits. Funding also can come from programs administered by Federal agencies such as the USDA Forest Service and USDI Bureau of Land Management.

There often are more requests by landowners to enter into a conservation easement than there is funding available. A landowner may sell the easement at less than its full value as a "bargain sale" and use the difference as a tax-deductible charitable donation.

The landowner may donate the entire easement to a nonprofit land trust or government entity. The amount of the charitable deduction is based on the impact of the easement on the value of the property (Rosan and others 1997). The value of an easement of more than $5,000 must be verified by a qualified appraisal if a contribution is claimed against Federal taxes.

Grassland Restoration—Grassland restoration is needed to return the ecosystems to productive and healthy states by enhancing the herbaceous cover that will increase productivity and biological diversity and also contribute to reduced erosion and to the reintroduction of fire into grassland ecosystems. Box (2002) says that land has been abused by people, the economy, and the weather. Many grasslands in the Southwest are in need of restoration because of past heavy grazing by herds owned by early ranchers and corporations who were trying to make a profit or just trying to make a living, combined with periodic drought that often extended for several consecutive years. This resulted in declines in herbaceous cover and species diversity, and increases in woody species. The changes in vegetation characteristics produced modifications of the hydrologic cycle, in particular, less infiltration of precipitation and increased surface runoff, and accelerated soil erosion and sedimentation.

Southwestern grasslands have been used and often abused for more than a century in parts of New Mexico. Sheep were an important exportable item during the Spanish Colonial Period in New Mexico, and large bands were maintained and herded down the Rio Grande Valley to Mexico (Gottfried and Pieper 2000). Large herds of cattle were imported into New Mexico and Arizona Territories after the suppression of the Apache Indians in the 1880s and the construction of the transcontinental railroad (Schickendanz 1980). In the early 1880s, three large cattle companies ran more than 60,000 head of cattle in Cochise County, Arizona, and Hidalgo County, New Mexico (Hadley and others 1999). Grazing continued in the area through the droughts of 1885, 1892 to 1893, and 1902 to 1903, when up to half of the cattle died of starvation and range resources were rapidly depleted (Hadley and others 1999). Heavy grazing also occurred in northern Arizona, and probably in other areas of the Southwest, during World War I to meet the demand for meat (Schubert 1974). Grazing during these earlier periods exceeded the carrying capacity of the land and resulted in a rapid decline in range vegetation resources, and in accelerated erosion, and channel downcutting. This combination of factors initiated desertification along the United States/Mexican border region (Hadley and others 1999) and throughout the Southwest. The impacts on watershed condition during this time still are apparent and can be identified by erosion rates, gully erosion, and soil compaction (USDA Forest Service 1993).

Two primary impacts of the loss of the herbaceous cover have been accelerated erosion and the encroachment of woody species, such as mesquite (*Prosopis* spp.) or juniper species (*Juniperus* spp.). The increased density of woody species has been linked to past heavy overgrazing by livestock and the consequential removal of fuels for wildfires as a natural control mechanism in many grassland communities (see the review by McPherson and Weltzin 2000). Exceptions are some drier desert grasslands where a continuous cover of fine fuels that would support large-scale fires did not occur (Buffington and Herbel 1965). Heavy overgrazing reduced grasses and fine fuels that were ignited by lightning during the late spring and early summer. Past fires eliminated or reduced the population of small trees and shrubs, maintaining low tree densities. Species that sprout after fires were kept in a subdominant position by repeated fires even if the plant survived. Under current conditions, many grasslands do not have the continuity of fuels to allow the uninhibited spread of fires. Without fire, trees that are adapted to a site will become established and regenerate successfully. However, the continued spread of invasive species such as Lehmann's lovegrass (*Eragrostis lehmanniana*), which is adapted to fire, may alter the present fire frequencies (McPherson 1995), and because of competition, these invasive species will impede the establishment of preferred native perennial grasses. The loss of ground cover and the concentrations of livestock in stream channels and wet meadows also resulted in severe erosion and loss of watershed condition on many grassland ranges.

The increase in woody species also has been linked to increased changes in the atmosphere or in the proportion of summer and winter precipitation. McPherson and Weltzin (2000) linked the increase in woody species to increased concentrations of carbon dioxide and other greenhouse gases in the atmosphere. Ranchers and researchers in southeastern Arizona, where most vegetation is dependent on summer rains, indicate that the proportion of annual precipitation has shifted to more winter precipitation when compared to the century-long average (Brown 1999, Valone and others 1998). This would favor the C_3 plants such as mesquite over the native C_4 grasses (Brown 1999); it also has been linked to declines in some key small mammal populations and increases in other species (Valone and others 1998). Aggressive fire suppression policies of Federal and State land management agencies have contributed to the situation by preventing potentially beneficial fires from burning. However, Allen (1998) indicates that fire suppression was not as effective as some maintain.

Many range restoration techniques call for the removal of trees. Restoration aims can range from complete removal of trees, although this is difficult, to the creation of savannas that retain a proportion of the trees as groups or individuals, to the creation of mosaics of grass openings and tree-covered zones. The retention of some residual trees and groups of trees would be beneficial for certain guilds of birds and as hiding and thermal cover for larger wildlife and livestock. Some prescriptions hope to establish a continuous and relatively dense stand of grasses and forbs so that prescribed fire eventually can be used to maintain the tree component in a less dominant position relative to the herbaceous species (Gottfried and others 1999).

Restoration efforts must be conducted with caution; one treatment will not be appropriate for all sites. Grasslands are complex ecosystems with different mixes of herbaceous and nonherbaceous species, site characteristics, and climatic conditions. Historical land uses have altered most areas, and the amount of change will affect present conditions and management options. Many areas that are perceived as grasslands are actually woodlands and never supported the grass communities that are projected to have been present. This includes areas adjacent to prehistorical and historical American Indian population centers where woodlands were heavily cut for domestic and agricultural purposes, and to areas surrounding Spanish-Mexican settlements and United States military posts in New Mexico. Many of the grasslands were actually savannas that contained scattered trees that provided the seed sources for an increase in tree density once fire events became less frequent. The presence of old or large trees would be a sign that caution is warranted. Some sites also have been so eroded and changed since the grasses were lost, that it may not be ecologically or economically possible to return to the original grassland communities. The introduction of invasive nonnative species, such as Lehmann's lovegrass and buffelgrass (*Cenchrus ciliaris*), has made restoration of native grasses more difficult (Weltzin and McPherson 1995). Long-term changes in greenhouse gases may make restoration difficult. Managers should recognize that even the best planned and executed treatments will not produce the desired results if weather conditions are not favorable. Except for ecologically critical areas, economical considerations may be the ultimate factor determining if, when, and what techniques should be employed to restore a grassland in a given location.

A wide variety of mechanical methods and prescriptions exist for use in reducing the woody species cover on grasslands (Vallentine 1971). The successes of these treatments depend on the characteristics of the woody species, including age, reproductive strategies, and stand densities; site characteristics such as slope, rockiness, terrain, and soil conditions; desired replacement vegetation; and soil seed bank availability. Herbicides also have been applied to mesquite and other woody species, but primarily on private lands. Climatic conditions before, during, and after an operation may determine if even the best planned and executed restoration treatment is successful. Forage increases after mesquite control, for example, will only occur if work is done during years of average or above average precipitation (Scifres and Polk 1974). It is recommended that livestock grazing be deferred from a site for a prescribed time after treatment so that new grasses can become established. Grazing by large and small wildlife species is more difficult to control. Deferral for 2 to 4 years has been recommended for many pinyon-juniper ranges (Gottfried and Severson 1993), but the amount of time will depend on the amount and condition of residual species, site potential, and weather conditions (Gottfried and Pieper 2000). Many of the failures of past pinyon-juniper treatments can be linked to premature grazing by livestock.

Unless satisfactory seed from residual grasses and forbs or from the soil seed bank are present, restoration will require seeding of native species. Forbs and shrub species could be seeded on some locations depending on site conditions and management objectives. It may be necessary and desirable to reestablish some shrubs, such as *Purshia* spp., which are important browse species. Although seed is usually not collected locally before a treatment, seed should come from a source as near to the project site as possible. The same principles that apply to tree seed provenance should apply to

Weather stations, such as this one on the Cascabel Watershed Area of southwestern New Mexico, provide data that are used to plan and evaluate grassland management and restoration treatments. (Photo by John Yazzie)

grass and associated species seed. Grassland species seed should, as closely as possible, be adapted to the site—for example, elevation and soils. Some desirable native species, such as black grama (*Bouteloua eriopoda*), may not be available from commercial seed companies, and adapted species may have to be substituted. The seeding of adapted nonnative species such as Lehmann lovegrass was common at one time but currently is discouraged.

Managers must consider the proper method of applying seed, whether seed should be drilled into the soil surface with a rangeland drill or if seed can be broadcast directly on the ground without cover. The creation of depressions in the soil surface in conjunction with seeding will enhance grass stand establishment in semidesert grasslands (Gottfried and others 1999). Each species has specific requirements. Timing of a treatment is important and should be done prior to or during the precipitation season for which the species is adapted. Optimum season for seeding will vary by site. Obviously, unprotected seed would be subject to granivory by insects and rodents or be blown or washed from the site. An established tree or shrub canopy that creates a moderate microclimate or reduces herbivore activity could result in reduced losses of viable seed and new seedlings. Restoration prescriptions should include monitoring before and for a period after treatment to ascertain if satisfactory results were achieved or if procedures should be modified for future success.

Research on the value of inoculating restoration sites with appropriate arbuscular mycorrhizal fungi to promote the survival and growth of the grasses could

A grassland restoration treatment designed to crush mesquite and seed native perennial grasses at the George Wright Pasture of the Diamond A Ranch Central Division in southwestern New Mexico. The crusher created depressions in the soil that enhance seeding success. (Photo by Ronald Bemis)

The sediment dam at Cascabel Watershed, New Mexico, is used to measure sediment production related to land management activities. (Photo by Gerald Gottfried)

enhance restoration successes. At present, there are no practical and affordable methods of reintroducing mycorrhizal plants to disturbed grasslands (Caplan and others 1999). Some work has been started on the relation between giant sacaton (*Sporobolus wrightii*) and mycorrhizae from a southeastern Arizona site (Elliott and others 1998). Seedballs consisting of clay, soil humus, grass seed, water, and a source of mycorrhizae may be an excellent way to disperse seed in disturbed areas (Caplan and others 1999). The ball should contain root segments of the native culture host plant with hyphae.

Watershed Condition—Many degraded grasslands have severe erosion problems; surface soils have been lost, and gullies and degraded stream channels are present. Most Southwestern grasslands are influenced by convectional, high-intensity thunderstorms during the summer monsoon. These thunderstorms have been linked to most surface runoff and erosion events, especially on degraded watersheds (Rich 1961). A change in surface conditions that increases infiltration capacity, whether by increasing the plant cover or by mechanical stabilization, should result in decreased surface runoff and soil erosion (Rich 1961). Increasing the herbaceous cover, especially in interspace areas between tree-influence zones will protect the soil surface from raindrop compaction and consequential soil pore sealing. Plant roots, especially the fibrous roots of monocotyledons, help buffer the soil from sealing, hold soil particles in place, improve the soil porosity and, thus, water movement in and through the soil, and increase water holding capacity because of organic additions to the soil. Plants also act as microbarriers to the free movement of surface runoff, allowing for greater infiltration and less soil movement within and from the larger watershed and the resulting losses in stream water quality. Increased infiltration provides more water for plant survival and growth. In many cases, changes in grazing season and intensities can reduce sediment loads from some grassland sites (Holechek and others 1998). Vegetation manipulations, by themselves, will not work on all sites; more expensive mechanical methods also may be necessary to prepare a site for restoration. Contour furrowing and trenching, ripping, or pitting has been used to retain water and reduce sedimentation (Brooks and others 2002). An early study in central Arizona found that surface runoff and erosion were reduced by a treatment that combined brush control, sloping of steep gully sides, placing cut brush in gully channels, and grass seeding (Rich 1961). Gullies develop when surface runoff is concentrated at a point where there is an abrupt change in elevation and slope and a lack of vegetative cover (Brooks and others 2002). Check dams and other barriers may be needed to retard head and down cutting and stabilize gullies until vegetation becomes established. Ranchers in Arizona's San Bernardino Valley are using local rocks to construct dams and structures to limit headcutting and to create meanders in relatively straight gullies. The meanders slow streamflow velocities, reducing potential bank cutting and increasing sediment deposition creating sites for the reestablishment of vegetation. Engineered structures may be necessary on larger streams to raise their base levels, thus reducing channel gradients and cross sections and streamflow velocities (Brooks and others 2002). Practices should not result in increased water pollution and should be conducted under existing Best Management Practice guidelines.

Prescribed Fire—The introduction of prescribed fire and the management of natural fire in grassland ecosystems are of major interest today. There is wide acceptance of fire as a paramount factor in maintaining native grass plant communities by reducing woody plants and removing the buildup of grass litter prior to Euro-American settlement. Landscape-level prescribed fires are usually designed to create mosaics of wooded and grass areas in addition to reducing general tree densities. In addition to reducing the cover of woody species, periodic fires may keep ungrazed grasslands vigorous by killing forbs and removing grass litter (Robinett and Barker 1996). However, some forbs may be desirable because they are high in the protein and moisture that are important for a number of animals (C.H. Sieg 2002, personal correspondence).

The prescribed burn at Baker Canyon in the Peloncillo Mountains, which straddle the border between Arizona and New Mexico, June 1995. The objective was to reduce tree densities and to create mosaics of grassland and tree dominated areas. (Photo by Gerald Gottfried)

Fire must be used judiciously after a thorough analysis of benefits and risks. A single prescribed fire may not achieve all of the ecosystem objectives for an area, and multiple burns may be necessary. Information is often lacking concerning pretreatment fuel loadings on an area dominated by trees. The Fuel Characteristic Classification System (FCC) (Ottmar and others 2003) has coverage for many pinyon-juniper sites and is currently being expanded to cover oak and juniper woodlands and savannas that are common in the Southwest. Several fire prediction models and systems may be appropriate in planning prescribed burns on sites with significant tree cover. A version of BEHAVE (BEHAVE PLUS), which was originally developed by Burgan and Rothermel (1984), was used in preparation for the 46,000-acre Baker II Burn in the southern Peloncillo Mountains of Arizona and New Mexico in June 2003 (P.A. Gordon 2004, personal correspondence). FCC information eventually will be linked to models such as BEHAVE and FARSITE (Ottmar and others 2003). Monitoring of fire effects is needed if a program is to be biologically and economically successful. Monitoring should be designed to include soil and hydrological responses as well as the common vegetation and wildlife inventories. The Revised Universal Soil Loss Equation (RUSLE) has been used to evaluate soil erosion from grasslands, but it tends to underestimate soil losses (Spaeth and others 2003). There is a shift to the Water Erosion Prediction Program (WEPP), especially for postfire predictions. A current effort in southeastern Arizona is attempting to define parameters for the Disturbed WEPP that could be used to predict soil losses following fires in semiarid grasslands (Paige and others 2003).

Wildfires can be beneficial, and each ignition should be evaluated prior to suppression actions. Fires obviously can be attacked if they are a potential danger to structures and improvements or if the landowner favors suppression; they can be monitored before a decision is made, or they can be allowed to burn. However, the last option must be taken with an understanding of the characteristics and needs of the specific area and of the actual and perceived impacts on encroaching population areas (Allen 1998). The Peloncillo Programmatic Fire Plan is an attempt to identify the prescribed fire and wildfire suppression philosophies of ranchers and landowners in this mountain range so that land managers may anticipate the appropriate actions (Allen 1999). Potential fire intensities may have changed since settlement times because of local buildups of larger woody fuels in savannas and riparian corridors. Current and desired fuel loadings are considerations, as are the habitat requirements of threatened, endangered, and sensitive (TES) species. The U.S. Fish and Wildlife Service and appropriate State agencies must be consulted on these issues.

Understanding the role of fire in maintaining habitats for rare species and the dynamics of fire in riparian areas will lead to more informed decisions (C.H. Sieg 2002, personal correspondence). The proper time to ignite prescribed fires is a concern where TES species habitats are present or where riparian areas could be adversely impacted. In semidesert grasslands, species are more influenced by fire season and frequency than by fire behavior (Steuter and McPherson 1995). Most wildfires in southeastern Arizona and southwestern New Mexico occur in the warm period as the annual monsoon develops, and most species are adapted to the relatively hot fires that occur prior to the growing season. Some authorities prefer cool season fires in the fall, winter, or early spring because they feel that less damage is done to the ecosystem. Others believe that cool season burning will cause a shift in species composition to favor shrubs and half-shrubs or will leave the soil surface open to accelerated erosion for a longer period. However, scientific data and documented observations are unavailable, and research is needed to determine if one season is better than the other. The research must address the impacts of the two burning prescriptions on other physical and biological ecosystem components, such as hydrology, sedimentation, arthropod populations, common animal species, and vegetation, as well as TES species. Ford and McPherson (1998) and Gottfried and others (2000) are studying some of these questions on the shortgrass prairies in eastern New Mexico and the oak (*Quercus emoryi)* savannas of southwestern New Mexico, respectively.

Fire frequency is a consideration in any prescribed burning program because frequent repeated fires may not allow for the recovery of many grass species. Pase and Granfelt (1977) recommended that at least 5 years separate fires so that herbaceous plants have an adequate period to recover and set seed. Kaib and others (1999) indicate that the grasslands of southeastern Arizona had low intensity fires every 4 to 8 years prior to the introduction of large herds of livestock. Research in southern Arizona has shown that grasslands in good condition can sustain a fire interval of between 5 and 10 years without a loss of productivity (Robinett and Barker 1996). Grass recovery depends on precipitation after treatment and the amount of herbivory (McPherson 1995).

Many prescribed fires in the Southwest occur in the early summer prior to the monsoon period. This also is a period when fire suppression and control resources for prescribed fires may be scarce because of wildfires. Prescribed fires are cancelled because of the lack of personnel and equipment during this period even when fuel and weather conditions are satisfactory and the risks to control are low. One suggestion is that the land management agencies consider creating a fire management organization, including personnel and equipment, to plan and conduct prescribed fires that is separate from the fire suppression organization (Bemis 2003).

Improving Soil Fertility—Heavy grazing and accelerated soil erosion have been linked to reductions in soil organic matter and soil fertility on many rangelands (Aguilar 1993). Any attempt to improve site potential and grass productivity must restore soil organic matter and associated nutrients; one recent method is to apply treated municipal sewage sludge to the soil surface. The Rocky Mountain Forest and Range Experiment Station conducted several experiments on grasslands in the Albuquerque area. Aguilar (1993), citing a number of studies by Fresquez and his associates, indicates that a one-time sludge application of 10 to 20 tons per acre increased plant production and ground cover without producing unsatisfactory levels of potentially hazardous constituents, such as heavy metals, in either soils or plant tissue. Blue grama (*Bouteloua gracilis*) yielded 1.5 to 2.7 times more production on treated compared to control plots. Total nitrogen, phosphorus, potassium, and electrical conductivity increased during the first year, and soil organic matter increased by the fifth year after application. Aguilar (1993) concluded that many areas with depleted soil nutrients would benefit from prescribed amendments of sewage sludge. A pot study in Texas found that applications of biosolids increased shoot growth of blue grama and tobosagrass (*Hilaria mutica*) because of increased soil NO_3-N (Mata-Gonzalez and others 2002). Low levels of biosolids and irrigation (40 percent of field capacity) resulted in increased root biomass, while high levels of both (80 percent of field capacity) resulted in a greater allocation of resources to the grass shoots. High applications of biosolids in the spring (34 and 90 $Mgha^{-1}$) produced greater soil nitrogen concentrations and shoot growth than did summer applications.

Another alternative is to use local materials as mulch. A recent study in the semidesert grasslands on the Santa Rita Experimental Range in Arizona examined the effects of combined mesquite overstory treatments and soil surface mulch treatments on herbaceous production (Pease 2000, Pease and others 2003). Overstory treatments consisted of removing the mesquite with or without sprout control and leaving the mesquite trees. These were combined with soil surface treatments consisting of mulches of lopped and scattered mesquite slash, commercial compost,

Prefabricated steel flumes, which can be assembled in the field, have been used to measure streamflow in remote, relatively inaccessible areas. (Photo by Gerald Gottfried)

A forestry technician downloads hydrologic data to a laptop computer from a Parshall flume installation at the Cascabel Watersheds, New Mexico. The small flume measures typical low streamflow events and the larger flume measures higher events. The 12 small instrumented watersheds will be used to evaluate the effects of warm and cool season prescribed burning and unburned conditions on the physical, chemical, and biological compenents of the oak savanna ecosystem. (Photo by John Yazzie)

and mesquite wood chips. The mesquite control treatment with no sprout controls resulted in a 79 percent increase in annual and fall herbaceous production of native species relative to the control plots; Lehmann lovegrass was not affected by the treatments. The mulching treatments had no effect on total annual, spring, or fall herbaceous production. The lack of response was attributed to low precipitation during the 4-year study, and to levels of mulch that were too low to impede evaporation (Pease 2000, Pease and others 2003). The lopped and scattered treatment did improve total soil nitrogen, plant available phosphorus, and soil pH relative to the controls.

Holistic Resource Management (HRM)—Savory (1988) introduced the idea of HRM into the United States from Zimbabwe. HRM is a grassland system or planning model that considers social, economic, and biological needs. However, it basically uses livestock to accomplish its goals. It involves a high-intensity, short-duration, time-controlled grazing system based on the phenological and physiological needs of the plant and animal species being managed, followed by periods of nonuse. The number of paddocks in the system is a function of the length of the deferment period that the plants require to maintain good health and high vigor; this usually is between 45 to 90 days, depending on the timing and amount of precipitation. Paddock use is scheduled to utilize both warm and cool season plants. Livestock are moved on a fast rotation during the growing season and are only allowed to remove 5 percent of the forage crop on each visit to a paddock before it is deferred. HRM requires a high degree of balance between annual forage consumption and annual forage production, the latter of which depends on annual climatic conditions, and HRM requires intensive herd management to achieve the desired effects.

High stocking densities are projected to improve water infiltration into the soil, increase mineral cycling, increase the number of plants consumed, improve the leaf area index, improve the distribution of grazing, increase the period when green forage is available, and reduce the percentage of ungrazed plants. However, Savory's system is controversial (Holechek and others 2000, Sayre 2001), and some of the claimed benefits have not been proven—for example, increased water infiltration has not been found, and sedimentation is higher under this system than under more moderate grazing systems. Holechek and his associates (1998, 2000) and Sayre (2001) provide a more complete analysis of HRM. They indicate that it may not be appropriate for arid grassland ecosystems because short growing seasons minimize the value of repeated periods of defoliation and nonuse. HRM may be best suited for monocultures of extremely grazing-tolerant grasses in subhumid environments, but it may not be suitable for lands where the goal is to manage for plant species diversity and variation in residual covers (C.H. Sieg 2002, personal correspondence). Nevertheless, some ranchers have adopted the short-duration grazing and have been satisfied with the results (Sayre 2001).

It is usually accepted that short-duration grazing is not the same as Savory's HRM, but many researchers have related their findings to HRM (Holechek and others 2000). Several studies have compared short-duration grazing to continuous grazing systems. A review of results indicates that the two systems produce similar results when stocking rates are equivalent with respect to forage production, plant succession and range condition, livestock productivity, and harvest efficiency (Holechek and others 2000). Short-duration/high-intensity grazing without the pasture rotation system has been used on some difficult restoration sites. For example, cattle have been grazed to rehabilitate mine tailing sites in central Arizona. Hay is often spread to encourage the cattle movements around the tailing site where they break up the tailing material by hoof action, fertilize and mix organic material into the substrate, and create small depressions to catch precipitation (Wheeler 1998). Hay also serves as mulch, reducing soil evaporation and soil surface temperatures. Wheeler (1998) claims that feeding also stimulates new growth on existing plants, and microorganisms associated with the hay and cattle droppings improve soil fertility and help buffer the soil pH of the tailings. Irrigation systems are available on many tailing sites. Short-duration/high-intensity grazing has been tried on a severely eroded site in the San Bernardino Valley of southeastern Arizona (Gottfried and others 1999). Mechanical restorations could not be used on the site because of valuable archeological resources. Native hay was spread on the fenced site before cattle were introduced for less than 3 days, and the site was seeded with a mixture of native grasses. Most of the resulting grass seedlings—cane beardgrass (*Bothriochloa barbinoides*) and, Arizona cottontop (*Digitorial californica*)—were from the hay. Initial results have not been as good as expected and were similar to the control area because of summer droughts and insect herbivory, both of which are common problems in the Southwest.

Watershed Management: Best Management Practices

Author: Penny Luehring, Southwestern Region

The Clean Water Act of 1972 (Federal Water Pollution Control Act, amendments of 1972; PL 92-500, 33U.S.C. 1311-1313, 1315-1317) defines a Best Management Practice (BMP) as: "a practice or a combination of practices, that is determined by a State (or designated area-wide planning agency) after problem assessment, examination of alternative practices and appropriate public participation, to be the most effective, practicable (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by non-point sources to a level compatible with water quality goals."

Nonpoint source pollution is water pollution that originates from many indefinable sources. Nonpoint source pollutants are generally carried over, or through, the soil and ground cover via streamflow processes. Resource management activities likely to occur in grassland ecosystems that may be considered nonpoint sources include runoff from grazing, construction, revegetation, restoration, prescribed burning, wildfire suppression, pest or invasive-plant control, developed recreation sites, mining, road construction, and road maintenance.

Best Management Practices must be designed on site-specific basis for planned activities, taking into consideration the degree of surface disturbance anticipated, drainage patterns, climate, slope, soil erodibility, and proximity to stream channels. BMPs may be operational or administrative. *Operational BMP* examples include putting drainage dips in roads, leaving untreated buffer strips between an activity and the stream channel, or applying mulch. Examples of *administrative BMPs* include controlling livestock numbers and season of use, timing of construction activities, and designing roads to minimize stream crossings. Most State water quality regulatory agencies and Federal land management agencies maintain handbooks of water quality protection techniques and tools that can be used to prescribe BMPs.

Monitoring of BMPs is a critical part of the process. Monitoring should be done first to see if the BMPs were implemented as prescribed. Then, BMPs should be evaluated as to their effectiveness so that prescribed protection measures can be continually improved and BMP knowledge can be recycled to benefit future projects.

Road Management

Author: Bill Woodward, Southwestern Region

Grassland roads provide legal access necessary for administration and use of Federal, State, and private lands. Federal grasslands are often fragmented, and as a result, grassland roads are generally branches of State, County, and other road systems that serve as primary access roads to these areas. Land fragmentation usually results in significant increases in road densities across previously undivided landscapes (Mitchell and others 2002).

Most grassland roads are for use by high clearance vehicles. Road densities on Federal lands are often less than 2 miles per square mile (equivalent to having a road around each section of land). For most grassland activities, such as cattle grazing and hunting, this is adequate to meet management objectives. However, in areas where oil and gas operations are a primary activity, road densities as high as 6 miles per square mile may be necessary to maintain wells and related equipment. One study in eastern Colorado found road densities for subdivided grasslands were between these two values. Road densities on two subdivided ranches were 3.4 to 5.7 miles per section compared to between 0.6 and 1.5 miles per section on two neighboring intact ranches (Mitchell and others 2002).

Existing roads and road construction and maintenance traditionally have been a concern in forest and grassland management. Older roads were often constructed along riparian corridors and across sensitive meadows and wetlands. Most of these roads were poorly designed, if designed at all, and had inadequate drainage. Poorly designed or constructed roads, properly designed roads that are not maintained, and roads with inadequate drainage and poor culvert design contribute significantly to erosion and sedimentation. Efforts to correct these problems through construction and maintenance have been hindered by budgets, priority safety items, and environmental considerations.

Today's emphasis for grassland roads on Federal lands is to locate, design, construct, maintain, and manage to minimize erosion and sedimentation effects and to reestablish wetland and riparian areas. Offroad use is often prohibited or restricted to minimize creation of unneeded travelways and the spread of noxious and invasive vegetation species. Unneeded roads are to be decommissioned or converted to other nonroad uses. Road conditions and management emphasis on private lands, which often have heavy use, are variable.

Standard USDA Forest Service manuals are available to ensure proper design, construction, and maintenance of roads and for implementation of soil

An illegal four-wheel drive vehicle road on steep slopes in the SP Crater area north of Flagstaff, Arizona. Such activities degrade vegetation, soil, water, and esthetic resources. (Photo by John Yazzie) and water conservation practices within grassland communities. Similar manuals also are available from other government agencies and from educational institutions.

Recreation Management

Author: Rick Atwell, Southwestern Region

Pressures for increased recreational opportunities are increasing throughout the West as the region's primarily urban populations continue to grow. While forests and associated lakes and streams are the primary recreational focus for many in the arid Southwest, grasslands also are being impacted by increased road traffic and offroad travel. Certain grasslands, especially mountain meadows, receive heavy use by the recreating public, and sometimes these sensitive areas must be protected from overuse.

One example: the measures used to protect the Kiwanis Meadow on the Sandia Ranger District of the Cibola National Forest. The meadow is used by hundreds of thousands of visitors who are traveling along the 1.5 mile trail between the Sandia Mountain Aerial Tramway and the Sandia Crest observation point. These two destinations receive the highest visitor counts for National Forests in northern New Mexico. The District employed a three-part plan. The first part was to reroute the popular Crest Trail from the meadow to its east side, and a second step was to build a buck and pole fence around the meadow. This provided a physical barrier to encourage visitors to stay on the established trails and not use the meadow for lounging or picnicking. The materials for the fence came from other areas of the District, often resulting in the creation of beneficial small wildlife openings. The third step was to place directional and informational signs around the area.

Integrated Weed Management

Author: Gene Onken, Southwestern Region

The introduction, adaptation, and spread of nonnative invasive plant species have become a serious threat to native grassland ecosystems of the Southwestern United States. Significant spread of invasive weeds in the Southwest has occurred relatively recently, especially during the past 15 years. Because of more xeric climate, harsh sites, and marginal soil productivity, the invasive weed problem reached serious dimension in the Southwest later than in other parts of the country. In the Southwestern Region of the USDA Forest Service, remoteness may also be a factor because the Region has a less dense road system than more highly developed agricultural production areas in other parts of the Southwest. These factors translate to a somewhat slower initial rate of weed introduction and spread.

Some of the most invasive of species evolved in Eurasia where the climate is similar to that of the Southwestern United States. Once these species were introduced here, the infestations spread rapidly. The weeds were and are often able to outcompete the native species for available moisture and space on the landscape. The overall problem is now highly significant both from an ecological and an economic perspective. To sustain the native grassland species, both *integrated* and *adaptive* management actions are now required for managing the invasive species threat.

Integrated weed management (IWM) is a systems approach to managing undesirable plants. It is defined in the Federal Noxious Weed Act of 1974 (PL 93-626, January 1975, U.S.C. 2801-2814) as:

> [A] system for the planning and implementation of a program, using an interdisciplinary approach, to select a method for containing or controlling an undesirable plant species or group of species using all available methods, including…education; prevention; physical or mechanical methods; biological control agents; herbicide methods; cultural methods; and general land management practices.

The concept of integrated weed management has already been adopted by most of the Federal and State agencies in the Southwest.

IWM involves using the best control techniques available for the target weed species. IWM requires a planned and coordinated program to limit the impact and spread of the weed. Control methods should be determined by: management objectives for the land, effectiveness of the control technique on the target species, environmental factors, land use, economics, policy and legal restrictions, safety to humans and the environment, and the extent and nature of the infestation.

An IWM approach for addressing invasive plant problems is particularly suited for an *adaptive management strategy*, which provides a way to describe and evaluate the consequences of dynamic and rapidly changing invasive plant populations on the landscape. Within a rather short time, invasive weeds spread rapidly from existing infestations into new locations. New species also may become established in any given locality. New technologies for weed control are continually being developed as new biological agents and herbicides become available.

An adaptive management strategy requires site-specific explanations of what actions the land managing agency or landowner will take under various conditions and what the environmental effects will be for those weed control actions. Weed management actions are continually reevaluated by land managers as monitoring indicates changes on the landscape and as new control options are developed. This reevaluation is done within the framework of the original integrated weed management plan. Then the next control actions are appropriately adjusted or adapted

Table 8-1. Map scale, vegetation, and appropriate imaging technology.

	Vegetation		Example			
Map scale	Potential	Existing	Potential	Existing	Image data type	
1:1,000,0001	Class	Lifeform	Close forest	Coniferous forest	AVHRR	
	Subclass	Lifeform	Mainly evergreen	Coniferous forest		
1:500,000	Group	Lifeform	Temp. evergreen needleleaf	Coniferous forest		
1:250,000	Formation	Structural stage	Pine forest	Seedling/sapling	Landsat MSS	
1:126,720						
1:100,000	Series	Cover type	Ponderosa pine		Landsat TM	
1:50,000	Subseries	Domance type	Ponderosa pine/ gambel oak		SPOT XS SPOT Pan	Air photos
	Association	Community type	Ponderosa pine/ Arizona fescue/ gambel oak phase	Ponderosa pine/ mutton bluegrass	Digital camera videography	Air photos

to the new and changing conditions and technologies. Thus, an adaptive management strategy is a continuing process of:

- Management action
- Monitor the results
- Evaluate the changes
- Adapt the treatment plan
- Implement the next (adapted) management action
- Monitor the results

An adaptive management approach shows how weeds will be treated without listing individual species or individual sites. Managers should specify an approximate number and/or percentage of treatment acres across a geographic area. This estimate forms the basis for predicting the environmental consequences and for defining the various weed management strategies.

Now that nonnative invasive species have become firmly established within the native grassland ecosystems of the Southwestern Region, aggressive and continuing eradication or control of new or existing nonnative invasive populations or infestations must be part of management to prevent spread or to restrict unacceptable impacts on existing native plant communities. Economic, environmental, or legal constraints may prevent eradication or the desired degree of invasive weed control. Therefore, implementing integrated weed management practices under an adaptive management strategy provides the most practical approach toward addressing this problem.

In summary, invasive plant species will continue to impact our landscapes and place our natural grassland ecosystems at risk. A perpetual expenditure of effort and resources will be required to sustain the grasslands.

Remote Sensing, GIS Applications, and Database Management

Remote Sensing

Author: Bill Krausmann, Southwestern Region

Remote sensing has a demonstrated potential to aid grassland managers in maintaining long-term viability of the resource. Remote sensing can be defined, in this context, as the analysis of grassland responses to electromagnetic radiation as collected and recorded in an image format. Examples of applications in remote sensing that deal with range management aspects of grassland ecosystems date back well into the 1930s (Tueller 1989).

The types of imagery and imagery-based analytical processes that can be applied to grassland management have grown exponentially over the past 25 years. Several forms of satellite imagery are currently available for analysis, in addition to various forms of aerial photography—a standard tool for managers. New satellite imaging systems are being brought on line each year, further increasing the potential value of remote sensing as a management tool. The marked increase in available imagery and the development of new techniques are fortuitous because of a need to augment standard inventory and monitoring methods to meet current requirements placed on resource management agencies.

Vegetation Mapping—Managers are interested in the type, distribution, and condition of grassland vegetation, and for grazing analysis, the forage base as it occurs across space and time. While remote sensing techniques have been utilized since the 1960s to map parameters of vegetation, the number of useful and fully applied procedures or techniques is quite low (Tueller 1989).

The level of detail that can be mapped within vegetation communities is directly related to the scale of a given project. Selecting an image data source that is appropriate for the scale of the mapping project is a critical element in the successful application of remote sensing in grassland management (table 8-1).

Aerial photography has long been a proven tool for mapping vegetation. Research has shown that photography acquired at a scale of 1:10,000 is optimum for mapping vegetation at the ecological site level (Tueller 1979). As photo scale becomes smaller, the range of vegetation units that can be detected becomes more general, as indicated in table 8-1. It is inappropriate, for example, to use Landsat Thematic Mapper imagery for a project where mapping at a species association level is of interest. On the other hand, mapping series level vegetation across an entire National Forest using aerial photography could be done but would be extremely laborious and expensive.

Computer mapping of vegetation types revolves around the processes of supervised and/or unsupervised image classification. In the supervised approach, areas of homogeneous training sites of known vegetation type are identified on the imagery. There are typically several training sites per class of vegetation. Statistics derived from the spectral response of the pixels within the training sites are used by one of several classification algorithms to separate the image into specified vegetation classes.

Generally, unsupervised classification seems to work best on grasslands (Tueller 1989). The analyst provides the computer with the number of clusters to be derived and parameters that drive cluster merging and splitting in the unsupervised process. Pixels in the image are then clustered into groups with similar spectral response by the computer. The groupings that are generated in this process are called spectral classes. They represent areas with similar spectral response. It is the analyst's responsibility to develop information classes from the spectral classes. The classes are developed by building relationships between the spectral classes and areas of the surface with known vegetation cover. Information classes can represent, among other things, plant communities, grazing allotments, or range improvements.

Mapping vegetation in rangelands, including grasslands, using satellite data is a difficult process in the Southwest. Several problems combine to reduce classification accuracies. Some of these problems include high soil background response, spatially heterogeneous precipitation patterns, spatially heterogeneous grazing patterns, and timber-covered range allotments. On average, for classifications produced at the vegetation series/sub-series level (table 8-1), overall classification accuracies between 65 and 75 percent could be expected.

Change Detection—The detection of change over time in grasslands and other rangelands is perhaps the most significant application of remote sensing to range management. Remote sensing instruments can produce imagery at scales from the site level to subcontinental areas that can be used to detect fluctuations in productivity.

Digital change detection involves using a computer to compare the spectral response of two or more images acquired on different dates. The comparison is performed at the pixel level and requires that the imagery data sets be accurately coregistered. There are several examples of change detection methods applied to range management issues (Chavez and MacKinnon 1994, Knight 1995, Pickup and others 1993, Ringrose and others 1999, Wallace and others 2003). Several algorithms have been developed for change detection, and a review of methodologies exists (Singh 1989), as does an assessment of Landsat Thematic Mapper imagery for change detection (Fung 1990). The most common change detection methodology is image differencing. In image differencing, change between two images is highlighted by subtracting one image's pixel values from the other.

Other Applications—Remote sensing has been used to monitor and evaluate grazing management. Pickup and Chewings (1988) used Landsat imagery and animal distribution models to estimate the distribution and grazing pattern of cattle on large pastures in Australia. Utilization levels of 25 to 40 percent are recommended in the semiarid Southwest (under 300 mm of precipitation annually) (Holechek and others 1998). Acquiring information on utilization levels requires considerable fieldwork. Repetitive aerial photography, videography, and digital camera imagery can provide baseline data on range readiness, utilization, livestock distribution, or other parameters that may reduce the amount of field work required to support management decisions.

Soil loss from gullying, overland flow, or eolian processes is a significant concern of grassland managers. Researchers have mapped soils, assessed soil loss, and measured gully erosion (Pickup and Nelson 1984, Westin and Lemme 1978) primarily using various forms of aerial photography.

Conclusion—In 1989, Tueller described the future of remote sensing applications in range management as "hazy" (Tueller 1989). The same could probably be said today. While research continues on applications of remote sensing to rangeland management, much of it relates to satellite monitoring of large, semiarid regions, and the questions being addressed are subcontinental in nature. This work does not address the needs of range managers working with several allotments on a USDA Forest Service Ranger District. If remote sensing is to become a standard management tool, procedures that directly address range readiness and utilization in a cost-effective manner at large scales must be developed.

New tools such as MODIS (a 36 band imaging radiometer) and the digital camera, combined with Landsat Thematic Mapper imagery and aerial photography, may provide opportunities to better monitor rangelands, including grasslands, at both the macro and micro scales. It is hoped that within 20 years much of the data required by managers may be derived from imagery.

Geographic Information System as a Tool for Managing Grasslands

Author: Pat Frieberg, Southwestern Region

The emergence of Geographic Information System (GIS) technology in the past few years has provided the grasslands manager with a decision support system that facilitates the decisionmaking process when addressing management issues. GIS provides the analytical capabilities for spatial data mapping, management, and modeling.

GIS is defined as a collection of computer hardware, software, geographic data, and personnel that are designed to hold, manipulate, analyze, and display all forms of geographically referenced information (Environmental Systems Research Institute 1995). GIS can address the impacts of multiple variables simultaneously—for example, to determine the capability of a grassland to sustain livestock grazing. Using basic layers of slope, aspect, vegetation, and water sources, along with certain known habits of livestock, one can model the grazing pattern and subsequent ability of the grassland to sustain grazing. Once the forage production and utilization information is measured and collected in the field, it can be entered into the model. The model then calculates the carrying capacity based upon actual use, and the criteria are entered into the model. The Apache-Sitgreaves National Forest in east-central Arizona is an example of a Forest that has used GIS for this purpose. The Range Management Staff developed a model using forage production, soil stability, distance from water, and steepness of slope to determine grazing capacities for allotments scheduled for new or revised allotment management plans.

Using the relationship between forage production and forest overstory density, a GIS map depicting herbaceous forage production classes for each allotment was developed and field checked. The result was an estimate of forage production in pounds per acre for each class. A percentage of allowable use for each of these forage production classes was established based upon range condition and management strategy. GIS maps showing limitations for soil stability, distance from water, and slope gradient were generated and used to further refine the percent allowable use in each forage production class. When the refined allowable use percentage was multiplied by the forage production in pounds per acre, the result was the pounds of forage available for consumption. This was multiplied by the GIS acres in each forage production class, then the total pounds of forage available for consumption by wildlife and livestock were computed for each pasture, allowing the range manager to estimate the livestock capacity for each pasture. The capacity of all pastures in the allotment was added to establish a capacity for the allotment.

GIS can be used for virtually every facet of resource management. Providing a spatial inventory of different aspects of the grassland resource, such as the location of noxious weeds, vegetation types, roads, and so forth, is a common use. Combining the database and mapping capabilities within the GIS system permits a variety of analyses to be performed, for example, analyzing counties and communities by economic characteristics. A study in the Sevilleta Long Term Ecological Research Program (LTER) is examining prehistoric and historic land use and exotic plant invasion by overlaying vegetation maps with archaeological site maps. Typically, GIS data are linked to large databases for decision support, such as the Forest Service's NRIS and INFRA databases and the Bureau of Land Management's Rangeland Information System (RIS). Remote sensing data can also be incorporated into GIS layers for analysis and for change detection purposes.

GIS data are increasingly being linked with modeling programs to create management applications. Some examples of the variety of applications:

- FRAGSTATS, a computer software program designed to compute a wide variety of landscape metrics for map patterns.
- The Agricultural Research Service's Arid Basin (ARDBSN) model to predict the amount of runoff resulting from rainstorm events.
- Predictive wildlife habitat models for current management or for potential reintroduction—for example, mountain lion (*Felis concolor*), black-tailed prairie dog (*Cynomys ludovicainus arizonensis*), and aplomado falcon (*Falco femoralis septentrionalis*).
- Texas A&M's PHYGROW, a hydrologic based plant growth simulation model using soil

characteristics, plant community characteristics, and weather data for a particular location to predict forage production.

GIS can also be an important tool in monitoring, not only to provide a spatial context to the monitoring effort but also to select sampling sites based on specific criteria, analyze changes in parameters over time, and for additional modeling to assess the temporal and spatial variability of ecosystem processes in an area.

Forest Service Corporate Databases That Cover Grasslands

Author: Reuben Weisz, Southwestern Region

The USDA Forest Service is currently investing in four major database projects that may be useful for grassland assessments:

- The Geographic Information System (GIS) Core Data Project
- The Automated Lands Project (ALP)
- The INFRASTRUCTURE (INFRA) Project
- The Natural Resources Information System (NRIS) Project

These database projects provide a set of interrelated databases and computer applications containing spatial (map) and tabular (numbers) data, collected and recorded in a consistent manner. The GIS Coordinator for a particular grassland area will be able to provide more details about applications of these databases.

Briefly described, the GIS Core Data Project is putting into place, "wall to wall" across all Forest and Grasslands boundaries, 15 standard GIS layers. These contain the minimum data required to do business everywhere in the Forest Service. Broadly speaking there are three categories of GIS layers:

1. **Land Information Layer—**The ALP database manages the spatial and tabular data associated with information about ownership, jurisdiction, land surveys, and restrictions and rights. Examples of ALP data include information about who owns a grassland area and what restrictions and rights apply to it.
2. **Constructed Features Layers—**The INFRA database manages spatial and tabular data describing those things in the ecosystem that are constructed or created by people. Examples of INFRA data include information about roads, trails, allotment and pasture boundaries, allotment and permit management information, and range improvements.
3. **Natural Resource Layers—**The NRIS database manages the spatial and tabular data about those parts of ecosystems that occur naturally such as air, water, terrestrial ecologic units, existing vegetation, threatened and endangered species occurrences, topography, water and watersheds, and the human dimension. In a given grassland area, this might contain useful information about water quality, water uses and water rights, threatened and endangered species, and watershed condition.

GIS Applications for Wildlife Management

Author: Bryce Rickel, Southwestern Region

GIS applications have been expanded to provide land managers with tools specific to wildlife management. Two examples are the Southwest Wildlife Information System and the Habitat Quality Index.

Southwest Wildlife Information System (SWIS)—This is an ArcView application that has been developed to provide field personnel with an easy way to search, query, and analyze basic wildlife species habitat relationships across landscapes. SWIS allows the user to perform species/habitat tabular and/or spatial searches and queries using ArcView. The focus of the current application is providing information at the Forest and project levels. SWIS also allows the user to display Arizona and New Mexico Natural Heritage data.

Habitat Quality Index (HQI)—This model is an ArcView application that allows Forest and District biologists to develop their own species habitat quality models. The primary modeling approach employed is simple and has been used for almost two decades. The idea behind the model is that a particular habitat type, per season, has certain cover and forage values for a particular wildlife species. An HQI model for a species will produce a GIS map with habitat qualities for each habitat type (or polygon) across a landscape.

Database Use to Assess Effects of Grazing on Southwestern Biodiversity: An Example

Authors: Curtis Flather and Patrick Zwartjes, Rocky Mountain Research Station

The decisions the Forest Service has made regarding the use of public lands for livestock grazing have become a contentious issue in the Southwest, resulting in many of these decisions being challenged (particularly through litigation) by a variety of parties with competing interests (for example, environmentalists and cattle ranchers). The Forest Service has recognized that National Forest managers and biologists have little information on the impact of grazing (both by permitted livestock and by native ungulates) on the various animal and plant species, as well as the overall biodiversity, found on National Forest System lands. To improve management decisions regarding species that may be at risk from permitted and/or native ungulate grazing, the Forest Service's Washington Office entered into a cooperative project with the Forest Service Southwestern Region (Region 3) and the Rocky Mountain Research Station (RMRS). The objectives of this project are to:

- Describe and classify habitat types on grazed lands in the Southwest.
- Determine species that are sensitive to grazing.
- Determine source habitats for sensitive species.
- Develop distributional maps and ecological/life history summaries for each species.
- Describe the effects of specific grazing management regimes on these species and their habitats.

Scope, Organization, Goals—Data acquisition for this project is focused on the compilation of existing information, both at a landscape level (including species distributions and general habitat associations), and at the individual species level (which includes data on specific habitat requirements and other pertinent ecological information). The geographic scope includes all land areas (not just National Forest System lands) in Arizona and New Mexico, with a taxonomic scope that includes select species of terrestrial vertebrates, aquatic vertebrates, and plants.

The responsibilities for the different objectives have been assigned to three units within the RMRS:

- **Fort Collins:** Characterize regional biodiversity patterns; develop models of species occupancy.
- **Albuquerque:** Identify species sensitive to grazing; develop species accounts for each with detailed habitat, ecology, and life history information.
- **Flagstaff:** Identify broad vegetative zones in the region; assess impacts of grazing on these zones as well as on select individual plant species.

In general, the goals of this project are to provide fundamental information on species distribution, occurrence, ecology, and life history for use by range and forest managers. It is important to note that this study is not intended to be a regional viability assessment, nor is it an assessment of Region 3's grazing program; rather, it is a tool to assess the habitat needs and vulnerabilities of individual species.

Geospatial Analyses—The objectives of this segment of the project are based on a geographic approach, one that will develop a geospatial database of species occurrence information as well as geographic information on land cover and land use throughout the Southwestern Region.

The format for this database is an ArcInfo 8/Microsoft SQL relational database, distributed as ArcInfo coverages. The content is divided into three map types:

- **Species Occurrence Maps—**Two general categories: (1) maps using species point observation data, collected from museum collection records, survey data (for example, USFWS Breeding Bird Survey), and biological atlases; and (2) predicted occurrence range maps, based on habitat associations, and produced by the National Gap Analysis Program. Of particular importance will be the capacity to generate species lists by spatial queries, such as within a particular National Forest or grazing allotment.
- **Biodiversity Maps—**Overlays geographic information for different species within a broad grouping (for example, birds, reptiles) to estimate species richness and relative biodiversity among geographical areas. Potential applications include identifying areas with the greatest number of species, and ranking different areas with respect to species richness.
- **Base Layer Maps—**Includes maps of topography/digital terrain, land use, land cover/vegetation, administrative boundaries, and so forth.

The main product for this project will be a Rocky Mountain Research Station General Technical Report (GTR) that will detail the methods and procedures utilized for this analysis, as well as the results of an analysis of biodiversity patterns in the Southwestern Region. The GTR will include a CD-ROM containing software and map files for analysis by the users. This CD-ROM will contain more than 30 GIS coverages, range maps for more than 800 species, more than 500,000 point locations for individual species, analysis algorithms, and a simple user interface.

Vertebrate Species Accounts—The project was designed to examine all terrestrial and aquatic vertebrate species in the Southwestern Region, and (1) identify those species with the greatest potential to be negatively impacted by grazing, and (2) collect detailed habitat and life history information into individual accounts based on information from both the published literature and the expertise of vertebrate zoologists working in Arizona and New Mexico.

Panels of zoological experts were assembled according to several broad categories of vertebrate taxa:

- reptiles and amphibians
- grassland-desert scrub birds
- woodland birds
- riparian birds
- small mammals
- carnivorous and ungulate mammals

The complete list of participants will include more than 40 zoological experts from Arizona and New Mexico, including university scientists, Federal and State scientists, wildlife experts, and private experts and consultants.

The panel process was organized into two stages, with an intervening period during which the Albuquerque RMRS developed draft species accounts based on the published literature:

- **Stage I—**The panel of experts considers and discusses the ecological and habitat needs of each species, followed by each panelist recording their assessment of the direction, magnitude, and likelihood of grazing impacts on this species. Species were selected for further consideration if at least one of the panelists determined the species had at least the *potential* to be severely negatively impacted (regardless of likelihood), or that the species is known to be negatively impacted (but of unknown magnitude).
- **Stage II—**Draft species accounts, developed for all species selected in stage I, are reviewed by panel members, who fill in gaps in information from the published literature, suggest additional sources of information, and make selections from a series of menus that contain habitat and life history descriptors that serve to characterize the ecological needs of each species. In addition, the stage II panel has the authority to either remove species from the database that were selected in stage I, or to add species, which they consider to be erroneously excluded.

The final version of the species accounts will be based on the inputs of the panelists in combination with the information from the published literature. The database of accounts will be contained on a CD-ROM in Microsoft Access format, with a user interface that allows for searches based on species names, or queries that will generate a species list based on shared habitat descriptors in the menu selections, such as general habitat association, specific attributes of habitat (tree heights, grass densities, and so forth), season of use, grazing effects, and others. Individual accounts will be able to be viewed on screen, or printed out individually according to the needs of each user. In addition, the CD-ROM will contain a ProCite file of all literature cited, and the Albuquerque RMRS will deliver to Region 3 the complete collection of literature used to develop all species accounts. We anticipate the final product to contain more than 300 terrestrial vertebrate and more than 50 aquatic vertebrate species, with a literature database of more than 2,000 entries. As of publication of this Assessment (2004), the species accounts database is well under construction, with a large percentage of accounts completed.

Progress and Schedules— The geospatial analyses and the panel process were largely completed by the start of Federal government fiscal year 2003. Development of the GTR, user interfaces for the CD-ROMs, and editing and completion of the individual species accounts are continuing as of the publication of this Assessment. The complete package should be available to forest managers and biologists as a new management tool by 2006.

Grasslands Education and Communication in the Southwestern Region

Author: Jean Szymanski, Southwestern Region

Education for public and private land managers and for the interested public is vital for sound grassland management in the Southwest. Managers need the knowledge to be good stewards of their lands and to conduct ecologically and, especially for private landowners, financially sound treatments. Educated ranchers and land managers can help with many activities on their own or on leased grasslands. One example would be public agency managers who understand how to monitor grassland conditions, thus allowing educated decisions on livestock stocking levels and rotations and providing a basis for discussions on permits with the public, who may question their management decisions. Education of teachers, especially at the high school and university level, often pays large dividends.

The Coconino National Forest has a session on rangelands as part of the Arizona Natural Resource Conservation Workshop for Educators program. In this session, participants lay out transects to measure plant composition and amount of ground cover. The emphasis in this program is on rangelands as a type of landscape with many uses, not just grazing.

The Cuba Soil and Water Conservation District is the lead sponsor of the New Mexico Forestry Camp; the Forest Service is one of the many co-sponsors, with involvement from the Regional Office and the Santa Fe, Cibola, and Carson National Forests. Two sessions on grasslands are conducted at this weeklong camp for New Mexico youth (13 through 18 years old). One session focuses on identification of forbs and grasses, and the other focuses on grazing studies, soils, noxious weeds, and weed control methods.

The Malpai Borderlands Group conducts three “Ranching Today” workshops on the Borderlands near Douglas, AZ. These workshops are 3 to 5 days long and attended by ranchers, the public, and Nature Conservancy members. People from throughout the Western United States often attend these workshops to learn if the Malpai model would work in their areas. The workshops discuss examples of grassbanking, private land conservation easements, ongoing research, and endangered species. The Coronado National Forest participates in these activities. The Malpai Borderlands Group also supports activities at Douglas High School related to the supervised breeding of endangered Chiricahua leopard frogs (*Rana chiricahuensis*), and supports local students who serve as summer interns on sponsored research studies.

Most of the Western State sections of the Society for Range Management hold weeklong summer camps for high school students to learn about rangeland management. The Arizona Section has conducted camps in the State since the 1960s. Camps have been held at the Sierra Ancha Experimental Forest or at Mormon Lake.

The Cibola National Forest, as a partner with the Playa Lakes Joint Venture, has access to several Grassland Education "trunks" to use as a teaching tool. The trunks contain books, tools, and videos to aid in educating about grasslands. The Kiowa National Grasslands has been involved in outdoor education days, where countywide third graders are brought out to the National Grassland and given environmental instruction about vegetation. The National Grassland also serves as a site for workshops for landowners on grazing management and riparian enhancement.

The Washington Office of the USDA Forest Service is producing a series of posters, similar to the Smokey Bear poster series, that relate to grassland ecology and management. These posters will be distributed throughout the country. Some topics are:

- Grassland fire ecologies
- Rainfall in grasslands
- Noxious weeds
- What keeps a prairie a prairie?
- State grasses
- The cycle of life on a grassland
- The anatomy of grass
- The function of grass roots

Other Federal agencies have education programs related to grassland and general range management. The USDA Agricultural Research Service in Boise, ID, incorporates students in its watershed program to provide an educational experience and to broaden their awareness of the biological and physical processes, landscape attributes, and social and economic factors that affect viability of range management (Northwest Watershed Research Center, USDA ARS, undated).

Educational activities also include measures designed to inform the public about grassland ecology and management decisions. The Kiwanis Meadow near the Sandia Crest, north of Albuquerque on the Cibola National Forest, receives heavy recreational use because of its proximity to the Sandia Tram. Recent efforts to protect the alpine meadow by rerouting the trail have included the placement of directional and informational signs. The directional signs show visitors the new routes and where traffic is prohibited, and the informational signs explain why the Forest undertook the project. Signs that explained that meadow protection benefited wildlife were especially effective.

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Appendix:
Discussion Question Summary

As part of the assessment effort, the team hosted a series of conference calls to supplement the information that had been collected and to identify any gaps in the assessment report. Persons participating in the conference calls included USDA Forest Service Southwestern Regional Office program managers, National Forest planners and resource staff officers, District Rangers and resource specialists, researchers, and academia. Participants were asked nine discussion questions, and they provided their answers either during the calls or through written responses. This appendix is a summary compilation of all the responses for each question and includes only the key points and not the supplementary detail.

1. What are the existing issues that you are dealing with in Southwestern grasslands?

- Fragmentation of grassland ecosystems, both biologically and politically (fragmented ownership), which increases the complexity of their management. We are losing grasslands to development as human communities expand into open space.
- Identifying desired composition and disturbances for grasslands (integrating social and economic factors into a landscape-level desired condition). There is the perception that grasslands have less value than forests.
- A lack of understanding of grassland ecology, the role of grazing in grassland ecosystems, and how grasslands respond to climatic variation.
- Monitoring is an issue due to a lack of consistency, a lack of resources (both people and funding), training challenges, and not having permanent photo monitoring points. The lack of personnel affects the level of management as well.
- A reduction in acreage and vegetation diversity of semidesert grasslands as a result of woody species encroachment and a shift from perennials to annuals. The increase in woody species is throughout all grassland types, and most respondents attributed the increase to overgrazing and lack of fire.
- A decrease in long-term soil productivity (assessed by Region 3 soil condition protocol) in regards to porosity, nutrient cycling, compaction, and erosion. Some unsatisfactory soils may never be able to be restored.
- A decrease in productivity in montane meadows, loss of meadows from forest encroachment, change of species composition (increase in Kentucky bluegrass), increase in elk populations causing damage (particularly when using winter range), lowering of the water table, and impaired or unsatisfactory soil condition. Some of these changes are due to livestock and elk grazing and recreation impacts such as RV camping. Soil recovery takes longer than vegetation recovery. Higher elevation meadows typically have better ground cover and are more resilient than lower elevation meadows, where the vegetation is more brittle and the soil quicker to erode.
- Maintaining the diversity of plants and animals in grassland ecosystems. Many grassland wildlife species appear to be declining in numbers and distribution. Some key species of concern include the Sonoran pronghorn, Mountain plovers, Northern Aplomado falcon, lesser-prairie chicken, black-tailed prairie dogs, and native nongame fish. Southwestern grassland ecosystems have received less attention than other ecosystems so it is probable that many other species warrant concern. Single-species management often creates conflicts in management activities and hampers our efforts to manage in an ecosystem context.
- Controversial management affecting grasslands include grazing, roads, and offroad vehicle use.
- Reintroduction of natural fire regimes. Concerns include smoke, public fear of fire, differences in landowners' views of fire, and difficulty in controlling grassland fires.

- Effects of drought, particularly extended droughts of several years.
- Spread of exotic or weed species into Southwestern grasslands.
- Land grant and Native American claims to land in northern New Mexico have resulted in difficult social and political issues, particularly grazing issues.
- Some conversions from shrub to grasslands in northern New Mexico done in the 1960s and1970s have since reverted.
- Lack of information, particularly habitat needs for designated threatened, endangered, and sensitive (TE&S) species.

2. What do you foresee as emerging issues in Southwestern grasslands?

- Increased need and pressure for regional planning to address issues.
- Urban encroachment, with ranchers selling their properties for development. Increasing wildland/urban interface conflicts will take many forms (fire concerns, an increase in nonnative plants, livestock conflicts, free-ranging dogs).
- Wildlife viability. The National Grasslands are becoming a repository for species declining on private lands. There is the potential for reintroductions of some species. White-tail prairie dogs may be in a more serious decline than black-tailed prairie dogs.
- Ecological viability is being threatened due to accelerated loss of soil productivity.
- Increasing populations of exotic or weed species.
- Greater public interest and demand for access to grassland areas, while being faced with limited recreation and public access facilities and infrastructure.
- Managing grasslands to reduce the risk of catastrophic fire. Using prescribed fire as a management tool is a challenge because of air quality concerns and the difficulty of fire planning with a variety of landowners.
- The drying of montane meadows from lowered water tables and loss of ground cover.
- Management tools: Best Management Practices (BMPs) for grazing, grassbanking, fencing, information management, and modeling.
- Implementing the Clean Water Act to address impaired streams.
- Recreation impact in grasslands, particularly in montane meadows. Offhighway vehicles are causing compaction of the soil.

3. What are the highest priority research needs for Southwestern grasslands?

- Establishment of reference areas and determination of reference conditions, not only flora but also fauna (such as Mirriam's elk). Identification of historic grassland distribution and conditions including disturbance regimes (fire frequency, ungulates, prairie dogs).
- Increasing depth of understanding of grassland ecology—disturbance patterns, the mechanism of recovery, effects of fragmentation on ecosystem function and plant species distribution, effects of changing climate (particularly drought), succession, linkages between factors, timing and density of utilization, the interrelationships between invasive and native plants, long-term implications of diversity, key indicators of grassland health, potential natural plant communities, the role of pollinators today relative to their historic role—and how they vary between ecotones. A study on what is causing the drying out of montane meadows.
- Understanding the tradeoffs between different composition mixes to help in the determination of desired conditions, and how/when to use disturbance events to move toward these conditions.
- Specific quantitative attributes for individual TE&S species habitat requirements. Identifying and understanding the role of small mammals in grassland ecosystems. Understanding of prairie dog habitat needs and reintroduction factors such as minimum numbers and distributional patterns. Social science research on dividing fisheries between sport fish and native nonsport fish, and fish ecology in general.
- A greater depth of understanding of the effects of management (grazing, timber, fire) on grassland ecology and priority species of concern (prairie dogs, mountain plovers). Determinations of the density of surface activities, such as the density of oil wells and their infrastructure. A study of the effectiveness of techniques to keep elk out of meadows.
- A values survey to capture the social aspects of why people think grasslands are special.
- Development of grassland ecosystem models and the understanding of the vegetation dynamics that feed the models.
- Exploring research opportunities on Tribal lands.

4. How can we achieve sustainability of the grasslands resource while addressing the wants and needs of people/communities?

Comments ranged from the general (understand and apply broad-scale ecologically sound approaches) to the specific (exclude cattle for 10 years, calculate stocking rate under the correct carrying capacity, then reintroduce fire and cattle). Conserving larger blocks of grasslands would make landscape-scale management possible, which is particularly important for wildlife habitat needs. Understanding the historic and present-day ecology of grasslands is necessary to develop desirable ecological structure, function, and processes that would sustain grassland resources and the needs of people. A public education program would help people understand how their needs and wants affect sustainability. Collaboration between academia, managers, and landowners is crucial. One mechanism to bring diverse interests together is through monitoring training.

5. What are the changes, both short term and long term (over 100 years), to Southwestern grasslands? Which changes are attributable to human influences and which are not?

- Fragmentation of larger contiguous grassland areas. Decrease in the amount of grasslands, as can be seen from historic photos. Development occurring at the landscape scale. Habitat fragmentation (for example, pronghorn antelope).
- Simplification of grassland types. Loss in quality of the grasses.
- Interruption of natural processes such as fire and water (flow patterns and the availability of surface water and groundwater), and a reduction in the nutrient and energy cycles from herbage removal. Drying out of montane meadows.
- Unsatisfactory soil conditions in many grassland areas and continued loss of soil and soil productivity.
- Increasing exotic species, particularly plants (accelerating). Some of these can not be eliminated, yet they change fire frequencies, wildlife needs, and so forth.
- Increasing juniper, pine, and shrubs in some grassland areas (rate of increase is slowing).
- Declining water tables and down-cut channels.

Where identified, changes were attributed to human influences with the exception of the increase in woody vegetation, which was viewed by some participants as a combination of human influence and climate.

6A. What do we need to be doing differently in the management of Southwestern grasslands?

- Implement larger scale management across regions. Continue broad-scale application of management based on ecological understanding at both local and regional scale. Having a separate grassland plan for each forest recognizes the uniqueness of grassland ecosystems.
- Take the time to develop relationships. Memoranda of understanding (MOU) are occurring at a larger scale. This takes dollars and people's time, and the ability to travel. Pursue cooperative and coordinated funding sources.
- Take an ecological approach rather than managing species by species as they are listed. Revise Forest Land Management Plans to reflect both an ecological approach while addressing individual species of concern. Consider revision of the Mexican Spotted Owl Recovery Plan as it applies to marginal forested habitats and historical savannah ecosystems.
- Develop staff working on the ground so they can effectively manage and monitor the resource. Find better ways to monitor grasslands given our dollar constraints.
- Improve management: Ensure plant recovery takes place each year rather than focus exclusively on grazing regimes. Manage offroad vehicle use. Monitor prescribed burning activities more closely. Address smoke management issues. Protect stable soils. Pursue more active restoration—not just a reduction in livestock numbers. Use more of an adaptive management approach rather than a "punitive" management approach.
- Recognize and incorporate the increasing emphasis and dependence on ecologically sound nature tourism to replace some of the failed economic efforts of the past.
- Increase public education programs on the value and role of grasslands.
- Map existing grasslands and compare to historic photos. Identify the desired pattern of grasslands within the Southwestern landscape. Establish photo points and frequency transects for monitoring.
- Work with the State Game and Fish Departments to develop realistic elk numbers, particularly in drought years.

6B. What do we need to continue doing?

- Take an ecological approach to management.
- Monitor and study prescribed and natural fire and woody plant control.
- Grassland restoration efforts.

- Use the Forest Interdisciplinary Team approach to handle NEPA work, thus freeing the Districts to continue their work.

7. What do you feel are the current trends in public perception and patterns of use related to Southwestern grasslands?

- Greatly increasing public interest and demand for a wide array of use opportunities, as well as enhancement of declining species and habitats. At the same time, there is a backlash toward State or Federal listed species. Even with increasing interest, the general public do not assign as great a value to grasslands as they do to other ecosystems and are poorly informed on grassland issues. However, the public is rapidly gaining a greater appreciation for open space and is generally becoming more aware of the environment.
- Increasing recreational use including offhighway vehicles driving and hunting. Most recreational use is concentrated in riparian areas. Hiking is likely to increase at a slower rate due to the constancy of vistas.
- Fear of loss of grazing use among permittees. Local support for subsistence grazing (northern New Mexico) with effective political lobbying.
- Backlash against prescribed burning.
- Further polarization between environmental and resource use nongovernmental organizations.

8. Who are your key stakeholders and why are they considered key stakeholders? How do you work with other grassland managers, stakeholders, and researchers, and what barriers do you face in initializing and maintaining cooperative efforts?

- Key stakeholders include: permittees, Ducks Unlimited, Playa Lake Joint Venture, N. Great Plains Initiative, Quivera Coalition. Barriers include lack of funding, current workload including litigation, lack of time needed to develop partnerships.
- Most significant stakeholder is private landowner as most grassland regions are predominately privately owned. Other big players are State wildlife agencies, State extension, agricultural research universities, NRCS, Fish and Wildlife Service (High Plains Partnership), Western Governors Association, rural development agencies, Nature Conservancy. Numerous efforts that often bring these folks together, such as Playa Lakes Joint Venture, Lesser Prairie Chicken Interstate Working Group, BTPD working groups. As a whole the key players and agencies seem much more united and working from the same viewpoint than occur with most regional applications in other parts of the Western United States, but sometimes it is hard to tie all the various group efforts together. One possibility would be to have an area serve as a "showcase."

9. If you had unlimited funding, what would you do differently to change or improve management of the Southwestern grasslands?

- Have more public education opportunities; do more monitoring work to put a grazing association in place; encourage more cooperative efforts across boundaries involving users and more of the public; make decisions that are ecologically based rather than economic based; implement more community based projects; focus more of our efforts on bringing about changes to improve grassland health.
- (1) More verified ecological understanding of effects of applied and natural processes on an ecoregion basis through research; (2) more public use/access facilities and programs/emphasis; (3) more funding to return landscapes to more natural ecological states (fire, ecologically based grazing, elimination of noxious/invasive plants such as the black locust, reintroduction of lost species, such as BTPD and lesser PC, where appropriate); (4) rural economic development emphasis on nature tourism; (5) financial incentives to private landowners to maintain or develop ecologically sound lands supported by income opportunities and/or public subsidy, such as the Farm Bill currently provides in CRP, WRP, WHIP, and development cost-sharing efforts, such as F&WS Partners for Wildlife, State programs, and so forth.
- Collection of baseline vegetation information NOW, rather than as current funding allows. Funding to support consolidation of grasslands, funding, and resources to support interagency/interorganization plans on landscape scale. This requires not only additional funding but also human resources and flexibility by all agencies/organizations to get past agency-specific barriers that make interagency implementation difficult. On our grasslands that are within our mountain districts, the biggest issue and area of need is pinyon-juniper encroachment in the small openings. Maintaining and restoring meadows through burning and/or mechanical thinning is needed.

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